Modelling urban ecosystem services:

Spatial patterns and implications for aspects of urban design

Meghann Amy Mears

Submitted for the degree of Doctor of Philosophy

Department of Animal and Plant Sciences
University of Sheffield

September 2010

A priest was in charge of the garden within a famous Zen temple. He had been given the job because he loved the flowers, shrubs, and trees. Next to the temple there was another, smaller temple where there lived a very old Zen master.

One day, when the priest was expecting some special guests, he took extra care in tending to the garden. He pulled the weeds, trimmed the shrubs, combed the moss, and spent a long time meticulously raking up and carefully arranging all the dry autumn leaves. As he worked, the old master watched him with interest from across the wall that separated the temples.

When he had finished, the priest stood back to admire his work. “Isn’t it beautiful”, he called out to the old master. “Yes”, replied the old man, “but there is something missing. Help me over this wall and I’ll put it right for you”.

The priest lifted the old fellow over and set him down. Slowly, the master walked to the tree near the center of the garden, grabbed it by the trunk, and shook it. Leaves showered down all over the garden. “There”, said the old man, “you can put me back now”.

Zen meditation story (kōan)

Acknowledgements

I am indebted to a great many people for helping me find my way through this PhD project. I am grateful to Phil Warren, Lorraine Maltby and Dave Raffaelli for their guidance, advice and patience throughout my PhD. Rob Holland helped with many issues that I encountered, both practical and philosophical. My work has benefited from discussions with Thomas Elmqvist and other staff and students at the Stockholm Resilience Centre. I would also like to thank Andrew Beckerman for his mentorship both before and during my PhD.

I am grateful to a number of people for freely providing data. Andrew Elleker at the Sheffield City Council provided AIRVIRO model output and Sheffield meteorological data; Jon Groocock gave permission to use data from, and information about, his personal weather station (http://www.sheffieldweather.co.uk/index.htm); the South Yorkshire Archaeology Service provided the Historic Environment Character dataset; and Anna Patrick at Ordnance Survey Research provided the OS MasterMap data. The Sorby Natural History Society also provided data that was not eventually used.

I am especially thankful to Anna, Marian, Cat, Emily, and my other friends and colleagues in the Aquatic Ecology group for their friendship and support throughout my PhD. Last but certainly not least I would like to thank the staff at the University Health Centre, and of course my Mum, Dad and sisters for keeping me going.

Summary

Urbanisation causes profound changes in natural ecosystems, often reducing or eliminating ecosystem services, i.e. benefits from nature to human well-being. Cities can nevertheless contain a substantial amount of greenspace, which has the potential to continue to provide such services. Knowledge of how to manage urban ecosystems from this perspective has the potential to improve citizens' quality of life and urban sustainability.

This thesis presents six urban ecosystem service models, namely: reduction of air pollution; heat island mitigation; stormwater runoff reduction; carbon storage; opportunities for cultural ecosystem services in public greenspaces; and provision of habitat for biodiversity. These are explored to examine the nature and spatial pattern of ecosystem service provision in an urban system, using the city of Sheffield, UK as a case study.

Key results from this are: (1) There is a general increase in ecosystem service production from the urban centre outwards, although there some service hotspots in the urban centre. (2) The production of different services is not spatially co-incident. (3) Perceived spatial pattern of service provision is dependent upon the spatial resolution used for analysis. (4) Certain features of urban morphology can improve levels of the modelled ecosystem services. (5) There is significant socioeconomic inequity in access to ecosystem services, with unskilled manual workers, multicultural communities, and young households being particularly deprived. (6) Combining information from these analyses allows identification of neighbourhood morphologies that provide higher levels of ecosystem services to the most deprived groups, with housing that they can still afford.

Overall, this study shows the potential for insights into urban ecosystem service provision from to be gained from tractable spatial models, and that these could provide starting points for enhancing the well-being of urban residents through appropriate urban design.

Table of contents

1. General introduction

1.1. Introduction

There is a continuing migration away from rural areas, with more than half of the world's population now living in cities (United Nations Population Fund 2007). In order to accommodate expanding populations, cities physically grow and larger areas of land become urbanised. Urbanisation has been called the "endpoint of landscape domestication…in which every element of the environment has been consciously or unconsciously selected to accord with human desires" (Kareiva et al. 2007). In other words, urbanisation involves the replacement of those aspects of the environment that are considered undesirable by urban planners and decision makers (Kareiva et al. 2007).

These changes can damage the ability of landscapes to function normally, by altering the biophysical infrastructure of the environment (Haines-Young and Potschin 2008); the evidence for environmental degradation of this type is reviewed in Section 1.2. Altered environmental functioning has consequences for human prosperity and, moreover, human well-being (Figure 1.1). This is because the functioning of the environment is responsible for a wide range of benefits to humans, such as the production of food and raw materials, regulation of the climate and of air and water quality, and the existence of landscapes that are aesthetically pleasing (De Groot et al. 2002, Wallace 2007, Haines-Young and Potschin 2008).

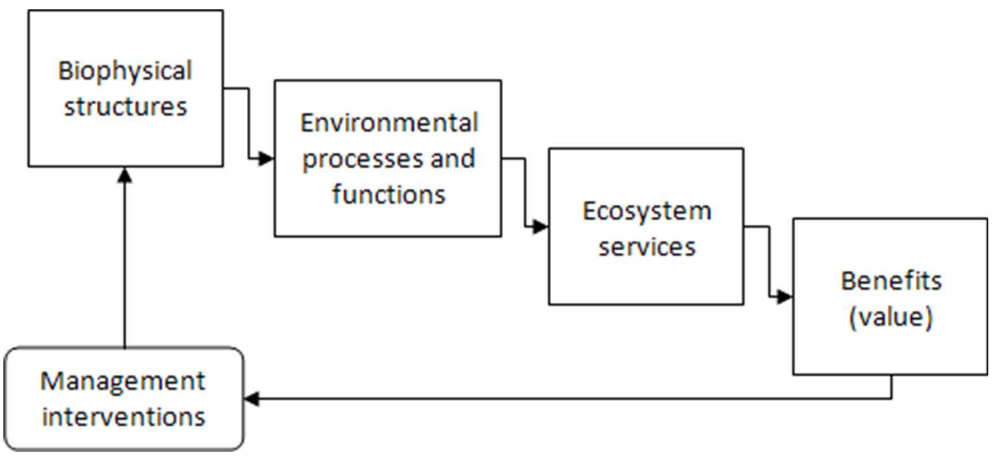


Figure 1.1. How ecosystem services are produced and influenced. Adapted from Haines-Young and Potschin (2008).

As indicated in Figure 1.1, the processes whereby humans obtain benefits from the environment are termed "ecosystem services" (Millennium Ecosystem Assessment 2003). Daily et al. (1997) define ecosystem services as the "conditions and processes through which natural ecosystems, and the species that are a part of them, help sustain and fulfil human life". Ecosystem services satisfy a variety of human needs and values, including the provision of adequate resources, regulation of a benign physical and chemical environment, protection from predators, diseases and parasites, and cultural fulfilment (Wallace 2007); Table 1.1 lists a range of examples.

These issues are a re-expression of well known anthropogenic impacts on the environment and the fact that natural resources are sometimes fragile and strictly finite. The new opportunity afforded by the concept of ecosystem services, however, is an easily understood and graphic illustration of the dependence of man on nature and vice versa, making explicit the links between anthropogenic impacts, the state of the environment and specific aspects of human welfare.

Table 1.1. Categories and examples of ecosystem services, and examples of the biophysical structures, processes and functions underpinning ecosystem services. Adapted from Wallace (2007).

Human values satisfied	**Examples of ecosystem services**	**Examples of underpinning structures, processes and functions**
Provision of adequate resources	Food Oxygen Potable water Energy for cooking Raw materials	Biological regulation Biomass Climate regulation Disturbance regulation Nutrient regulation Pollination Production of raw materials Soil formation and retention Waste regulation Water flows Biodiversity Soils and geomorphology Water Air Energy
Protection from other species	Protection from predators, diseases and parasites	
Regulation of a benign physical and chemical environment	Benign environmental regimes: Temperature Moisture Light Chemicals Disturbances/disasters	
Cultural fulfilment	Access to resources for: Social, spiritual & philosophical contentment & development Recreation & leisure Meaningful occupation Aesthetics	

Furthermore, the concept of ecosystem services lends itself to a place-based approach (Haines-Young and Potschin 2008): ecosystems are understood as holistic units within landscapes, the components of which are interconnected (Pickett and Cadenasso 2002). Environmental management processes should also adopt a place-specific approach (Christensen et al. 1996). The concept of ecosystem services therefore has the potential to be used as a tool in environmental management and policy decision making (Chan et al. 2006, Nelson et al. 2009).

To date, ecosystem services have been studied less in urban contexts than in natural and semi-natural contexts. This follows the general trend in ecology to study less anthropogenically influenced systems (Pickett et al. 2001, Niemelä et al. 2009). Nevertheless, the concept of city as ecosystem is now recognised as valid (Grimm et al. 2000, Pickett et al. 2001, Niemelä et al. 2009). Ecological and environmental processes occur in cities just as in pristine environments, albeit in different combinations and at different rates; the key difference between these types of ecosystem is that anthropogenic infrastructure and processes also exist as interconnected components of urban, but not pristine, environments (Grimm et al. 2000, Pickett et al. 2001, Niemelä et al. 2009). These urban ecosystems can be important providers of ecosystem services to residents (Section 1.2).

The pervasiveness of human influence in urban ecosystems, together with the reliance of those humans on the state of the environment for their welfare, makes a strong case for managing the urban environment as an ecosystem of which humans are a component (Christensen et al. 1996, Grimm et al. 2000, Pickett et al. 2001). The ecosystem services concept, with its explicit links between humans and the environment, therefore has a potential to contribute significantly to urban management that has not yet been fully exploited. The central aim of this thesis is to explore this potential. This chapter will first discuss some of the environmental issues caused by urbanisation to clarify the idea of urban ecosystem services, before looking at what is required of the ecosystem services concept in order to contribute to urban planning and decision making. Having established the context of this thesis, the research aims and objectives will be set out. Finally, the case study area and the structure of the rest of the thesis will be introduced.

1.2. Urbanisation and ecosystem services

Ecosystem service research began in earnest in the mid 1990s and took off following the Millennium Ecosystem Assessment, which started in 2001. Research into urban ecosystem services began at the same time, but at a slower pace. Much of the research that has occurred has concerned itself with remnant fragments of ecosystem types that are known to be valuable providers of ecosystem services, especially wetlands and forests (e.g. Ehrenfeld 2004, Hogan and Walbridge 2007, Nowak et al. 2008), or with urban parks (e.g. Jim and Chen 2006b, Fuller et al. 2007). There are fewer studies investigating other types of "green" ecosystem (e.g. gardens, roadside verges, allotments), especially types that are typically found in small land parcels, or looking across the urban matrix of land types/uses. There are, however, some examples (e.g. Gill 2006, Tratalos et al. 2007).

This section discusses some of the key findings of urban ecosystem service research. First, several terms must be defined and explained.

1.2.1. Definitions

Several authors have produced different schemata, similar to Figure 1.1, describing how ecosystem services relate to the environment and to humans (Raffaelli et al. 2007). Authors variously recognise biophysical structures, processes, functions, services and benefits to be important, but these terms do not necessarily mean the same things for different authors (Raffaelli et al. 2007). It is therefore necessary to define what is meant by these terms in this thesis.

The form of Figure 1.1, which is essentially the same as that of Haines-Young & Potschin (2008) was chosen because it clearly articulates all the links that give the ecosystem services concept its potential for application in management: from biophysical infrastructure to processes, to the delivery of a service, and finally to benefits to humans. Also importantly, this schema implies that there is no service without a beneficiary, and that biophysical structures and functions that do not in themselves provide a benefit are not ecosystem services.

Biophysical structures and *environmental processes and functions* are defined here as the components of an ecosystem and the interactions between those components respectively. The *production* of an ecosystem service refers in this thesis to the net results of these two boxes in Figure 1.1: the generation of a flow of the entity or process that provides the service, by the biophysical structure. For example, if the ecosystem service is flood control by reduction of surface runoff during storms, then the process providing the ecosystem service is the abstraction and infiltration (processes) of precipitation to soils and vegetation (biophysical infrastructure). In this particular case, the *actual* production (i.e. actual abstraction and infiltration given a particular storm event) may be less than the *potential* production (the total capacity for abstraction and infiltration).

The *ecosystem service* itself is considered in this thesis to be intangible: it is the result of particular infrastructure/processes, but is in itself valueless. It is the supply of an ecosystem service to beneficiaries that gives *value*. Typically, researchers have attempted to place a value on ecosystem services in economic terms, and indeed some have argued that this is critical to making ecosystem services equivalent to other considerations in decision making processes (Daily et al. 2009). The economic approach is however at risk of marginalising moral issues concerning the benefit of ecosystem services to human welfare. This thesis therefore considers both the welfare and economic value of ecosystem services.

The *supply* of an ecosystem service, as the term is used in this thesis, depends on spatiotemporal characteristics that determine how a produced ecosystem service 'flows' to its beneficiaries. Supply is represented in Figure 1.1 by the arrow leading from "ecosystem services" to "benefits (value)". Costanza (2008) describes and provides examples of five different ways in which an ecosystem service may flow spatially to its beneficiaries, which are listed here. *Global non-proximal ecosystem services* do not depend on the proximity of the site of production to the beneficiaries; climate regulation by carbon storage and sequestration is a good example of this category. *Local proximal ecosystem services*, such as disturbance regulation, waste treatment and biological control services depend on the beneficiary being close to the site of production. *Directional flow related ecosystem services* are those which are related to a downstream flow to the point of use, such as water-related services and erosion and sediment control. *In situ ecosystem services* provide a service at the point of production: resource production services fall into this category (although the resources might be transported by humans for use elsewhere). Finally, *user movement related ecosystem services* depend on the movement of people to particular sites; these services generally fall into the cultural category (see Table 1.1).

Similar categories of temporal characteristics could also be defined. For example, ecosystem services with a directional flow will take time to flow to the beneficiaries. Carbon storage and sequestration now will have long-term benefits, while disturbance regulation ceases to be of benefit once the regulation stops.

Urban ecosystem services are produced according to the general schema shown in Figure 1.1, where the biophysical structure is in an urbanised location and includes anthropogenic infrastructure. Figure 1.1 does not show the true complexity of ecosystem services, however. Ecosystems are in general multi-functional, providing a range of different types of ecosystem services, and different management options can provide different suites of ecosystem services (De Groot 2006, Rodríguez et al. 2006). The interconnectedness of components within an ecosystem means that within the black boxes representing "biophysical structures" and "environmental processes and functions" there will be interactions and feedbacks. Figure 1.1 also does not show that over-exploitation of ecosystem services can negatively affect the same or other services by damaging their biophysical infrastructure, nor that trade-offs are inherent between ecosystem services (De Groot 2006, Rodríguez et al. 2006). The following discussions of current research into urban ecosystem services illustrate some of the consequences of trade-offs and over-exploitation.

1.2.2. Urbanisation, land use and land cover

Kline (2006) noted that urbanisation inherently involves the replacement of "open space" land with developed land uses, and that, furthermore, because of the marginal nature of demand for such land it is possible for considerable damage to remnant natural environments to accrue before local preservation becomes a major issue. However, it seems that only one study has specifically quantified the changes to landscapes that occur as a result of urbanisation. This study compared the inner, middle and outer urbanised regions of five UK cities, and found that the proportions of land covered by non-sealed surfaces, greenspace, gardens and trees did indeed decrease towards the inner areas (Tratalos et al. 2007).

The loss of greenspace is significant because, although cities can indeed be considered as ecosystems, it is recognised that the biophysical infrastructure necessary to support ecosystem service production is primarily found in landscapes with greenspace components (Handley et al. 2003, Jim and Chen 2006a, Scottish Executive Development Department 2008). To clarify, the term urban greenspace refers here (as in most definitions) to lands primarily consisting of permeable, i.e. non-sealed surfaces (Handley et al. 2003, Swanwick et al. 2003, Jim and Chen 2006a, Scottish Executive Development Department 2008). This might include, but is not necessarily limited to, public and private parks and gardens, amenity greenspace (e.g. land used for separating buildings or for informal or social activities), playgrounds and sports grounds, green corridors (such as river or canal corridors or disused railway lines), residual natural or seminatural greenspace, allotments and burial grounds (Swanwick et al. 2003).

Many authors have noted the importance of these land covers/uses for the production of urban ecosystem services (see references in this section). Table 1.2 shows a non-exhaustive list of these ecosystem services, as well as some of the biophysical structures and environmental processes provided by urban greenspace. Although not all of these services will be relevant in all cities, the importance of greenspace land uses and land covers in contributing to human welfare is obvious. This

Table 1.2. Examples of ecosystem services and biophysical infrastructure that may be provided by urban greenspace. Sources: Colding et al. (2006), Jim and Chen (2006a).

Human values satisfied	**Ecosystem services**
Provision of adequate resources	Potable water Production of materials to be used for: Food Ornamental resources (e.g. flowers) Fire wood
Protection from other species	Biocontrol of pests & disease vectors by species in greenspace
Regulation of a benign physical and chemical environment	Air pollution filtration Noise reduction Regulation of microclimate (temperature, winds etc.) Carbon storage and sequestration Surface water drainage
Cultural fulfilment	Access to resources for: Aesthetic satisfaction Inspiration Nature education Recreation Social relations Sense of place
Biophysical structures, processes and functions	Habitat for flora and fauna Soil formation and retention Seed dispersal Pollination Water cycling Nutrient retention (in water bodies)

is a fact that has been recognised by government agencies, which in recent years have recommended development of urban greenspace networks a planning priority (Handley et al. 2003, Scottish Executive Development Department 2008). Nevertheless, this does not change the fact that urbanisation by definition reduces the amount of space available for ecosystem service production (Kline 2006).

1.2.3. Wetland and freshwater habitat ecosystem services

Wetlands (including riparian habitat) and rivers are especially valuable providers of ecosystem services in urban areas, contributing to a wide variety of ecosystem services. Forming part of the water drainage network of a city, wetlands and rivers are able to store and remove precipitation, contributing to flood protection (Bolund and Hunhammar 1999, Ehrenfeld 2004). There is evidence that the river network plays a more important role in removing storm runoff in more urbanised areas (Paul et al. 2006). In addition to pollutants and waste products received in drained water, waste (including sewage) is often deliberately disposed of in wetlands and rivers (Ehrenfeld 2004, Singer and Battin 2007).

Although wetlands and rivers are indeed able to store water and attenuate sewage and pollution, over-reliance on, and abuse of these ecosystem services can damage the biophysical infrastructure and impair the ability of the ecosystem to perform these and other services (Bolund and Hunhammar 1999, Ehrenfeld 2004, Hogan and Walbridge 2007). High levels of anthropogenic impacts have been associated with reduced capacity of wetlands and rivers to retain excess nutrients, increased soil erosion, changes to the biodiversity and trophic structures of these habitats, and a suite of other hydrological, chemical, geomorphological changes (Chadwick et al. 2006, Paul et al. 2006, Hogan and Walbridge 2007). In fact, the detrimental effects of urbanisation on rivers are so characteristic that they have been termed the "urban river syndrome" (Paul et al. 2006).

This degradation of ecosystem services is often compounded by losses of riparian vegetation, which is a common occurrence during urbanisation (Ozawa and Yeakley 2007). Furthermore, wetlands have historically been considered as wastelands and breeding grounds for undesirable species (e.g. mosquitoes), resulting in disregard for their value; this means that although these habitat types are challenging to develop over, significant areas have been lost (Ehrenfeld 2004, BenDor et al. 2008).

For all these reasons, it is common for the ability of urban wetlands and rivers to provide ecosystem services to be much reduced in comparison to those that are less impacted. As an example, the Sanyang wetland in Wenzhou city, China, contributes to the production of food, water and raw materials, flood protection and drought recovery, water pollution filtration, carbon storage and provision of habitat for biodiversity (Tong et al. 2007). However, failure to correctly value these ecosystem services means that the value of the current provision of services is only 10.5% of the potential, if the wetland were restored to pristine condition (Tong et al. 2007). In addition to the ecosystem services already mentioned, wetland and river habitats in good condition are able to contribute to the reduction of air pollution, microclimate regulation and noise reduction, and provide opportunities for recreation and aesthetic satisfaction (Bolund and Hunhammar 1999).

1.2.4. Soil ecosystem services

Urbanisation has profound effects on soils. In the process of urban development, large areas may be covered by impervious surfaces, which causes precipitation to run straight off the surface rather than be intercepted or abstracted, and prevents the emission and sequestration of gases such as carbon dioxide and nitrogen oxides (Lorenz and Lal 2009). Even where the soils are not sealed, urban soils are likely to be compacted, to have increased input of pollutants and nutrients, and to have an altered microclimate as a consequence of increased temperatures and changes to the cloud and hydrologic regimes (Pavao-Zuckerman 2008, Lorenz and Lal 2009).

As a result, urban soils have a fundamentally different biochemistry compared to undisturbed soil of the same type. There are a few general patterns, such as soil carbon stocks that are typically higher than disturbed soil in nearby rural areas but lower than undisturbed local soils, and reduced microbial metabolism and decomposition rates due to heavy metal pollution (Pavao-Zuckerman 2008, Lorenz and Lal 2009). However, the exact nature of urban soils depends heavily on the properties of the undisturbed local soil type, as well as local variables such as climate, land cover, previous land uses, time since establishment of current land use, and management factors such as fertilisation and irrigation (Pouyat et al. 2006, Byrne et al. 2008, Pavao-Zuckerman 2008, Lorenz and Lal 2009). These complex patterns are presently poorly understood.

1.2.5. Cultural ecosystem services

Most studies of cultural ecosystem services in an urban context focus on public parks. Jim and Chen (2006a, 2006b), using Guangzhou in China as a case study, found that residents visited urban parks for a variety of reasons including relaxation, quietude, exercise, and appreciation of nature and aesthetics; and that at least 70% of interviewees rated parks as important or very important for “places for recreational activities”, “opportunities to know and contact nature”, “cultural and educational connotation” and “aesthetic enhancement”. These findings, together with the fact that Chinese citizens typically have a high willingness-to-pay for the creation, maintenance and use of urban parks (in China it is typical for parks to charge an entrance fee), indicate a recognition and appreciation of the benefits obtained from such greenspaces (Jim and Chen 2006a, Jim and Chen 2006b, Chen and Jim 2008).

Tzoulas et al. (2007) review evidence for cultural ecosystem services provided by urban greenspace from a predominantly Western perspective, finding that much the same services are provided and valued. Amongst the identified benefits of greenspace use or the existence of greenspace were emotional release and restoration, relaxation, improvement of attention, aesthetic appreciation, and increased physical health and longevity. Thus it appears that well maintained urban greenspaces are capable of providing a wide range of cultural ecosystem services.

1.2.6. Biodiversity and associated ecosystem services

There exists a vast literature investigating the effects of urbanisation on biodiversity and ecology. This relationship is complex as there are many environmental changes along urban to rural gradients that affect biodiversity in different ways. For example, landscape fragmentation reduces the total space available for biodiversity and precludes species that require large contiguous areas of habitat (McKinney 2008). However, the greater local habitat heterogeneity found in urban areas means that biodiversity (especially beta biodiversity) might increase (McKinney 2008). High rates of introductions of non-native species can also increase diversity, but at the same time risk loss of native diversity due to outcompetition (McKinney 2008).

Furthermore, changes in microclimate, hydrology, pollution levels and disturbance regimes caused by urbanisation profoundly alter the ecological niche space, meaning that community composition, as well as species diversity, is liable to change (Pickett et al. 2009). In particular, there are concerns that urbanisation promotes homogenisation of communities between different cities, with particular "synanthropic" species, which are often non-native, being most likely to persist in large numbers in the urban environment (McKinney 2006).

Birds, which are important for ecosystem services through their contribution to seed dispersion, pollination and insect control (Schlesinger et al. 2008), have been especially well-studied and illustrate many of these patterns. Meta-analyses and reviews indicate that local factors are the major determinants of urban bird species richness, with the species richness of the surrounding area being less important (Clergeau et al. 2001, Evans et al. 2009). It is local habitat features such as high structural complexity of the habitat, habitat diversity, supplementary feeding and low levels of disturbance by humans that are responsible for increasing species richness in urban areas (Evans et al. 2009). Consequently, bird species richness often peaks at intermediate levels of urbanisation due to colonisation by synanthropic species (e.g. more aggressive species), but without pushing out more sensitive species (Evans et al. 2009).

The evidence also suggests that biotic homogenisation occurs, i.e. that urbanisation cause bird communities to be more similar between cities than expected due to similar patterns of environmental change (Evans et al. 2009). Furthermore, the stress of the urban environment can cause behavioural changes in some species (e.g. changes in singing time or volume, less time spent feeding due to increased vigilance), and the types of food available are often less nutritious, especially for nestlings; as a result, reproduction tends to be less successful compared to non-urban populations (Slabbekoorn and Ripmeester 2008, Chamberlain et al. 2009).

Although these meta-analyses obscure species- and city-specific results, they illustrate the complex but generally detrimental (at least to the native community) nature of the effects of urbanisation on bird biodiversity – especially at higher levels of urbanisation. Fewer studies have investigated other groups of species, but a recent meta-analysis indicates that species richness of invertebrates and non-avian vertebrates tends to decrease from peri-urban areas with less than 20% impervious surface area to suburban areas (20-50%), but does not further decrease at high levels of urbanisation (McKinney 2008). In contrast, plant diversity is greatest in suburban areas and lowest outside the urban areas – but this may be because non-native plant species tend to be found in especially high numbers in urban gardens, parks and other landscaped areas (McKinney 2008). For all groups there is variation in the responses of individual species (McKinney 2008), but again, the general conclusion is that urbanisation is detrimental to the native community of organisms.

Despite the effects of urbanisation on biodiversity, urban greenspaces are often able to provide valuable habitat that sometimes supports rare species. For example, the National Urban Park in Stockholm, Sweden, hosts 75% of species recorded in the geographic region in 1% of the area, including 1200 Coleoptera, 250 bird, 60 red listed insect, 32 red listed fungi and 20 red listed vascular plant and vertebrate species (Barthel et al. 2005). Urban golf courses, which typically provide large contiguous blocks of greenspace, are often hotspots of biodiversity and sometimes include remnant patches of rare native habitat types (Colding and Folke 2009). Even smaller areas of greenspace, such as private gardens, can provide valuable habitat for both plant and animal diversity (Smith et al. 2006). Furthermore, it appears that different types of urban greenspace provide habitat for different species, indicating that site-specific management factors and land use/cover patterns can influence local biodiversity: for example, bird and bumblebee communities differ between allotment gardens, cemeteries and urban parks in Stockholm, Sweden (Andersson et al. 2007).

Indeed, management can influence many properties of greenspace patches that are important for biodiversity. Garden et al. (2006), who reviewed urban ecological dynamics in Australia, suggest that a number of characteristics operating at within-patch, patch and landscape scales have consequences for the biodiversity within them. These characteristics are listed in Table 1.3.

Table 1.3. Urban greenspace patch characteristics operating at differential spatial scales that can have consequences for biodiversity. Adapted from Garden et al. (2006).

Spatial scale	**Habitat characteristics**
Within-patch (e.g. <1 ha)	Vegetation composition and structure Ground cover type and proportion Soil compaction Nutrient levels
Patch (e.g. 1-100 ha)	Size and shape Area:edge ratio Distance to other landscape features (e.g. patches of similar type, river, road) Time since isolation
Landscape (e.g. >100 ha)	Total habitat area Habitat fragmentation and connectivity Density of human developments

Providing suitable conditions for biodiversity is important in the context of this thesis because of the role that biodiversity is believed to play in contributing to ecosystem service production. This is in addition to the psychological benefits directly provided by exposure to a diversity of species and habitat types, which in itself constitutes a cultural ecosystem service (Fuller et al. 2007, Lindemann-Matthies et al. 2010). There is evidence from both experiments and observations in real urban parks that people can to some degree perceive species richness of plants, birds and butterflies, and that high species diversity (both perceived and actual) is associated with self-reported psychological benefits including aesthetic appreciation, opportunity for reflection, and a sense of identity and continuity of identity across time established through association with a particular greenspace (Fuller et al. 2007, Lindemann-Matthies et al. 2010).

The following line of reasoning supporting the idea that high biodiversity is good for the production of ecosystem services is built on an assumption, namely that increasing or at least maintaining levels of ecosystem services in general (i.e. without reference to specific services) is desirable. This seems a reasonable assumption, since ecosystem service production contributes to many characteristics of the idealised desirable city (e.g. low levels of pollution, low risks of environmental disturbance, reasonable climate).

The hypothesised contributions of biodiversity to other ecosystem services are indirect. Every species interacts with other species and the abiotic environment, i.e. it performs certain biogeochemical, hydrological and ecological processes (Walker 1992, Naeem 1998, Folke et al. 2004). For example, the primary producers in an ecosystem produce biomass, while detritivores play an important role in returning nutrients from organisms to the soil (Naeem 1998). The term "functional traits" is often used to describe the processes performed by an organism.

Many of these environmental processes are related to ecosystem service production (Balvanera et al. 2006). For example, primary production can produce materials for food, fuel, etc., and the production of plant root biomass can also reduce soil erosion (Balvanera et al. 2006). In some cases, multiple species are required to optimise the production of an ecosystem service via environmental processes. For example, a plant community composed of species of different rooting depths might better stabilise the soil against erosion, i.e. the species have complementary resource use (Jiang et al. 2009). A similar phenomenon, facilitation, refers to different processes performed by multiple species increasing the production of an ecosystem service, such as when a nitrogen fixing plant species enables a species with high potential growth rate to fulfil that potential (Jiang et al. 2009).

It therefore follows that preserving functional diversity (i.e. preserving species performing different environmental processes) is important for the production of ecosystem services (Walker 1992, Folke et al. 2004). Furthermore, given that urban ecosystems are especially prone to environmental disturbances (such as those described in the previous sections) that may cause local extinctions, it seems likely that preserving biodiversity within functional groups (groups of species with similar functional traits) may provide some stability, or resilience, of ecosystem service production in the case of temporal change (Walker 1992, Folke et al. 2004). Thus, at least in theory, preserving biodiversity is desirable for the preservation of ecosystem service production.

It is unclear at present whether the preservation of *native* biodiversity is especially important. There does not appear to have been any research into whether synanthropic communities, or communities including invasive species characteristic of urban environments, show similar functional diversity to native communities. If synanthropic or invasive communities have lower functional diversity, however, this would indicate that native biodiversity is critical.

The past decade or so has seen extensive research into the role that that biodiversity plays in ecosystem functioning, a large component of which can be directly related to ecosystem services, yet considerable contention remains to be resolved (Thompson et al. 2005, Balvanera et al. 2006, Cardinale et al. 2006, Hector et al. 2007, Caliman et al. 2010). There have been many biodiversity manipulation experiments, in which microcosms (e.g. bacteria), mesocosms (e.g. greenhouses) or field plots are used to create communities of varying biodiversity (either in terms of species or functional richness) from a set species pool (Balvanera et al. 2006). A recent meta-analysis of these studies (summarised in Table 1.4) found that, for most environmental processes that could be linked to ecosystem services, there was on average a positive relationship between biodiversity and the environmental processes (Balvanera et al. 2006). In summary, increasing biodiversity at a given trophic level increases productivity at that and sometimes at other trophic levels; increased plant or mycorrhizal biodiversity improves soil erosion prevention; high biodiversity increases some (but not all) processes related to control of nutrient cycling; and high plant biodiversity improves the regulation of biological diversity (in terms of diversity higher up the trophic chain and

Table 1.4. Effects of biodiversity manipulation on ecosystem properties related to ecosystem services. Refers to manipulation of primary producer biodiversity except where otherwise stated in the "response" column. (Source: Balvanera et al. 2006)

Ecosystem service	Ecosystem property	Response to increasing biodiversity
Productivity	Primary producer abundance	Increase (plant, mycorrhiza)
		No change (primary consumer)
	Primary consumer abundance	Increase (plant, primary consumer)
	Secondary consumer abundance	No change
Erosion control	Plant root biomass	Increased (plant)
		No change (mycorrhiza)
	Mycorrhiza abundance	Increased (mycorrhiza)
Nutrient control	Decomposer activity	Increased (plant, decomposer)
	Plant nutrient concentration	Increased (mycorrhiza)
		No change (plant)
	Nutrient supply from soil	No change (plant, multitrophic)
Regulation of biological diversity	Primary consumer diversity	Increase
	Plant disease severity	Decrease
	Decomposer diversity	Increase
	Invader fitness	Decrease
	Invader diversity	Decrease
Stability	Consumption resistance	Increase
	Invasion resistance	Increase
	Drought resistance	No change
	Resistant to other disturbances	No change
	Natural variation	No change

resistance against pests and invaders) and the resilience of ecosystem services to disturbances that are biological in origin, but not to abiotic disturbances (Balvanera et al. 2006).

This meta-analysis supports a long-held consensus in the result of biodiversity manipulation experiments: that a minimum set of species is needed for ecosystem service production in the absence of disturbances, and that a larger number of species probably aids ecosystem service stability in changing environments (Loreau et al. 2001). A separate meta-analysis found that, in general, these relationships could be explained by the "sampling effect", i.e. the fact that more biodiverse communities are more likely to contain species performing high process rates: the most diverse polyculture tends not to outperform the best performing monoculture (Cardinale et al. 2006).

Of course, the environmental processes studied so far represent only a small fraction of the total number of ecosystem services. Furthermore, there is considerable debate about the applicability of the conclusions of these experiments at landscape scales

(Loreau et al. 2001, Schmid 2002, Thompson et al. 2005, Hector et al. 2007, Jiang et al. 2009). In these experiments, biodiversity is the only factor driving process rates, whereas in natural ecosystems environmental factors determine biodiversity and thus have an indirect (as well as a direct) influence on processes (Loreau et al. 2001). Community dynamics in experimental systems are unrealistic, with the importance of complementarity and the sampling effect being exaggerated, because community evenness tends to be greater than is found in nature and the typically short timespan of experiments limits competitive interactions and exclusion (Thompson et al. 2005, Jiang et al. 2009). Few studies to date have included realistic levels of trophic complexity, and there has been a strong habitat bias toward studies of productivity in grassland ecosystems (Loreau et al. 2001, Caliman et al. 2010). There is also a far greater level of heterogeneity in natural environments, although it is unclear what the implications of this are for extrapolating from experimental results (Jiang et al. 2009).

Some authors thus suggest that the results of biodiversity manipulation experiments are irrelevant for more complex natural ecosystems, especially when realistic patterns of extinction are considered (Thompson et al. 2005, Jiang et al. 2009). Indeed, although this is only a single ecosystem service, results from natural ecosystems typically (but not always) find a hump-backed relationship between productivity and biodiversity of primary producers, with the highest levels of biodiversity being found at intermediate levels of productivity (Schmid 2002, Thompson et al. 2005). On the other hand, it is also argued that the complexity inherent in natural ecosystems is precisely the reason why biodiversity manipulations in homogeneous environments need to be performed: to isolate the effect of biodiversity when all other factors are controlled for (Hector et al. 2007). Moreover, one recent article used theoretical and empirical evidence to argue that biodiversity manipulations underestimate the importance of biodiversity for environmental processes where multiple processes (or ecosystem services) are provided in a spatially and temporally complex ecosystem or where patterns of extinction are non-random (Duffy 2009).

Thus the weight of current evidence strongly indicates that local levels of biodiversity are important to the production of ecosystem services. In urban environments, where rates of local extinctions and local disturbances are particularly high (Ehrenfeld 2004, McKinney 2006, Heckmann et al. 2008, Schlesinger et al. 2008, Evans et al. 2009), it follows that regional levels of biodiversity are also important to maintain a supply of colonisers for disturbed and/or changing environments. Therefore, in order to maintain the provision of ecosystem services in urban areas, the evidence presented here indicates that it is necessary to provide appropriate infrastructure for biodiversity at local and regional scales.

1.3. Using the ecosystem services concept in urban planning

Despite the long-standing recognition that ecosystem management should integrate knowledge of the socioeconomic and biophysical systems in order to sustain the provision of ecosystem services (Christensen et al. 1996, Shandas et al. 2008), few studies to date have investigated the extent to which this principle is used in urban ecosystem management situations (Shandas et al. 2008). Two studies that have performed such investigations, however, both identified a need for tools to assist urban planners in incorporating scientifically founded ecosystem management into decision making (Yli-Pelkonen 2006, Shandas et al. 2008).

1.3.1. Urban morphology

One reason why the inclusion of ecological information into urban ecosystem management is important is that there is evidence that urban morphology, which is largely determined by urban planning processes, is an important factor related to ecosystem service provision (Whitford et al. 2001, Bierwagen 2005, Tratalos et al. 2007). Urban morphology describes the form, or spatial structure, of human settlements. A basic component of urban morphology is the amount of greenspace found in the urban environment, which, predictably, influences the ability of an area to perform ecosystem services such as heat island mitigation and flood protection (Whitford et al. 2001, Tratalos et al. 2007).

There are also more subtle ways in which urban morphology influences biodiversity and ecosystem services. For example, a study of the connectivity of greenspace using simple metrics quantified from land use maps found that connectivity is determined at the city scale by the number of distinct “urban patches” and the extent of those patches (as described by the radius of gyration, or the average distance that can be travelled in one direction from the centre of a patch before encountering its edge), which has implications for the persistence of biodiversity (Bierwagen 2005).

To date, most research into urban form has been from the perspective of "sustainable" urban design, as evidenced by two major works in the field (Williams et al. 2000a, Jenks and Jones 2010). The provision of ecosystem services is implicit in the stated definition of "sustainable" (Williams et al. 2000b):

> "A form [i.e. urban morphology] is taken to be sustainable if it: enables the city to function within its natural and man-made carrying capacities; is 'user-friendly' for its occupants; and promotes social equity."

While these ideals are central to the ethos of ecosystem service science, most of the work described in Williams et al. (2000a) takes a far less ecological viewpoint than the research discussed thus far. However, while some useful insights have emerged from this work, there is a case for approaching the subject of urban morphology from a different perspective. At present, the ecological perspective that makes ecosystem services central and explicit is relatively nascent, with general hypotheses and conceptual models in place but few empirical results (Alberti 2005). Knowledge of the types of urban morphology that include appropriate infrastructure to support biodiversity and ecosystem service production would therefore constitute a useful tool for urban planners to incorporate ecosystem management principles into decision making.

1.3.2. Social equity

The above definition notes that a sustainable urban morphology promotes social equity. Indeed, social equity is a key issue in the ethos of ecosystem services (Costanza 2000). It is widely recognised that the residents of poorer cities tend to suffer more from ecosystem service degradation (Millennium Ecosystem Assessment 2005c). This indicates that, at least at the inter-city scale, poorer individuals are deprived not only of social and economic opportunities, but also the welfare benefits provided by nature, i.e. ecosystem services.

However, there is only one previous study that touches on the relationship between socioeconomic status and ecosystem service production *within* cities (Tratalos et al. 2007). This study found that the more affluent areas in five UK cities tended to have greater carbon sequestration rates than less affluent areas from the same cities, but found no evidence that neighbourhood affluence was related to reduction of stormwater runoff or heat island mitigiation, and very little evidence for a relationship with the capacity to support biodiversity (Tratalos et al. 2007). However, the sample size of this study was small, and the inclusion of multiple cities means that idiosyncratic patterns may have been obscured. A tool that facilitated the study of relationships between socioeconomics and ecosystem services might be valuable in identifying and reducing inequities in ecosystem service provision between different strata of society.

1.3.3. Requirements for tools

In recent years there have been calls to "operationalise" the concept of ecosystem service, i.e. to turn it into useful tools to help managers, planners and decision makers to integrate human activity and environmental protection (Chan et al. 2006, Armsworth et al. 2007, Daily et al. 2009). Efforts to fulfil this need have identified several features that are required for such tools to be useful; these features are reviewed here.

1.3.3.1. Spatially explicit

Ecosystem service production, demand and supply, as well as urban planning, are all inherently spatial issues. Natural environments are spatially heterogeneous, with the production of a given ecosystem service occurring only where the appropriate environmental and ecological infrastructure is in place (Chan et al. 2006, Naidoo et al. 2008). However, the ecosystem service is only delivered (i.e. human welfare benefits only accrue) when human beings with a demand or need for that service are in the right place (Costanza 2008, Naidoo et al. 2008).

Land use, which reflects human activities and the built environment, is also obviously spatially heterogeneous. At least in countries like the UK, spatial patterns of land use tend to be formally planned. In the UK, planning occurs at two levels in order to consider social and environmental conditions at multiple scales: Regional Spatial Strategies set out the development needs across a large region; and local authorities plan a Local Development Framework to determine exactly what changes are to be implemented at a local scale (Office of the Deputy Prime Minister 2004). Local-scale planning of land use should both influence and be influenced by who lives where, and what those people require of their natural and built environment – in other words, spatial demographics (Communities and Local Government 2008b).

For these reasons, it is essential for ecosystem service tools to be spatially explicit in order to produce output that is useful for management and decision making (Chan et al. 2006, Nelson et al. 2009). The most obvious form of spatially explicit output is maps of ecosystem services, which are a powerful tool because they can simplify and summarise complex, sometimes intangible information, thereby facilitating new understanding (Krygier and Wood 2005). To date, however, very few studies have attempted to take ecosystem service maps further and use them as a basis for making planning recommendations or investigating social equity, for example (but see Chan et al. 2006, Nelson et al. 2009). Therefore the first requirement for ecosystem service tools developed in this thesis is that they are spatially explicit and produce output that is suitable for integration with other types of spatial data.

1.3.3.2. At relevant scales

Some studies that have mapped ecosystem services have done so at global scales (Costanza et al. 1997, Naidoo et al. 2008). Such large scale maps, while able to make strong statements about the state of the environment and human dependence upon it (Costanza et al. 1997), are not appropriate for informing actual planning and decision making, which tends to happen at a local scale (Naidoo et al. 2008). There is a need for ecosystem service mapping tools that produce output at scales relevant to management (Daily et al. 2009); for example, the ecosystem service conservation planning maps of the California Central Coast ecoregion produced by Chan et al. (2006). The InVEST (Integrated Valuation of Ecosystem Services and Tradeoffs) tool, which is currently in development, is designed to be implemented at small scales (Nelson et al. 2009). However, InVEST is not being designed with urban ecosystems and urban ecosystem services in mind.

1.3.3.3. Including multiple ecosystem services

The infrastructure required for ecosystem service production differs between ecosystem services, meaning that not all ecosystem services can be produced in one environment (Rodríguez et al. 2006). Consequently, spatial correlations between ecosystem services can be low and are sometimes negative (Chan et al. 2006, Naidoo et al. 2008). This means that one ecosystem service cannot necessarily be used as an indicator of other ecosystem services; therefore, until the relationships between ecosystem services are better understood, it is important to include as many ecosystem services as is feasible in assessments in order to reduce the chances of drawing incorrect conclusions due to excluded ecosystem services showing vastly different spatial patterns (Chan et al. 2006, Naidoo et al. 2008, Nelson et al. 2009).

1.3.3.4. Based on production functions

Costanza et al.'s seminal paper (1998), "The value of the world's ecosystem services and natural capital", mapped the economic value of ecosystem service production from broad land use type and an estimate of the value of each land cover for seventeen ecosystem services. Other studies applying land cover specific estimates of ecosystem service values have followed, although it could be argued that this approach is not truly spatially explicit because it assumes that each land cover type is homogeneous (Nelson et al. 2009).

An alternative approach to spatial assessment of ecosystem services, which takes into account at least some heterogeneity, is to develop ecosystem service "production functions" to model how ecosystem services vary with spatial ecological and social variables in addition to land cover (Nelson et al. 2009). Examples of mapping projects taking this approach include global mapping of the production of four ecosystem services using regionally explicit modelled data (Naidoo et al. 2008), and mapping of six ecosystem services for the central coast ecoregion of California using knowledge of the soil, land cover and land use, and other local data (Chan et al. 2006). However, increasing the number of variables included in the production function naturally makes model implementation more resource intensive, thereby increasing resource investments per ecosystem service (Nelson et al. 2009). Ecosystem service tools must therefore find an appropriate trade-off between scope and detail.

1.3.3.5. Using widely available data inputs

There is a need for ecosystem service tools that can be implemented in multiple locations (Daily et al. 2009). It follows that tools should therefore use widely available data as inputs; if not, the resources required to collect new data risks making implementation unfeasible The InVEST suite of models, for example, uses land use and land cover patterns to estimate the generation and economic value of ecosystem services produced in a region (Nelson et al. 2009).

1.4. Research aims

The key points from Section 1.2 and Section 1.3 can be summarised as follows. Urbanisation is associated with changes in patterns of land cover and land use, especially the loss of greenspace. These alterations cause profound changes to the biophysical structure, processes and functions operating in all parts of the biosphere, which in turn affect the generation and provision of all types of ecosystem services. Despite this, urban ecosystems can be capable of delivering a wide range of ecosystem services. Due to their impacts on human welfare, ecosystem services are a matter of social justice and equity, as well as of environmental protection. It is increasingly recognised that ecosystem services should play a role in the planning and management of urban areas in order to aid achievement of social and environmental goals. To date, however, there are no tools that are able to undertake this task.

The core aim of this thesis, which takes up this research opportunity, is to develop and demonstrate the utility of a toolbox for modelling urban ecosystem services. The toolbox aims to meet the following criteria:

- Based on "production functions" including as much ecological and social details as feasible;
- Including multiple ecosystem services;
- At a scale, or scales, relevant to decision making;
- Uses widely available data inputs so that it can be applied in multiple geographic areas;
- Providing output that could be useful to decision makers.

In order to develop tools fulfilling these requirements, ecosystem services suitable for urban spatial characterisation had to be selected and modelled. To achieve this, three methodological aims were identified. A further three analytical aims, which make use of the models to produce output of potential use to urban planners and decision makers, were also identified. These six aims are as follows:

1. To identify a suite of ecosystem services suitable for modelling at the neighbourhood scale within a city using existing data sets and methods;
2. To collate the input data required by the models from existing data sources;
3. To implement the ecosystem service models;
4. To investigate relationships between the modelled ecosystem services in the study area in order to contribute to understanding of how to undertake ecosystem service assessments;
5. To analyse relationships between urban morphology and ecosystem services in order to make urban planning recommendations;

6. To analyse social inequity in access to ecosystem services at the neighbourhood scale, in order to identify any sectors of society with limited provision.

1.5. Case study area: Sheffield

The case study area for this thesis is the metropolitan borough of Sheffield, South Yorkshire, UK. A large-scale Ordnance Survey map of Sheffield is shown in Figure 1.2. This section provides an introduction to the city of Sheffield, including its geographical and topographical setting, a brief history of its origins and development, the present state of the city and its relation to the surrounding areas.

1.5.1. Regional geography

Sheffield is an inland city lying slightly north of the centre of England, and covering an area of 368 km^2. Topographically, Sheffield is hilly and lies over a wide altitudinal

Figure 1.2. Ordnance Survey map of Sheffield; bold black line shows boundary of Sheffield metropolitan borough. Map shows area approximately 30km across. Source: EDINA Digimap (http://edina.ac.uk/digimap/index.shtml), Ordnance Survey maps ‘Roam’ application, accessed 12/04/2010.

range, from 592m above sea level in the west to 19m a.s.l. in the east (Ordnance Survey, no date); a map of Sheffield's topography is shown in Figure 1.3. There is also a strong west-east divide in soil types and geology. The east of the city sits over coal measures, and the soils are loam and clay (Fine 2003, Cranfield University 2009). To the west are areas of millstone grid, and the soils are peaty, with blanket peats at the higher altitudes (Fine 2003, Cranfield University 2009).

As a consequence of these characteristics of the physical environment, there is a strong longitudinal pattern in land cover and land use in Sheffield. Ninety five percent of the population live in the urbanised eastern part of the city (Beer 2003); travelling west there is first a region of arable and pasture land interspersed with areas of woodland, and then moorland and upland bogs (Figure 1.2).

Sheffield is adjoined to Rotherham in the east (Figure 1.4). Barnsley is located to the north and Doncaster to the north-east of Sheffield, while Chesterfield lies to the south (Figure 1.4). The large city of Manchester lies to the west on the other side of the Peak District. Hydrologically, most of Sheffield lies within the Don catchment, which ultimately drains to the Humber estuary to the northeast. The rivers Sheaf, Rivelin,

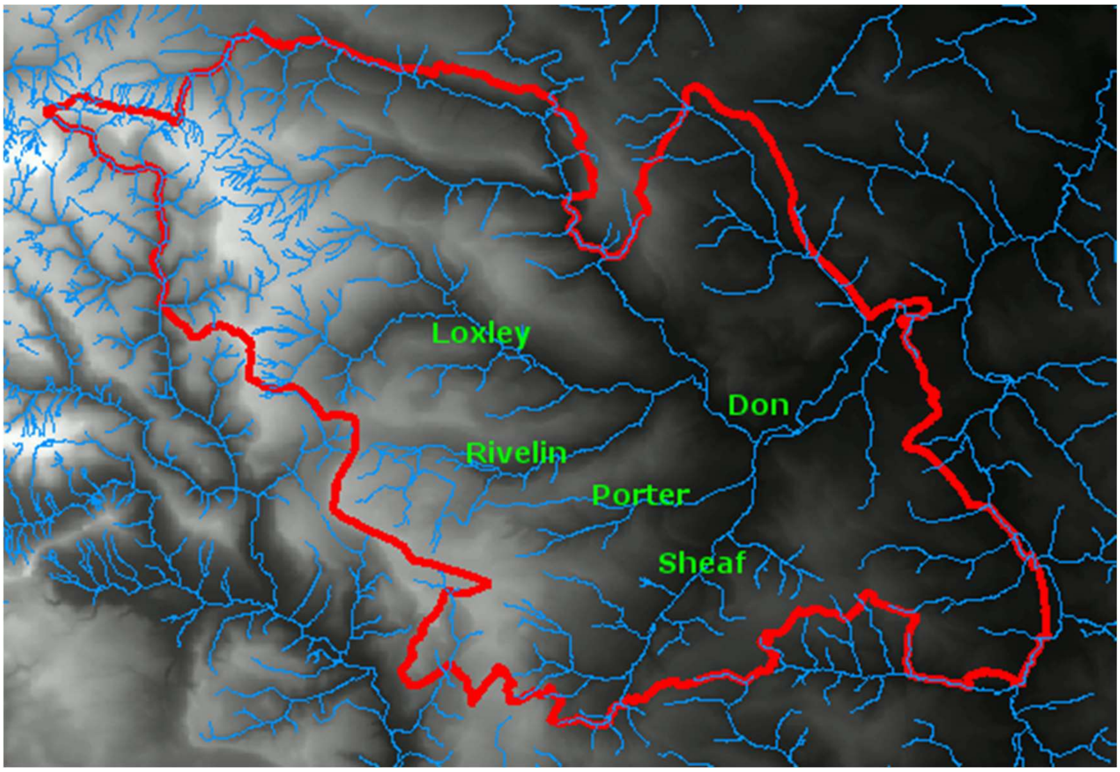


Figure 1.3. Topography and rivers of Sheffield (boundary in red; main river names in green) and the surrounding area. Lighter colours indicate higher altitude and vice versa (altitudinal range within Sheffield boundaries 592m-19m above sea level). Sources: Office for National Statistics (2004), Ordnance Survey (no date).

Figure 1.4. Regional Ordnance Survey map of Sheffield and the surrounding towns. Dark yellow shows urbanised areas; western area with a darker brown background indicates the Peak District National Park. Source: EDINA Digimap (http://edina.ac.uk/digimap/index.shtml), Ordnance Survey maps 'Roam' application, accessed 13/04/2010.

Loxley and Porter all meet their confluence with the river Don within the boundaries of Sheffield (Figure 1.3).

1.5.2. History

1.5.2.1. Early history

The following account of Sheffield's history is summarised from David Hey's (1998) 'A History of Sheffield', except where otherwise noted. The earliest archaeological evidence from the area that is now Sheffield dates from the Bronze Age (2100-750 BC). By 500 BC the Sheffield area was thought sufficiently worthy of defence that a fort had been built on a hilltop with a commanding view of the River Don. It is however unknown whether the area was occupied continuously until after the Roman occupation in AD 69.

The first written evidence of Sheffield appears in the Domesday book (1086), which refers to small settlements at the edge of the moors. The name 'Sheffield' refers to open country (i.e. not forested) by the River Sheaf. Sheffield would later become famous for its markets; the first evidence of a market held at Sheffield dates from 1281, and a royal charter for a weekly market was obtained in 1296. At this point, the parish of Sheffield comprised six townships, each with small settlements that later developed into villages (see map in Figure 1.5). The Sheffield township was located where the city centre is today and stretched to the southeast. Ecclesall township was to the southwest of Sheffield township, and Upper and Nether Hallam to the northwest. Brightside was to the north and Attercliffe-cum-Darnall to the east. These place names still exist today.

Sheffield would also later become known as a centre for industry. The first known reference to a Sheffield cutler was made in 1297, and there is also documentary evidence of the use of water to power a mill dating from the thirteenth century. The natural resources in and around Sheffield – including several rivers suitable for powering mills, forests, millstone suitable for grinding, coal measures and ironstone – were largely responsible for this early success of industry in Sheffield.

The medieval city developed a distinctive character as a market town where cutlery (i.e. implements with a cutting edge) was made. In the late sixteenth century, industry in Sheffield benefited from the development of the mineral wealth in the estate of George Talbot, the earl who controlled Sheffield. Blast furnaces and forges were built at Brightside and Attercliffe-cum-Darnall, the coal seam in the park in Sheffield township was exploited, and springwoods were managed to produce wood for charcoal.

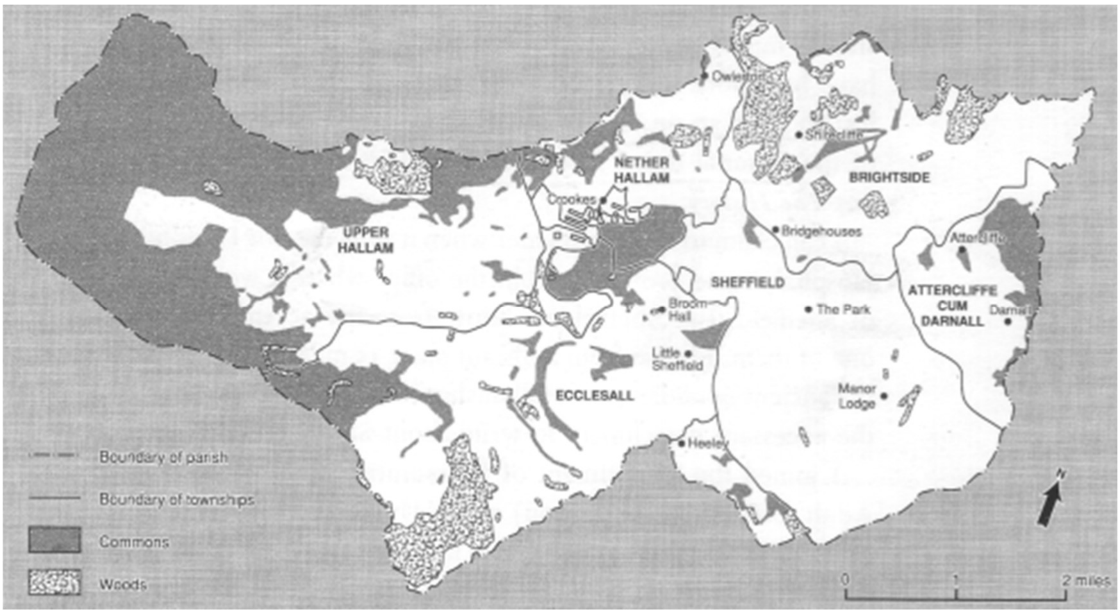


Figure 1.5. The ancient parish of Sheffield, divided into six townships. Source: Hey (1998).

A variety of mills had been built by this time, including some used for cutlery grinding. Spanish iron was being imported for cutlery production, as it produced a better cutting edge than local resources, and Sheffield had become the second biggest producer of cutlery in the UK (following London).

1.5.2.2. Industrial Sheffield

By 1590, Sheffield cutlery had a national reputation for its quality. By 1616, approximately 60% of Sheffield's workforce were engaged in the cutlery trade; and in 1624 the Company of Cutlers was incorporated by an Act of Parliament and empowered to enrol apprentices, to admit freemen to the trade and to act on behalf of its members. As a consequence of its industries, however, Sheffield was also gaining a reputation for being a dirty, smoky place.

In 1616 the population of Sheffield was 2,207. Sheffield expanded rapidly throughout the seventeeth and eighteenth centuries; by 1806 the population of the Sheffield township was 31,315, with a further 14,441 living in the rest of the parish. Many new houses were built, but the response to both population and economic growth was slow. The market, which had grown considerably and serviced parts of Derbyshire, Nottinghamshire and the West Riding of Yorkshire, did not receive a much-needed reorganisation until the 1780s and 1790s. The first cultural events and facilities, including a theatre and library, were not developed until the second half of the eighteenth century.

The new developments were modest, however, as the parish had no rich merchants or wealthy corporations to build fine houses, and did not attract the gentry. Nevertheless, it seems that despite the fact that most Sheffielders were not rich, there were few poor quarters at this time. In general, there was a reduction in levels of poverty during the eighteenth century. Public health was also surprisingly good considering the levels of pollution, thanks to winds from the moors providing clean air, good drainage to the rivers, and the fact that few people resided in cellars (unlike in other cities at the time).

It was also during the seventeenth and eighteenth centuries that Sheffield became a reknowned and innovative centre of steelmaking, spurred on by the established cutlery and tool making industries. By 1740 Sheffield had enough cementation furnaces (for cementation steel) to supply the needs of local cutlers and tool makers. In 1742 Benjamin Huntsman, who had recently moved to Sheffield to be at the centre of the steel industry, invented cast steel; cast steel is better than cementation steel for items such as rolls, dies and fine work. The following year, Thomas Boulsover, a local cutler, discovered Old Sheffield Plate. Old Sheffield Plate is a method for plating copper with silver, and was ideal for producing cutlery more cheaply than with solid silver. By this time, the Cutlers' Company had also expanded and was challenging London as the chief manufacturer of cutlery in the UK.

As a landlocked city in a hilly region, however, transporting Sheffield wares to more distant outlets was challenging. The idea of making the River Don navigable was first proposed in 1698, but the Cutlers' Company was not receptive to the idea at first and it took until 1751 for the Don to be made navigable to Tinsley, located at the eastern edge of Sheffield. A road completed the transport link to Sheffield's centre. In the following decades, many of the roads around Sheffield were also improved, meaning that the parish suddenly had far better connections with other towns. It was at this time that the town's fortunes began to change.

In the first half of the nineteenth century, the population again tripled to 135,310 by 1856. In 1893 Sheffield acquired city status, and had incorporated many new areas beyond the medieval boundaries. By 1901 the population had again risen dramatically to over 400,000. During the Victorian era (1837-1901), Sheffield acquired its modern character, with its central streets becoming a commercial centre, steelworks in the east end, many red brick terraces in working class suburbs, and middle class suburbs located in the west. Although Sheffield never became a commercial centre on the scale of Leeds, it was home to large numbers of shops, cafes, hotels and so on.

It was also during this expansion that socioeconomic inequalities began to appear: areas in east and north Sheffield were developed as working class districts while areas west of the town centre became fine Victorian suburbs. Notably, it was the west of the town that remained cleaner, thanks to the prevailing winds from the Peak District; these regions also avoided the flooding that regularly occurred south of the River Don. By 1901 the difference in character between the middle class west end and the working class east end was pronounced. Living conditions in the town centre were poor, and Sheffield continued to make a negative impression on visitors as a dirty, smelly and smoky town – albeit one surrounded by exceptional countryside. The large numbers of back to back terraces were a public health risk, sewerage was poor, and both the air and water were dirty; consequently, in the 1850s Sheffield had some of the highest death rates from infectious diseases in the whole country.

During the late nineteenth century a number of much-needed improvements were made to the city's infrastructure and services. A main sewerage and drainage system was constructed, and a street widening scheme was begun in the city centre. New dams and reservoirs were built to improve the provision of drinking water. Horse buses, then horse trams and later electric trams, were instituted to enable people to live further away from their place of work. A number of hospitals were built, including several specialist hospitals to provide services such as mental health, women's and children's care. Sheffield became the first Yorkshire town to establish a public library in 1856. Private donors provided higher education facilities in the forms of the Medical School, Firth College and Sheffield Technical School, which merged in 1897 to form the University College of Sheffield; university status was granted in 1905. The Public Health Act of 1872 reduced the prevalence of neglected areas, and a number of large public open spaces were instituted to provide a respite from Sheffield's smoky and dirty environment. Some of the slums were cleared and rebuilt. Nevertheless, by World War I thousands of people still lived in poor conditions.

The construction of the railway in the mid nineteenth century provided a stimulus to industrialisation of the east end of Sheffield. By this time Sheffield had long since outgrown London as a producer of cutlery. By 1814 the Cutler's Company was unable to maintain authority over the vast number of local cutlers and entry to the trade was made entirely free. Sheffield exported wares to London, America, Europe and the Commonwealth. Nevertheless, specialisation, outsourcing to other local firms and handicraft skills remained important to the structure of the cutlery industry.

Local cutlers remained confident that the quality of their wares would withstand competition from cheap cutlery mass-produced elsewhere. They also believed that introducing machinery would reduce their products' quality. Consequently they failed to modernise, and by the 1870s this had caused the loss of the American and European markets. During the World War I the industry received a respite due to demand for items such as army knives, bayonets and surgical instruments, but the trade remained dominated by small firms using traditional methods. After the war, however, only the firms that were prepared to modernise could prosper.

In the second half of the nineteenth century, steel overtook cutlery as Sheffield's biggest industry. Steel was much in demand for the production of railways, and later also for armaments in the arms race leading up to World War I. In 1858 Henry Bessemer, who had invented a process for producing mass amounts of relatively low quality steel – suitable for railways – opened a works in Sheffield. Sheffield firms were also early adopters of the Siemens furnace, which produced steel of a higher quality suited to items such as guns and ship drive shafting.

The strong competition in the Sheffield steelmaking industry resulted in experimentation and innovations, with many new alloys and improvements to processes being discovered. Manganese steel, which hardens with use and is ideal for armaments, and tungsten tool steel, which is very hard and thus suited to cutlery, were both invented in Sheffield. Stainless steel, which was invented in Sheffield by Harry Brearley in 1912, was a key development. Thus, even when the railway export trade was lost by 1980 (due to new processes that had not been implemented in Sheffield) the steel industry remained successful.

As well as playing a leading role in the cutlery and steel trades, Sheffield was noted for its production of tools: the huge amounts of crucible steel that could be produced in Sheffield ensured world dominance in the tool industry. Coal could also be profitably sold on the north east coal market, and the collieries were important employers in the central and eastern areas of the city. Brewing was another prosperous industry: in 1831, Sheffield was home to more than 1500 public houses.

1.5.2.3. World War I onwards

Sheffield suffered a long-lasting recession following World War I. During the war, America and Europe had improved their special steel infrastructure and consequently much of Sheffield's market was lost. Once armaments were no longer in such demand, levels of unemployment in Sheffield rose and stayed high until the rearmament prior to World War II. The firms that remained successful were largely amalgamations of large firms, with a few small firms still making high grade alloys. The cutlery industry was also hit hard by a post-war recession, despite an initial boom in the market for luxury goods. Many Sheffield firms still refused to modernise and could not compete with more efficient foreign firms, and consequently many went backrupt in the 1920s and 1930s.

Public health was improved during the period between the wars as more slums were cleared and rebuilt to be less overcrowded and of lower density. However, this focus on housing in combination with the recession meant that other services, especially education, were economised; and despite the efforts to improve housing quality in the old slums, the socioeconomic polarisation between the west and east ends became even more pronounced.

Sheffield's industries became more prosperous in the lead up to, and during, World War II. Despite being a centre of industry, Sheffield was not heavily damaged during the war. However, transport, water, gas and electricity were disrupted during a series of attacks in 1940, and several steelworks were damaged.

In the decades following the war Sheffield continued its redevelopment and improvement plans. The 1956 Clear Air Act resulted in a large reduction in air pollution. The council's Smoke Control Order required smokeless domestic fuels to be used in the windward side of the city from 1959. A power station and a coking plant were also closed. Clearing and rebuilding of the slums continued; although the huge demand for new houses meant that many were built at high density and to reduced building standards. High rise developments were built near the city centre during the 1950s and 1960s, but they were soon deemed to be a technical and social failure and so further redevelopments were in the form of low level compact estates.

The structure of Sheffield's industries was again forced to change after the war. In the 1950s, although many cutlery firms remained, most were small and had poor management; the more successful firms had German origins, were larger and used mass production methods. It was only these larger firms that could survive the flooding of the market by Asian imports during the 1960s. Very few small firms remain today.

The steel industry remained competitive following the war thanks to its knowledge and skills base, and the research centres that developed new technologies. When the world demand for steel collapsed during the 1970s global recession, however, Sheffield's steel manufacturing base began to shrink drastically due to competition from foreign firms and a reduced national engineering market. Many firms closed during the 1980s, with unemployment in Sheffield hitting a high of 16%.

Sheffield remained in a deep recession for several years, due partly to tension between the central Conservative government and local left-wing Labour government. The turning point came in 1986 when the Sheffield Economic Regeneration Committee, a collaboration between the public and private sectors, was set up to promote Sheffield and establish new jobs, especially in the Lower Don Valley. New housing and industrial parks were developed in the east end. A science park was built in the city centre for hi-tech industry, and a Cultural Industries Quarter was established as a centre for art and music. Leisure and sport were identified as future providers of jobs, and

Sheffield's hosting of the 1991 World Student Games prompted the construction of many world-class sports facilities. The Meadowhall shopping centre was also opened in 1990, and the Don Valley link road was built to link the M1 and the city centre to purpose-built business and retail parks.

During the 1990s, further economic regeneration resulted from cooperation between the City Council, Sheffield Development Corporation, the Chamber of Commerce, the universities and other institutions. Public investment was targeted in the city centre. The rapid growth of the universities since the 1960s has also impacted on the landscape, economy and prestige of the city. Although the universities (and their students) are major contributors to the local economy, and they provide expertise to local industries and public services, the social consequences in the residential areas popular with students have been seen as undesirable by many people.

The increase in Sheffield's ethnic diversity since the World War II has also resulted in social tensions. Groups to have immigrated to Sheffield in large numbers include Polish armed forces who did not want to return to Russian-occupied Poland after the war, people from the Indian subcontinent, and Somali political refugees. Many of the immigrants moved into the areas around the old steelworks in the northeast of the city, forming distinct communities. There have been hostilities both with the native population and between groups of immigrants.

1.5.2.4. Sheffield today

Sheffield has lost much of its industrial character in only a few decades, and in 2006 the manufacturing sector employed only 14% of the workforce (Hey 1998, Sheffield City Council 2007). Nevertheless, the economy has recovered from the recession of the 1980s, with the unemployment rate in 2008 being only very slightly (around 0.3%) higher than the national rate (Sheffield City Council 2009b). In 2003 more than three quarters of the workforce were employed in the service sector: 32% were in public administration, education and health services, with a further 44.3% in other services (Sheffield City Council 2003). The remaining manufacturers are mostly in hi-tech industries (Sheffield City Council 2009b).

Administratively, Sheffield is split into the 28 wards shown in Map 1 (all maps are shown in Appendix D). Ward boundaries are drawn to include a similar number of people within each, so Map 1 also gives an approximation of population densities. In 2005 the total population of Sheffield numbered 520,700 – almost exactly the same as twenty years previously (Lovatt 2007). During this time the proportion of the population made up by people from black and minority ethnicities has almost doubled, and was over 13% in 2005 (Lovatt 2007). However, there is much spatial variation in the ethnic composition of communities: more than 70% of the population of some areas to the north and east of the urbanised area identify as an ethnic group other than white British, while the population of much of the north and south of Sheffield is more than 90% white British (Lovatt 2007). The city centre has quite high proportions both of people from ethnic minorities and of students (Lovatt 2007). In 2005, almost 8% of Sheffield's population was made up of students at the two universities: these students are concentrated mainly in central Sheffield (wards such as Crookes, Walkley, Broomhill, Central and Nether Edge; see Map 1).

Despite being one of the largest cities in the UK, many of Sheffield's communities retain distinctive characters (Hey 1998). This is in large part due to its topography, meaning that communities are physically separated, and making the description of Sheffield as "the biggest village in the world" – first recorded in 1887 – still relevant today (Hey 1998). Nevertheless, Sheffield has become a regional entertainment centre, attracting many visitors to the theatres, museums, musical events at the City Hall, cinemas, nightclubs, and sports events (Hey 1998).

Sheffield is also known for being a "green city". A recent survey found that 55% of the city is comprised of greenspace uses such as parks, woodlands, farms, private gardens and "incidental" greenspace such as verges (Beer 2003). One third of the area within Sheffield's boundaries lies in the Peak District National Park, and more is under agricultural land uses (Beer 2003). There are 150 woodlands and 50 public parks (Beer 2003). This greenspace network is recognised as a culturally and environmentally valuable resource by the City Council (Sheffield City Council 2009a).

Despite more than a century of attempts to reduce levels of deprivation, Sheffield remains a socioeconomically divided city (Hey 1998, Sheffield City Council 2009a, Sheffield City Council 2009b). Figure 1.6 shows how patterns of affluence, education and health spatially coincide. Residents of the north and east parts of the urbanised area are more likely to be receiving income support, job seekers' allowance or pension credits than residents elsewhere (Figure 1.6a), and are less than half as likely to obtain a good set of GCSE results as residents of the Dore, Totley and Fulwood areas (Figure 1.6b). The life expectancy of residents of many of these areas is ten years less than that of areas with the highest life expectancies (Figure 1.6c), and there more than twice as many emergency hospital admissions per person (Figure 1.6d).

Reduction of deprivation levels is a core part of Sheffield's continuing regeneration programme (Sheffield City Council 2009a). The north-east and south-east have been targeted for housing market renewal and service improvements; economic regeneration of the old manufacturing centre of the city is ongoing. There is also an emphasis on on renewing existing areas of low-demand housing instead of expanding the developed area, and on protecting, improving and interconnecting the city's greenspaces.

1.5.3. Advantages of Sheffield as a case study

The administrative boundary of the metropolitan borough of Sheffield is used at the study area for this project. Using an administrative boundary as the extent of the study area makes any output from the project more relevant to decision makers than would be

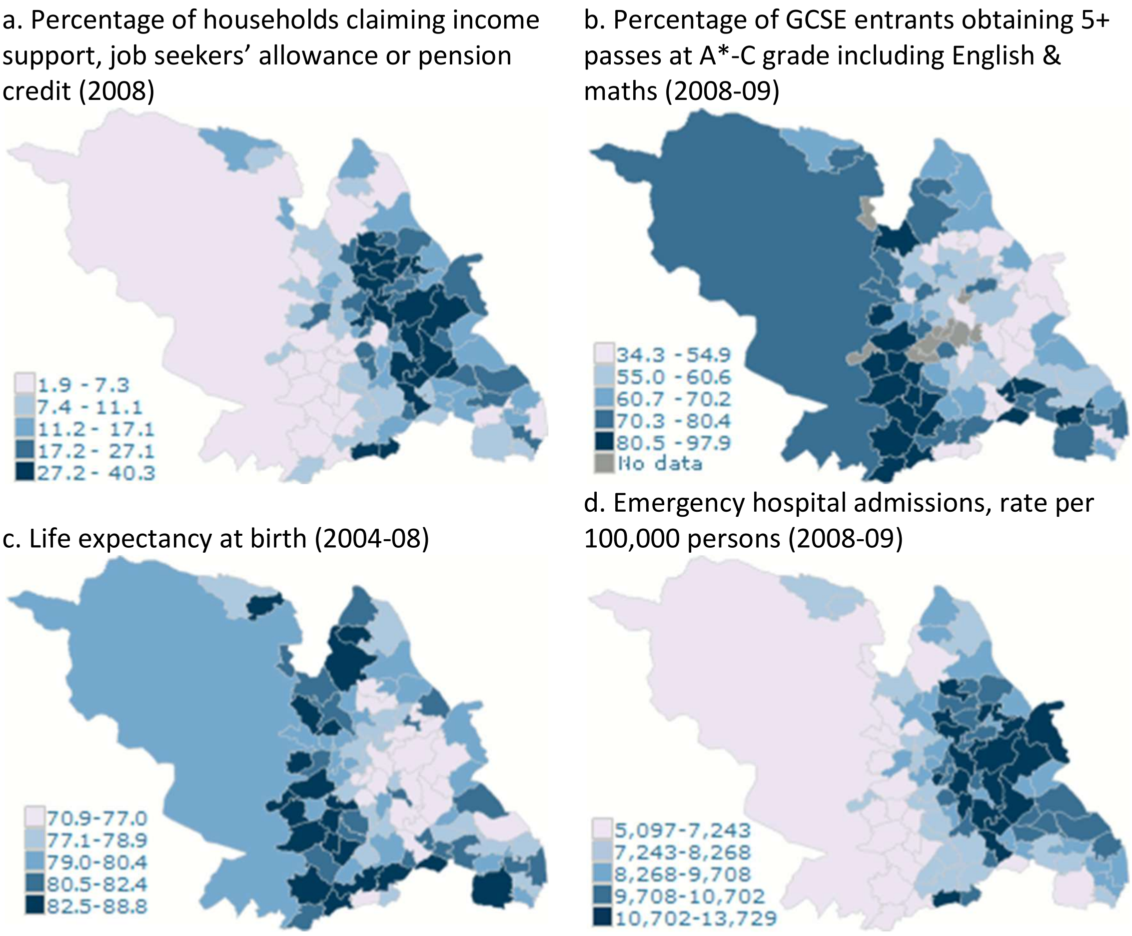


Figure 1.6. Socioeconomic inequalities in Sheffield. Source: Neighbourhood Health & Well Being Atlas (http://www.sheffield.nhs.uk/healthdata/atlas/NHoodSingle0410/atlas.html, accessed 11/05/2010). Dotted Eyes © Crown Copyright 2010, Licence No. 10019918.

possible with an area defined by some environmental or ecological criterion. The reasons that Sheffield is advantageous as case study area for this project are several:

- Sheffield has an established and valued urban greenspace infrastructure, indicating that ecosystem services are implicitly valued even if they are not directly managed.
- Sheffield contains within its boundaries a number of broad ecological and environmental gradients, so that consequently there is considerable variation in land use and land cover and thus in the quality and quantity of ecosystem services generated.
- The piecemeal development and redevelopment of the city have resulted in a wide variety of urban morphologies, permitting analysis of the relationship between urban morphology and ecosystem services.

1.6. Research and thesis structure

This chapter has introduced the subject of this thesis. Chapter 2 identifies the ecosystem services that will be the focus of the toolbox created by this thesis; then provides details of the spatial datasets used as input to the six ecosystem service models, including the methods and data used to generate a suitable land cover map, and a brief investigation of the spatial coincidence of the different datasets. Chapter 2 also describes the different spatial units used as a basis for the analyses undertaken. Chapter 2 meets the first research aim listed in Section 1.4, and describes some of the data needed to satisfy the second aim.

Chapter 3 to Chapter 8 each contains details of one ecosystem service model. These chapters start with an introduction to the problems that the ecosystem services counteract and a conceptual discussion of how the ecosystem service is generated. An overview of how each model works is then given, along with descriptions and evaluations of the model limitations. Further technical details of the models are provided in Appendix A, along with additional information about model parameterisation. For the more complex models, where the effects of altering individual parameters are not obvious, Appendix A also details sensitivity tests that are summarised in the individual chapters. Finally, these chapters present and discuss the output from each model singly. These chapters complete the second and third research aims.

The next three chapters address the three analytical research aims. Chapter 9, which meets research aim four, discusses the output of the ecosystem service models from a multiple service perspective. Chapter 9 explores ecosystem service correlations and hotspots, and discusses the influence of the choice of spatial units of analysis on the conclusions drawn.

Chapter 10 investigates how urban morphology is associated with ecosystem service provision, both across all land uses and within two more common specific land uses, and makes planning recommendations based on these analyses. Chapter 11 then looks at the provision of ecosystem services to different socioeconomic groups, to identify which parts of society are affluent and deprived in terms of ecosystem services. These chapters address the fifth and sixth research aims.

Finally, Chapter 12 draws together the research presented in this thesis in a general discussion and evaluation.

2. Preliminaries to ecosystem service modelling

2.1. Identifying ecosystem services for modelling

The core aim of this thesis is to develop and demonstrate the use of models of ecosystem services in an urban area. The first step towards achieving this, as encapsulated in the first research aim in Section 1.4, is to identify a suite of ecosystem services for inclusion in this project. Ecosystem services were chosen for inclusion on the basis of the existence of suitable models, or models that could be easily adapted; and on the basis of using data inputs that were likely to be readily available for the study area. The likelihood of potential ecosystem services being of relatively high priority to decision makers in an urban context was also considered in selection. The following six ecosystem services were identified:

- Reduction of air pollution by vegetation;
- Mitigation of the heat island effect by vegetation;
- Reduction of storm water runoff through retention in soils and by vegetation;
- Carbon storage in soils and vegetation;
- Opportunities for cultural ecosystem services (e.g. recreation and relaxation) in greenspace;
- Provision of habitat for flora and fauna.

These ecosystem services vary in the factors controlling their production and delivery, their spatial and temporal characteristics, and the type of benefits provided to humans. Table 2.1 summarises some of these issues, which are discussed further in the sections below. The services also vary in the type of benefits provided to humans Reduction of air pollution, mitigation of the heat island effect, reduction of storm water runoff and carbon storage and sequestration can all be considered as regulating services, and as such are valuable to humans because they contribute to a benign physical and

Table 2.1. Summary of important factors influencing the demand for, and production of, selected ecosystem services; and spatiotemporal characteristics of the provision of services to beneficiaries. These lists are not intended to be exhaustive.

Ecosystem service	**Factors influencing demand**	**Factors controlling production**	**Nature of supply**
Reduction of air pollution	Pollutant concentration, population density	Capacity of land cover surfaces to intercept pollution (vegetation and ground; e.g. transpiration rates, surface microtopography), pollutant concentration	Local proximal reduction of air pollution concentration, i.e. benefit to individuals near to the site of service production, when pollution levels are dangerous to health
Mitigation of the heat island effect	Degrees > comfort level, population density	Evaporation and evapotranspiration rates, surface specific heat capacity, building mass, local climate and solar factors	Local proximal reduction of surface temperature, i.e. benefit to individuals near to the site of service production, when temperatures are uncomfortably high
Reduction of storm water runoff	Flood risk, population density	Capacity of land cover and soil to intercept precipitation (e.g. infiltration rates), recent weather patterns, storm event characteristics	Directional flow related reduction of flood risk, i.e. reduced flood risk to individuals down-catchment from runoff reduction, after specific storm events
Carbon storage	N/A – non-spatial	Amount of biomass and carbon in soil	Global non-proximal contribution to climate regulation, i.e. all humans benefit regardless of location, with long-term benefits
Opportunities for cultural ecosystem services	Population density	Park accessibility and condition	User movement related access to physical and psychological health benefits, i.e. people move to parks, with service provided during visits to parks
Provision of habitat for flora and fauna	N/A – non-spatial	Land cover matrix	Contribution to spatiotemporally variable production of direct ecosystem services

chemical environment (Millennium Ecosystem Assessment 2005b, Wallace 2007). The value of cultural ecosystem services arises through their contribution to personal fulfilment (Millennium Ecosystem Assessment 2005b, Wallace 2007). Habitat for biodiversity is beneficial to humans directly through existence values, and also indirectly by contributing to the environmental processes and functions that produce ecosystem services (Figure 1.1), as discussed in Section 1.2.6 (De Groot et al. 2002, Chee 2004, Millennium Ecosystem Assessment 2005b). These ecosystem services are discussed further in the following six chapters describing the models.

2.1.1. Defining the ecosystem service

In the context of the models produced in this thesis, it is necessary to define what is meant by the *ecosystem* service. This is because, in several cases (air pollution reduction, storm water runoff reduction, carbon storage), artificial surfaces are able to contribute to the service: as shall be seen in the following chapters, artificial surfaces provide a deposition surface for pollutants and abstract small amounts of storm water, and soils under artificial surfaces still contain some carbon. The purpose of the modelling in this thesis is to identify the additional service generated by the urban greenspace infrastructure. Thus, for example, it is the ability of greenspace surfaces to remove *more* pollutants than artificial surfaces that it is interest.

The land cover map introduced in Section 2.2.1 shows that, of the total artificial surfaces in Sheffield, 28% are buildings and 72% are ground-level surfaces. These two land cover types have different properties with regard to some ecosystem services. Thus the ecosystem services modelled in this thesis are quantified and mapped as the difference between the amount of, say, air pollution reduction that occurs given the actual land cover composition, and the amount of air pollution reduction that would occur if the same area were composed of 28% buildings and 72% other manmade surfaces (as though the greenspace currently present was replaced by buildings and manmade surfaces in the same proportions as present in the existing artificial surface areas of the city).

The ecosystem services of air pollution reduction, heat island mitigation, storm water runoff reduction and carbon storage are defined in this way, i.e. as the difference between the modelled rate/quantity given the actual land cover composition and the hypothetical rate/quantity for 28% building and 72% manmade surface cover. This definition is not however relevant for opportunities for cultural services, which, as seen in Chapter 7, is based on modelling the proportion of the area that is suitable for providing such services: if there were no greenspace infrastructure, this area would be zero. The index of habitat provision also does not involve comparison to a hypothetical land cover; that is, an area with no greenspace would not be considered to be providing any habitat for biodiversity (Chapter 8).

2.2. Collation and creation of data inputs

In order to model the six ecosystem services identified in Section 2.1, a variety of input variables are required. This section describes the spatial data sets used as input to the models, in partial satisfaction of the second research aim listed in Section 1.4 (the remaining input data are data parameters, which are described in the chapters relevant to each ecosystem service).

Modelling these six ecosystem services requires four spatial data inputs. A land cover map is essential for all of the ecosystem service models except the provision of habitat for biodiversity, which instead requires land use data. Three of the models – mitigation of the heat island effect, reduction of stormwater runoff and carbon storage – require soil-specific parameters, meaning that a soils map is needed; and, although not strictly necessary for the way it is modelled in this thesis, modelling the reduction of air pollution also benefits from knowledge of the spatial distribution of air pollution in terms of putting the results into context.

Three of these spatial data sets were already available for use, but a land cover map was made to suit the requirements of this project. This section describes and documents the spatial datasets, including the methods used to generate the land cover map. The section also analyses how particular land covers, land uses and soil types spatially coincide, in order to facilitate understanding the implications of the analysis undertaken in later chapters.

2.2.1. Land cover map

Land cover is defined as the directly observable "layer of soils and biomass, including natural vegetation, crops and human structures that cover the land surface" (Verburg et al. 2009), i.e. the directly observable physical characteristics of the land (Fisher et al. 2005, Haines-Young 2009). Land cover is in contrast to land use, which "refers to the purposes for which humans exploit the land cover" (Verburg et al. 2009), or the "economic and social functions of that land" (Haines-Young 2009). Thus while land cover describes the vegetation type and structure, one type of land cover can be used in different ways: for example, grassland can be used for grazing, golfing or biofuel production (Verburg et al. 2009). Similarly, a single land use can include multiple types of land cover, such as a golf course including grass, scrub, trees, water and so on (Haines-Young 2009).

In terms of biodiversity, ecological processes and ecosystem services, the spatial interaction between land cover and land use is often critical (Verburg et al. 2009). For example, grassland that is used for hay will produce a different set of ecosystem services to unmanaged grassland, with only the former producing fodder, and also potentially excluding some more sensitive species if disturbances associated with agricultural activities make the habitat unsuitable. However, most land classification schemes include elements of both land cover and land use, which can make it difficult to interpret ecological and environmental situations (Fisher et al. 2005, Haines-Young 2009). One reason for this is the different approaches required to identify land cover and land use: land cover maps tend to be generated using satellite imagery, using either automated or manual interpretation, with the few land uses that can be distinguished being included as separate categories; whereas most land uses require more contextual information for identification (Fisher et al. 2005)

Nevertheless, land use and land cover typologies must be suitable for purpose (Fisher et al. 2005). Several of the models in this thesis are parameterised for conflated land cover/land use typologies; consequently, a land cover map that includes some land use categories is not unsuitable in the present case, especially if it contains information not included in the separate land use map.

2.2.1.1. Data sources

In the UK, the Centre for Ecology and Hydrology's Britain-wide Land Cover Map is a widely used spatial land cover dataset that also includes a few land use categories. The second version of this map, the Land Cover Map 2000 (LCM2000), was one of two data sources used in the construction of this land cover map.

The LCM2000 is a satellite imagery-derived classification of land cover into the Joint Nature Conservation Committee's Broad Habitat Classification (Jackson 2000, Fuller et al. 2002). The level two vector version of LCM2000 was used in this study, which includes the following 26 classes: broad-leaved woodland; coniferous woodland; arable cereals; arable horticulture; non-rotational horticulture; improved grassland; setaside grass; neutral grass; calcareous grass; acid grass; bracken; dwarf shrub heath; open dwarf shrub heath; fen, marsh, swamp; bog; inland water; montane habitats; inland bare ground; suburban/rural developed; continuous urban; supra-littoral rock; supra-littoral sediment; littoral rock; littoral sediment; saltmarsh; and sea/estuary. The accuracy with which these categories are distinguished varies; more information is given by Fuller et al. (2002).

LCM2000 has a minimum mappable unit of 0.5ha and the GIS layer has a 25m cell size (Fuller et al. 2002). LCM2000 was derived from satellite imagery from both the summer and winter of 1998, or the nearest available year.

The resolution of the LCM2000 is low compared to the scale of land cover heterogeneity in densely urbanised areas. Ordnance Survey style mapping is of a more suitable resolution. An Ordnance Survey product entitled MasterMap topography layer was used in this project to identify units of relatively homogeneous land cover as a basis for land cover mapping.

The MasterMap topography layer comprises sets of points, lines and polygons showing landscape features (for example, buildings, the natural environment, transport infrastructure and water bodies); administrative boundaries; heritage features; terrain and height; and annotation of these features including road names and selected house numbers. More information on this layer can be found in Ordnance Survey (2008). Only the topography area polygon features were used in generating the land cover map.

Each MasterMap polygon is described by a number of data, of which the relevant ones are as follows:

- Make – whether the object represented by the polygon is manmade or natural. Gardens are classified as being of 'multiple' makes, and some polygons are unknown or unclassified.
- Theme – classified as one or more of: buildings, water, land, rail, roads, tracks and paths, structures.
- Descriptive group – the primary classification of a feature, into at least one of 21 categories usually describing physical topographic features; for example building, built environment, general surface, inland water, natural environment.
- Descriptive term – further classification information, especially about natural environment polygons. Examples values are: coniferous trees, scattered coniferous trees, rough grassland, rock, heath. May have zero, one or multiple values.

The topography layer does not have an absolute spatial resolution below which features are not included. Instead, features considered by OS to be important for their mapping purposes are included. In practice this means that in urban areas small features such as garden greenhouses are often included, and in rural areas small notable rock formations may be shown. The layer was first released in 2001, and includes updates (established by ground and aerial survey) up to the date of data receipt at the end of December 2008 (Ordnance Survey 2008). Where land cover has changed more recently than 1998, there are therefore discrepancies between LCM2000 and the MasterMap topography layer.

2.2.1.2. Typology development and classification procedure

The topography area polygons are used as the land cover map polygons, i.e. each polygon is assigned to a single land cover type. The polygon information is also used in the actual classification algorithm as described below.

The MasterMap topography layer is used preferentially for polygon classification; however, there are some polygons for which it cannot resolve a land cover type. For example, some polygons are classified as unknown, and a specific issue is that farmland is classified as “general surface”. In these situations the LCM2000 is used to attempt to classify the polygon.

The typology developed for this study was designed to include as many categories that could be distinguished with reasonable accuracy as possible. Table 2.2 shows the draft typology and the classification rules used to determine class membership. Many categories are distinguished within the MasterMap ‘descriptive group’ (see Section 2.2.1.1) of “natural environment”, and many of these polygons have multiple ‘descriptive term’ values. To simplify the land cover data structure by avoiding one-to-many relationships, and to facilitate integration with LCM2000 data (which only detects the tallest layer of vegetation from satellite imagery), the approach taken was to classify the polygon by the dominant, i.e. tallest, layer of vegetation. Consequently, if a polygon included trees and scrub, the polygon was classified as having tree vegetation; or if it included scrub and grass, it was classified as scrub. The order of dominance was considered as follows: trees > scrub > heath/moorland > grass > unvegetated.

The ArcGIS Zonal Statistics tool was used to determine the LCM2000 composition of MasterMap polygons that could not be classified using only MasterMap data. The LCM2000 category with the highest proportional cover was used for classification. A

Table 2.2. Draft land cover typology and classification rules described using Boolean logic (AND should be applied between multiple properties). 'make', 'theme', 'descriptive group' and 'descriptive term' are properties of the MasterMap topology layer area polygons used in classification. Values listed in these columns may appear alongside other values provided they are not disallowed by the stated NOT rules. LCM2000 majority cover refers to the largest percentage cover of the MasterMap polygon by a single Land Cover Map 2000 category. See text for further explanation. Some categories are designated to multiple combinations of properties.

Category	MasterMap 'make'	MasterMap 'theme' and 'descriptive group'	MasterMap 'descriptive term'	LCM2000 majority cover
Building	Manmade	Building OR glasshouse		-
Manmade surface	Manmade	NOT (building OR glasshouse)		-
Garden	Multiple	-	-	-
Water	Natural	Water	-	-
Woodland (deciduous and mixed)	Natural	Natural environment NOT water	Non-coniferous trees OR scattered non-coniferous trees	-
	Natural OR unknown OR unclassified	NOT (natural environment OR water)	-	Deciduous/mix-ed woodland
Woodland (coniferous)	Natural	Natural environment NOT water	Coniferous trees OR scattered coniferous trees	-
	Natural OR unknown OR unclassified	NOT (natural environment OR water)	-	Coniferous woodland
Scrubland	Natural	Natural environment NOT water	Scrub NOT (non-coniferous trees OR scattered non-coniferous trees OR coniferous trees OR scattered coniferous trees)	-
Moorland (heath)	Natural	Natural environment NOT water	Heath NOT (non-coniferous trees OR scattered non-coniferous trees OR coniferous trees OR scattered coniferous trees OR scrub)	NOT bog
Moorland (bog)	Natural	Natural environment NOT water	Heath NOT (non-coniferous trees OR scattered non-coniferous trees OR coniferous trees OR scattered coniferous trees OR scrub)	Bog

small number of polygons (<30) could not be classified using MasterMap and were found to have a majority LCM2000 cover of fen/marsh/swamp. These polygons mainly appeared in unlikely locations (e.g. playing fields); they were therefore manually classified using aerial photography downloaded from Google Earth, which indicated

Table 2.2 continued.

Category	MasterMap 'make'	MasterMap 'theme' and 'descriptive group'	MasterMap 'descriptive term'	LCM2000 majority cover
Grassland (rough)	Natural	Natural environment NOT water	Rough grassland NOT (non-coniferous trees OR scattered non-coniferous trees OR coniferous trees OR scattered coniferous trees OR scrub OR heath)	-
	Natural OR unknown OR unclassified	NOT (natural environment OR water)	-	Natural grassland OR calcareous grassland OR acid grassland OR bracken
Grassland (improved)	Natural OR unknown OR unclassified	NOT (natural environment OR water)	-	Improved grassland
Unvegetated land	Natural	Natural environment NOT water	(Rocks OR scattered rocks OR boulders OR scattered boulders) NOT (non-coniferous trees OR scattered non-coniferous trees OR coniferous trees OR scattered coniferous trees OR scrub OR heath OR rough grassland)	-
	Natural OR unknown OR unclassified	NOT (natural environment OR water)	-	Inland bare ground
Arable (cereals)	Natural OR unknown OR unclassified	NOT (natural environment OR water)	-	Arable cereal
Arable (horticulture)	Natural OR unknown OR unclassified	NOT (natural environment OR water)	-	Arable horticulture
Unknown natural surface	Natural	NOT water	-	Suburban/rural developed OR continuous urban
Unknown surface	Unknown OR unclassified	NOT water	-	Suburban/rural developed OR continuous urban

that none of these locations actually were fen/marsh/swamp. Aside from this, classification was automated, using a VBA script according to the rules presented in Table 2.2.

2.2.1.3. *Legend validation – procedure*

The land cover classification method and draft typology were validated by georeferencing aerial photography downloaded from Google Earth, and classifying the majority land cover within polygons by eye. Regions for validation were selected by imposing a 250x250m grid over the map of Sheffield and using the proportion of impervious cover within grid squares to stratify each square into one of six quantiles according to urbanisation (although two quantiles included squares with zero impervious cover, as such squares account for around one third of the total). Ten grid squares were randomly selected from each quantile, with grid squares lying partially outside the Sheffield boundary excluded from selection. The sample grid squares used for validation were representative of the land cover distribution across the whole area, with ≤ 5% differences in the proportion of the area under each land cover (Table 2.3).

The main land uses in quantiles 1 and 2 (no impervious area) were farmland and woodland; one site included part of a reservoir and another was a small disused quarry, now scrubland. Quantile 3 (0% < impervious area < 3%) was mainly moorland, farmland and woodland, with some allotments and reservoir. Quantile 4 (3% < impervious area < 17%) included farmland and rural developments, with some suburban

Table 2.3. Land cover composition of the total study area, and the area sampled for validation, according to the draft typology. Proportions rounded to 2 d.p.

Category	Total	Sample
Arable (cereals)	0.01	0.01
Arable (horticulture)	0.03	0.03
Building	0.06	0.05
Garden	0.12	0.09
Grassland (improved)	0.12	0.19
Grassland (rough)	0.18	0.13
Manmade surface	0.10	0.10
Moorland (bog)	0.05	0.10
Moorland (heath)	0.11	0.06
Scrubland	0.01	0.01
Unvegetated land	0.01	0.01
Water	0.02	0.01
Woodland (coniferous)	0.03	0.04
Woodland (non-coniferous and mixed)	0.12	0.12
Unknown natural surface	0.05	0.05
Unknown surface	0.01	0.00

development, allotments, industrial development and railway infrastructure. Quantile 5 (17% < impervious area < 35%) was mainly residential and institutional urban areas, including some recreational grounds and other non-continuous urban development. Finally, quantile 6 (impervious area > 35%) was denser residential, institutional, commercial and industrial developments with significant transport infrastructure.

It should also be noted that there were instances where polygons did not line up with photographed features, which may have been due to poor georeferencing, temporal changes and/or errors in mapping. Because of this, as well as the fact that polygons were classified according only to majority cover, there is additional error not quantified by this validation.

2.2.1.4. Legend validation – results and discussion

The validation results are shown as confusion matrices. Each land cover category as classified by the automated procedure is shown in the first column (the row names), and the proportion of the sampled area with that cover is shown in italics in the second column. The remaining columns show the proportion of the row category area that was classified into the column's category during validation, rounded to two decimal places. It was not possible to visually differentiate cereal and horticultural arable covers during validation, so these categories are amalgamated into a single column. Empty boxes indicate that no polygons were classified as such, while a display of 0.00 means <0.005. Shaded boxes indicate where the classification 'should' lie if there was no error. The visual land cover classification of polygons that could not be classified automatically is shown at the bottom of each table.

The overall results are shown in Table 2.4a while the results for individual quantiles are shown in Table 2.4b to Table 2.4f. As mentioned above, the arable classes were impossible to differentiate visually. The visual differentiation between the two grassland and the two woodland categories may also have lower accuracy than that of the other classes, although cues such as livestock grazing, tractor marks, tree shape and colour were used as appropriate. Nevertheless this should be borne in mind during interpretation of the validation results.

The overall results (Table 2.4a) show the lowest accuracy for arable land (0.42), followed by rough grassland (0.58). However, much of the rough grassland was classified by eye as improved grassland, which is difficult to distinguish. Most incorrectly classified arable land was grassland. This has three possible explanations:

Table 2.4. Confusion matrix of land cover classification validation results over all quantiles. Abbreviated names in first column correspond to column headings. Cells in second column (in italics) show proportion of the sampled area under the land cover category named for that row. Remainder of cells show the proportion of the row category area classified into the column's category. Blank cells indicate no such combination; 0.00 means <0.005 (results shown to 2dp).

	Proportion of total area	Arable (cereal & horticulture)	Building	Garden	Grassland improved	Grassland rough	Manmade surface	Moorland (bog)	Moorland (heath)	Scrubland	Unvegetated land	Water	Woodland (coniferous)	Woodland (non-conif./ mixed)
a. Over all quantiles														
Ar. (cereal)	*0.01*				0.74	0.26								
Ar. (hort.)	*0.03*	0.42		0.01	0.40	0.11				0.03	0.01			0.02
Building	*0.05*		0.99	0.00			0.01			0.00	0.00			0.00
Garden	*0.09*		0.00	0.97	0.00		0.02			0.00				0.01
Gr. (imp.)	*0.19*	0.08		0.00	0.85	0.05	0.00			0.01	0.01			0.00
Gr. (rough)	*0.13*	0.00		0.00	0.20	0.58			0.16	0.02	0.02		0.01	0.02
Man. surf.	*0.10*		0.00	0.00	0.02	0.00	0.95		0.00	0.00	0.02			0.00
Moor (bog)	*0.10*							0.83	0.17					
Moor (hth)	*0.06*			0.00	0.01	0.04	0.00	0.10	0.85	0.00				0.01
Scrubland	*0.01*						0.01			0.78				0.21
Unveg land	*0.01*				0.07	0.02	0.01			0.08	0.79			0.04
Water	*0.01*											1.00		
Wd. (conif)	*0.04*					0.02							0.77	0.21
Wd. (nc/m)	*0.12*	0.03		0.00	0.18	0.04	0.00			0.02	0.02		0.00	0.70
Unknown natural surface	*0.05*	0.17	0.01	0.01	0.17	0.06	0.09			0.10	0.25			0.13
Unknown surface	*0.00*				0.61		0.17			0.01	0.17			0.05
b. Quantiles 1 and 2														
Ar. (cereal)	-													
Ar. (hort.)	*0.07*	0.58			0.31	0.08				0.01	0.01			0.01
Building	-													
Garden	*0.00*			1.00										
Gr. (imp.)	*0.19*	0.06			0.86	0.06	0.00			0.02	0.00			0.00
Gr. (rough)	*0.22*	0.00			0.16	0.56			0.22	0.00	0.02		0.01	0.02
Man. surf.	-													
Moor (bog)	*0.30*							0.83	0.17					
Moor (hth)	*0.14*							0.12	0.88					
Scrubland	*0.00*									1.00				
Unveg land	*0.00*										1.00			
Water	*0.00*											1.00		
Wd. (conif)	*0.02*					0.13							0.87	
Wd. (nc/m)	*0.02*				0.10									0.90
Unknown natural surface	*0.04*					0.00					1.00			
Unknown surface	-													

Table 2.4 continued.

	Proportion of total area	Arable (cereal & horticulture)	Building	Garden	Grassland improved	Grassland rough	Manmade surface	Moorland (bog)	Moorland (heath)	Scrubland	Unvegetated land	Water	Woodland (coniferous)	Woodland (non-conif./ mixed)
c. Quantile 3														
Ar. (cereal)	*0.02*				0.59	0.41								
Ar. (hort.)	*0.01*				1.00									
Building	*0.00*		0.91								0.08			0.02
Garden	*0.00*			0.91			0.09							
Gr. (imp.)	*0.29*	0.00		0.00	0.91	0.08				0.00				0.00
Gr. (rough)	*0.23*			0.00	0.14	0.73			0.10	0.03				0.00
Man. surf.	*0.01*						0.99		0.01					0.00
Moor (bog)	-													
Moor (hth)	*0.06*				0.02	0.23			0.75	0.00				
Scrubland	*0.01*									0.75				0.25
Unveg land	*0.00*					1.00								
Water	*0.01*											1.00		
Wd. (conif)	*0.12*					0.01							0.65	0.34
Wd. (nc/m)	*0.24*	0.06			0.28	0.08				0.01	0.02			0.54
Unknown natural surface	-													
Unknown surface	-													
d. Quantile 4														
Ar. (cereal)	*0.03*				0.83	0.17								
Ar. (hort.)	*0.01*			0.31	0.57					0.00				0.11
Building	*0.01*		0.99											0.01
Garden	*0.02*			0.97										0.03
Gr. (imp.)	*0.38*	0.17		0.00	0.80	0.00	0.00			0.03				0.01
Gr. (rough)	*0.05*			0.04	0.68	0.17				0.09				0.01
Man. surf.	*0.07*				0.06	0.01	0.78				0.14			0.00
Moor (bog)	-													
Moor (hth)	*0.00*													1.00
Scrubland	*0.02*									0.86				0.14
Unveg land	-													
Water	*0.05*											1.00		
Wd. (conif)	*0.02*												0.88	0.12
Wd. (nc/m)	*0.09*	0.04			0.22	0.01				0.02	0.00		0.00	0.70
Unknown natural surface	*0.24*	0.83		0.02			0.01			0.08	0.03			0.04
Unknown surface	-													

Table 2.4 continued.

	Proportion of total area	Arable (cereal & horticulture)	Building	Garden	Grassland improved	Grassland rough	Manmade surface	Moorland (bog)	Moorland (heath)	Scrubland	Unvegetated land	Water	Woodland (coniferous)	Woodland (non-conif./ mixed)
e. Quantile 5														
Ar. (cereal)	-													
Ar. (hort.)	*0.04*				0.52	0.29				0.11				0.08
Building	*0.13*		0.99	0.00			0.00				0.00			0.00
Garden	*0.39*		0.00	0.99	0.00		0.00			0.00				
Gr. (imp.)	*0.07*				0.78	0.08	0.02				0.08			0.04
Gr. (rough)	*0.01*				0.02	0.28					0.70			
Man. surf.	*0.16*			0.00	0.00	0.01	0.97			0.01	0.00			0.00
Moor (bog)	-													
Moor (hth)	*0.00*			0.02	0.90		0.07							
Scrubland	*0.01*						0.05			0.91				0.05
Unveg land	*0.08*										1.00			
Water	*0.02*											1.00		
Wd. (conif)	-													
Wd. (nc/m)	*0.04*				0.01					0.17				0.82
Unknown natural surface	*0.01*	0.17		0.00	0.22	0.05	0.02			0.05	0.36			0.12
Unknown surface	*0.04*				0.69		0.09			0.01	0.21			
f. Quantile 6														
Ar. (cereal)	-													
Ar. (hort.)	-													
Building	*0.16*		0.98	0.00			0.02			0.00				0.00
Garden	*0.13*		0.00	0.88			0.05			0.01				0.05
Gr. (imp.)	-													
Gr. (rough)	*0.03*				0.32	0.68					0.00			
Man. surf.	*0.34*		0.00	0.00	0.02		0.97			0.00	0.00			0.01
Moor (bog)	-													
Moor (hth)	-													
Scrubland	*0.01*									0.41				0.59
Unveg land	*0.19*				0.34		0.03			0.41	0.01			0.21
Water	*0.00*											1.00		
Wd. (conif)	*0.01*													1.00
Wd. (nc/m)	*0.00*			0.00	0.02	0.05	0.01			0.07	0.02			0.84
Unknown natural surface	*0.00*		0.01	0.01	0.20	0.08	0.15			0.13	0.25			0.16
Unknown surface	*0.13*				0.30		0.47			0.01				0.22

poor ability to resolve these categories from aerial photography (i.e. validation errors); field rotation systems; or a temporal trend towards converting arable land to grazing land in the decade between LCM2000 satellite imagine and Google Earth aerial photography, with error arising as a consequence of the automated classification procedure (see Section 2.2.1.2). In total, 2.5% of the sampled area classified automatically as arable was visually classified as grassland, while 1.5% was classified vice versa, suggesting field rotation accounts for much of the disparity. Since the pattern of arable and grassland will change annually there is no way of accurately accounting for it, and as such the automatic classification is not altered.

The only other categories with accuracy <0.8 were the two woodland categories. For coniferous woodland, the majority of inaccuracy likely arises from difficulty in differentiating the two woodland categories using aerial photography. However, a considerable amount of non-coniferous and mixed woodland was classified as improved grassland. This may be due to genuine land cover changes, but is more likely to be a consequence of automatically classifying polygons as woodland even if they are described as contain multiple land covers including sparse trees (as described in Section 2.2.1.2). Nevertheless, this decision was made because the structurally dominant vegetation, especially trees, tends to be the most important provider of ecosystem services, with greater levels of, for example, carbon storage, reduction of stormwater runoff and removal of gaseous air pollutants (USDA-NRCS 1986, Cruickshank et al. 2000, Zhang et al. 2001).

Another problem, arising in rural areas, was misclassification between rough grassland and moorland (polygons classified automatically as rough grassland were visually determined to be mostly heath). This mainly arises from the fact that the polygons in these areas were exceptionally large, while land cover varied in management and vegetational succession at smaller scales.

In quantile 5 much of the automatically classified grassland appeared to be unvegetated land (Table 2.4e), although this only accounted for a very small proportion of the total cover. This was located at the site of an industrial works, suggesting recent disturbance. In quantile 6 most of the automatically classified unvegetated land was visually determined to be grass-, scrub- or woodland (Table 2.4f), suggesting growth of new layers of vegetation due either to planting or to natural reinvasion. Similarly, scrubland largely had slightly lower (but still good) accuracy rates than other categories, with much of it appearing to be woodland; this was also true for other quantiles, suggesting either difficulty in visual differentiation or growth from saplings/shrubs into larger trees and bushes. These kinds of temporal changes unfortunately cannot be taken into account in analysis.

These problems were not considered to be great enough to necessitate manipulation of the actual polygon geometry, given the time investment this would have required. There was also no obvious way of improving the automated classification procedure for the stated purpose of ecosystem service modelling, except to amalgamate the two arable classes due to poor discrimination ability. The grassland categories were left unaltered, in order to provide a snapshot of the arable rotation system and also because MasterMap surveying and LCM2000 spectral analysis were likely to have greater discernment ability than classification from aerial photography in this particular case. The woodland categories were also not amalgamated for this latter reason.

2.2.1.5. Final land cover map and legend

The final land cover legend is described in Table 2.5. The numerical codes and category names used in Table 2.5 are referenced throughout this study. The map itself is shown in Map 2 (maps shown in Appendix D).

Table 2.5. The final land cover legend used in this study. Final column shows the percentage of the study area under each land cover.

	Land cover	Description	%
1	Arable	Land used for arable farming	3.9
2	Building	Built structures, including sheds and glasshouses	5.5
3	Garden	Private land associated with housing	11.7
4	Grassland (improved)	Land dominated by managed grass	11.9
5	Grassland (rough)	Land dominated by semi-natural grass	17.5
6	Manmade surface	Impervious ground-level surfaces	10.0
7	Moorland (bog)	Upland areas dominated by bog	4.8
8	Moorland (heath)	Upland areas dominated by dwarf shrub heath	10.9
9	Scrubland	Land dominated by shrub or herbaceous vegetation	1.5
10	Unknown natural surface	Unknown pervious land cover	4.9
11	Unknown surface	Unknown land cover (may be pervious or impervious)	0.6
12	Unvegetated land	Pervious land without vegetation	0.6
13	Water	Inland water bodies including rivers and reservoirs	0.6
14	Woodland (coniferous)	Land dominated by coniferous woodland	2.7
15	Woodland (non-coniferous and mixed)	Land dominated by non-coniferous or mixed woodland	12.0

For the purposes of ecosystem modelling, gardens, unknown surfaces and unknown natural surfaces are treated as mixed-type land covers. Unknown and unknown natural surfaces are given the same land cover composition as determined by validation (Table 2.4). Gardens are given the land cover composition determined for Sheffield gardens by Tratalos et al. (2007): 37% improved grassland, 33% manmade surface, 22% scrub and 8% unvegetated land.

Table 2.5 also shows the proportion of the study area that is under each land cover type. From the land cover map it can be estimated that 19.9% of Sheffield is under impervious cover, i.e. buildings and manmade surfaces (presuming that assumptions about the proportions of gardens, unknown natural surfaces and unknown surfaces under impervious cover are correct). An estimated 75.5% of the study area is vegetated. The remaining 4.5% is either water or unvegetated land.

2.2.1.6. Comparison with Land Cover Map 2000

The work in this section documents the creation of a novel map of land cover within Sheffield. The key benefit of this new map is that it resolves small areas of land (e.g. gardens, individual buildings, roads and paths) and borders between land cover types more accurately than the existing LCM2000 is able to. This is particularly valuable in urban areas, where land covers are often heterogeneous at a fine spatial scale.

Nevertheless, LCM2000 is at the time of writing the key data source for UK-wide land cover. Table 2.6 shows the land cover composition of Sheffield according to the new land cover map and according to LCM2000, with land cover classes aggregated to facilitate comparison. Within broad categories, the land cover compositions are within a few percent of each other, suggesting that the new map is at least reasonably accurate. Differences in land cover composition between LCM2000 and the new map likely arise due to differences in the legends and the scope of types of land cover within them, and also because of the ability of the new map to resolve much smaller land parcels.

2.2.2. Land use mapping

2.2.2.1. Map source

The land use map used in this project is from the South Yorkshire Historic Environment Character GIS dataset. This dataset was produced as part of English Heritage's nation-wide historic characterisation programme, which aims to improve the

Table 2.6. Comparison of land cover composition of new land cover map of Sheffield, and Land Cover Map 2000 of the same area.

Land cover category	%	LCM2000 category	%
Arable	**3.9**		**5.2**
1 Arable	3.9	4.1 Cereals	1.3
		4.2 Other	3.8
Developed	**27.2**		**32.1**
2 Building	5.5	17.1 Suburban/rural developed	21.3
6 Manmade surface	10.0	17.1 Continuous urban	10.8
3 Garden	11.7		
Improved grassland	**11.9**		**12.9**
4 Grassland (improved)	11.9	5.1 Improved grassland	12.9
Rough grassland	**17.5**		**15.7**
5 Grassland (rough)	17.5	6.1 Rough grassland	9.6
		7.1 Calcareous grassland	2.8
		8.1 Acid grassland	3.3
Moorland	**15.7**		**14.8**
7 Moorland (bog)	4.8	10.1 Dwarf shrub heath	9.0
8 Moorland (heath)	10.9	10.2 Open shrub heath	2.3
		12.1 Bog	3.5
Unvegetated land	**0.6**		**1.5**
12 Unvegetated land	0.6	16.1 Inland bare ground	1.5
Water	**0.6**		**1.2**
13 Water	0.6	13.1 Inland water	1.2
Coniferous woodland	**2.7**		**3.3**
14 Woodland (coniferous)	2.7	2.1 Coniferous woodland	3.3
Non-coniferous & mixed woodland	**12.0**		**13.1**
15 Woodland (non-coniferous & mixed)	12.0	1.1 Broad-leaved and mixed woodland	13.1
Other & unknown	**7.0**		**0.3**
9 Scrubland	1.5	9.1 Bracken	0.3
10 Unknown natural surface	4.9	11.1 Fen, marsh, swamp	<0.1
11 Unknown surface	0.6		

understanding and management of historic environment resources (South Yorkshire Archaeology Service 2005). The area that is presently the metropolitan borough of Sheffield was characterised in 2005.

The characterisation describes the different land uses of each parcel of land through time, usually at least as far back as 1066. Sheffield is split into 2393 land parcel polygons in the GIS dataset, with wide variation in the size of individual polygons. The polygon size is determined by the area of each historically “distinct character area”

(South Yorkshire Archaeology Service 2005). In urbanised areas this is commonly 5-10 ha, while agricultural polygons are typically around 90 ha (South Yorkshire Archaeology Service 2005); the largest polygons of moorland are 1000-3000 ha. Characterisation was performed using current and historical maps and aerial photography, with some field visits (South Yorkshire Archaeology Service 2005).

Each polygon is associated with at least one database record indicating the character type and dates of that character, as well as other attributes depending on the character type (South Yorkshire Archaeology Service 2005). For the present study, however, only the most recent land character is of interest. This land character is used to indicate present land use.

2.2.2.2. *Land use typology*

The typology of the Historic Environment Characterisation is hierarchical, with twelve broad character types that each include one or more specific character types. Table 2.7 shows this typology (only classes that are present in the study area are shown). Further explanation of the character types can be found in the dataset documentation (South Yorkshire Archaeology Service 2005).

2.2.2.3. *Land use map*

Map 3 illustrates the Historic Environment Character GIS dataset polygons and the present day broad character types in the study area. A very small area of reservoir at the western edge of the study area is not included in the Historic Environment Character dataset; this area is also pointed out in Map 3.

Table 2.8 shows the proportions of land within the study area that falls into each of the broad character types listed in Table 2.7, and the proportions of the twenty one narrower character types with a prevalence of 1.0% or greater.

2.2.2.4. *Use of land use data in ecosystem service models*

The land cover map did not become available until a relatively late stage of the project, after most of the ecosystem service models had been completed; so this map was only used in the cultural ecosystem services model. Land use data could potentially have been used in the carbon storage model, in order to take the consequences of management regimes beyond simple land cover into account. The index of habitat for biodiversity might also have been designed differently if land use data had been

Table 2.7. Character types present in the study area.

Broad character	Character types
Commercial	Business park, suburban commercial core, urban commercial core, distribution centre, entertainment complex, markets, offices, retail park, shopping centre, warehousing
Communications	Airport, bus depot, canal lock ladder system, canal or river wharf, car park, motorway and trunk road junctions, ring road/bypass, train depot/sidings, train station, tram depot, transport interchange, tunnel portal, viaducts/aqueducts
Enclosed land	Agglomerated fields, assarts, crofts, piecemeal enclosure, strip fields, surveyed enclosure (parliamentary/private), valley floor meadows
Extractive	Clay pits/brickworks, landfill, quarry, reclaimed coal mine, refractory material mine and works, spoil heap
Horticulture	Nursery
Industrial	Chemical, craft industry, heavy metal trades, light metal trades, support metal trades, textile trade, utilities, water powered site, other industry
Institutional	Barracks, cemetery, civil & municipal buildings, fortified site, hospital complex, municiptal depot, nursing home/almshouse, religious workshop, school, university or college, workhouse/orphanage/children's home, other military, other religious
Ornamental, parkland and recreational	Allotments, caravan/camping site, deer park, golf course, inner city farm, leisure centre, playing fields/recreation ground, private parkland, public park, public square, sports ground, tourist attraction
Residential	Back to back/courtyard houses, burgage plots, elite residence, farm complex, high rise flats, low rise flats, planned estate (social housing), prefabs, private housing estate, Romany or other traveller community site, semi-detached housing, terraced housing, vernacular cottages, villas/detached housing
Unenclosed land	Common and greens, moorland, regenerated scrubland
Water bodies	Reservoir
Woodland	Ancient woodland, plantation, semi-natural woodland, spring wood

available, for example to estimate the regularity and degree of anthropogenic disturbance. Nevertheless, the lack of such data did not inhibit the modelling of these two ecosystem services – as it would have done for cultural ecosystem services.

2.2.3. Soil mapping

2.2.3.1. Map source and coverage

The soils data used in this study are derived from the LANDis National Soil Map GIS dataset (NATMAP Vector) and associated soil attribute data (SOILSERIES and

Table 2.8. Land use composition of Sheffield. (a) All broad character types, listed in order of decreasing area cover. (b) Character types with at least 1.0% area cover.

a. Broad character type	**%**
Unenclosed land	27.8
Enclosed land	22.4
Residential	22.2
Woodland	8.9
Ornamental, parkland & recreational	5.6
Industrial	4.4
Institutional	3.1
Commercial	2.1
Water bodies	1.4
Communications	1.2
Extractive	0.8
Horticulture	<0.1

b. Character type	**%**
Moorland	25.6
Surveyed enclosure	8.8
Planned estate (social housing)	8.2
Piecemeal enclosure	7.6
Semi-detached housing	4.8
Plantation	4.0
Ancient woodland	3.4
Agglomerated fields	3.1
Private housing estate	3.1
Villas/detached housing	2.5
Terraced housing	2.4
Regenerated scrubland	2.1
Public park	1.9
School	1.9
Heavy metal trades	1.7
Strip fields	1.6
Other industry	1.5
Semi natural woodland	1.4
Reservoir	1.4
Assarts	1.1
Golf course	1.0

HORIZON data tables). The GIS has a spatial scale of 1:250000 and is based on published soils maps and surveying of additional areas (Cranfield University 2009).

The map is composed of 'map units' that are identified by soil associations, i.e. combinations of soil series. Each soil association is described by a name, description, geology, characteristics and composition of soil series. Data tables define the soil series, including its hydrological properties and pesticide and agronomy-related characteristics; and further tables describe the soil profile of each series in terms of horizons, with information on the physical, chemical and hydraulic properties of each horizon. For many soils, multiple estimates are given for different land use groups: arable, permanent grass, ley grassland and other.

NATMAP data was not available for the small portion of the study area (approximately 7.7%) lying outside of the Don Catchment. For these areas, the soil properties of the closest known area were used. Map 4 shows the extent of the NATMAP dataset and the area not covered.

2.2.3.2. Soils in the study area

Map 4 shows the spatial distribution of soil types in the study area. Table 2.9 shows a description of the soils and geology of each soil association, as well as the composition of soil series within soil associations; a brief description of the soil series is given in Table 2.10. Further soil properties required for ecosystem service modelling are given in the relevant chapters.

The distribution of soils can be summarised as follows. Peat soils occur in the westernmost parts of the study area, where moorland is present. Slightly further east the predominant soils have peaty surface layers with highly acidic loamy subsurface layers. Over the majority of the remainder of the study area the soils are seasonally wet loams, although some areas are better drained. There is also a vein of silty/clayey soils associated with rivers.

2.2.4. Modelled air pollutant concentration

The spatial air pollution data used in the reduction of air pollution model is derived from output from Sheffield City Council's AIRVIRO model, courtesy of Andrew Elleker at Sheffield City Council. AIRVIRO models pollutant dispersion using knowledge of point and diffuse pollution sources. AIRVIRO output was obtained for a series of dispersion scenarios of nitrogen oxides (NO_x) and particulate matter (PM_{10}), using typical meteorological conditions for each month of the year (Andrew Elleker, personal communication). A finer temporal resolution could not be obtained due to time and data handling constraints. The spatial resolution is $500m^2$.

The modelled field does not cover the western edge of the study area of moorland, as shown in Figure 2.1. However, the only diffuse pollutant sources in this area are two roads, both of which are dead ends and so are unlikely to be busy. In addition, there are believed to be no point sources of pollution in this area. Therefore the grid squares not covered by the modelled field are given the pollutant concentration of the closest covered grid square. This is likely to be a small overestimation, as pollutant concentrations diminish with distance from Sheffield centre, but the values at the edge

Table 2.9. Description and soil series composition of NATMAP soil associations present within the study area. Data source: Cranfield University (2009).

Association	Soil description	Geology	Series	%
6 Lake	Lake or water body		9982	100
54106 Rivington 1	Well drained coarse loamy soils over sandstone. Locally associated with similar soils affected by groundwater.	Carboniferous and Jurassic sandstone	1713 1243	67 33
54107 Rivington 2	Well drained coarse loamy soils over rock. Some fine loamy soils with slowly permeable subsoils and slight seasonal waterlogging.	Palaeozoic sandstone and shale	1713 2243 718	55 28 17
54125 East Keswick 2	Deep well drained fine and coarse loamy soils.	Drift from Palaeozoic and Mesozoic sandstone and shale	406 1303 2225	65 20 15
63101 Anglezarke	Well drained very acid coarse loamy soils over sandstone with a bleached subsurface horizon. Some shallow soils with a peaty or humose surface horizon.	Palaeozoic and Mesozoic sandstone	17 1735	60 40
65101 Belmont	Coarse loamy very acid upland soils over rock with a wet peaty surface horizon and thin ironpan. Some shallow peaty soils.	Carboniferous and Jurassic sandstone	113 1735 17	62 19 19
71201 Dale	Slowly permeable seasonally waterlogged clayey, fine loamy over clayey and fine silty soils on soft rock often stoneless.	Carboniferous and Jurassic clay and shale	300 1933 103	60 20 20
71301 Bardsey	Slowly permeable seasonally waterlogged loamy over clayey and fine silty soils over soft rock. Some well drained coarse loamy soils over harder rock.	Carboniferous mudstone with interbedded sandstone	103 1713 1933 718	41 29 18 12
72103 Wilcocks 1	Slowly permeable seasonally waterlogged fine loamy and fine loamy over clayey upland soils with a peaty surface horizon. Coarse loamy soils affected by groundwater in places. Very acid where not limed.	Drift from Palaeozoic sandstone mudstone and shale	2235 1022 508	72 17 11
81102 Conway	Deep stoneless fine silty and clayey soils variably affected by groundwater.	River alluvium	236 228 505	59 24 17
101102 Winter Hill	Thick very acid raw peat soils. Perennially wet.	Blanket peat	2242	100

Table 2.10. Name and description of NATMAP soil series present within the study area. Data source: Cranfield University (2009).

#	Series name	Series description
17	Anglezarke	Light loamy material over lithoskeletal sandstone
103	Bardsey	Medium loamy or medium silty drift over clayey material passing to clay or soft mudstone
113	Belmont	Loamy material over lithoskeletal sandstone
228	Clwyd	Medium silty river alluvium
236	Conway	Medium silty river alluvium
300	Dale	Clayey material passing to clay or soft mudstone
406	East Keswick	Medium loamy drift with siliceous stones
505	Fladbury	Clayey river alluvium
508	Fordham	Light loamy drift with siliceous stones
718	Heapey	Medium loamy material over lithoskeletal siltstone and sandstone
1022	Kielder	Loamy over clayey drift with siliceous stones
1243	Melbourne	Light loamy material over lithoskeletal sandstone
1303	Neath	Medium loamy material over lithoskeletal sandstone
1713	Rivington	Light loamy material over lithoskeletal sandstone
1735	Revidge	Loamy or peaty lithoskeletal sandstone
1933	Ticknall	Medium silty material passing to soft shale or siltstone
2225	Wick	Light loamy drift with siliceous stones
2235	Wilcocks	Loamy drift with siliceous stones
2242	Winter Hill	Mixed eriophorum and sphagnum peat
2243	Withnell	Light loamy material over lithoskeletal sandstone
9982	(Water)	-

of the field are already very low. It should also be noted that Gaussian models are optimal for areas below 20km^2, which is smaller than the actual area modelled (26km^2); although this is not such a problem as to prevent the use of this model for regulatory reporting work.

The modelled pollutant concentration is shown in Figure 2.2. The pattern is similar for both NO_x and PM_{10}, with the highest levels of pollution in the city centre, and also high close to the northeast border with Rotherham. There is also a hotspot of PM_{10} pollution in the southwest of Sheffield, presumably due to the industrial works in the area. There is far less pollution in the west of the study area.

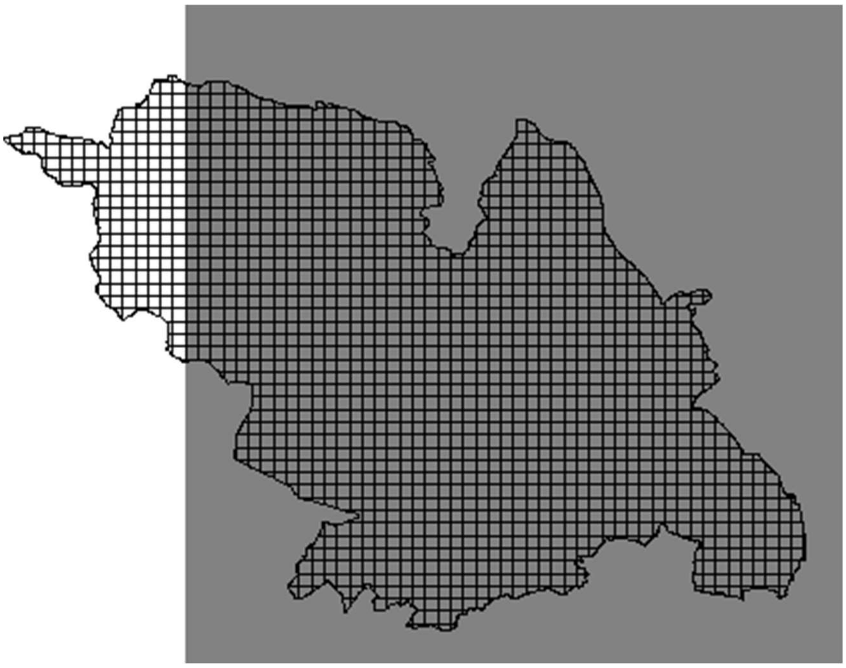


Figure 2.1. AIRVIRO model field (shaded grey). Black outlines show study area with 500m grid squares.

2.2.5. Relationships between land cover, land use and soil type

Map 2, Map 3 and Map 4 show the distribution of land covers, land uses and soil types within Sheffield respectively. Some qualitative patterns can be observed between these maps. For example, comparison of Map 2 and Map 3 reveals that the moorland predominating in the west of the study area is shown in the land use map primarily as unenclosed land. Further east, there is a belt of mainly enclosed land, i.e. farmland, which thus shows on the land cover map as improved grassland and arable. It is clear from the land use map that within the densely developed urban area, there are large residential areas, which show on the land cover map as areas predominated by gardens. The proportion of gardens then decreases towards the urban centre (which is centred on the river network), with a concurrent increase in the amount of manmade surface. On the land use map, the urban centre shows that industrial, commercial and communication land uses predominate. These patterns also correspond with changes in the dominant soil types (Map 4).

These patterns confirm that certain land uses are associated with particular land covers, and likewise tend to occur on certain soil types. However, there are also cases where, for example, the maps indicate that one land cover can be found within multiple land uses. For example, rough grassland is present, albeit in variable proportions, throughout the study area.

Both types of pattern can be further understood by quantitative analysis. This section analyses and discusses how land cover, land use and soil type are related to each other spatially within the study area. The aim is to identify patterns that may later assist in interpreting and identifying the implications of the ecosystem service model output.

2.2.5.1. Land cover versus land use distribution

Table 2.11 shows the areal distributions of land covers within land uses, while Table 2.12 shows the converse: the composition of land covers in terms of land use. (It should be borne in mind that both maps have some associated error when interpreting these tables.) These two tables confirm some obvious spatial coincidences between land cover and land use: for example, unenclosed land contains land covers that are usually unmanaged or extensively managed (rough grassland, moorland) and, similarly, almost all moorland is unenclosed land. The land use and land cover categories encompassing water and woodlands also provide more or less the same spatial information.

The tables also confirm a high level of land cover heterogeneity within more urban land use types. Institutional land uses, for example, contain a high proportion of buildings, grassland, manmade surfaces and woodlands, while ornamental, parkland and recreational uses contain a various vegetation types as well as some manmade surfaces.

2.2.5.2. Land cover versus soil type distribution

Table 2.13 shows the areal proportion of each land cover type that exists over the various soil types in Sheffield, and Table 2.14 shows the proportional distribution of soil types found under each land cover type. There are fewer strong associations than

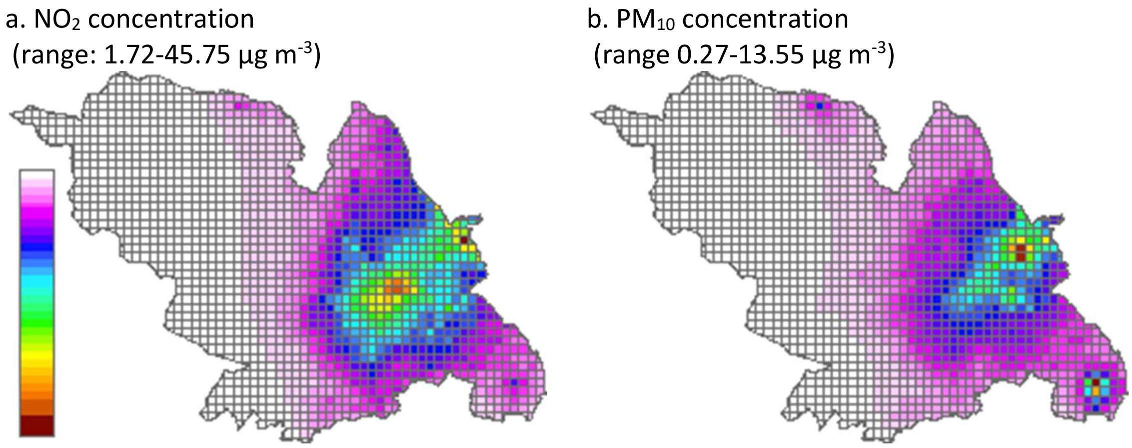


Figure 2.2. Annual average pollutant concentrations across Sheffield for (a) NO_2 and (b) PM_{10}. Maps use the pseudo-continuous linear colour ramp shown in (a), where white represents the lowest values in the graph and deep red the highest values.

was seen for land cover and land use distributions. Soil ID 6 is water bodies, so a strong association is expected with the water land cover type; and soil type 63101 is only found in one small polygon in the study area, so even though all soil of this type is found under rough grassland, very little of the total rough grassland is over this soil.

There is a substantial association between the moorland land cover types and the thick peat soil type 101101, and the moderately peaty types 65101 and 72103. Aside from these, the distribution of land covers over soil types, and vice versa, tends to be quite wide.

2.2.5.3. *Land use versus soil type distribution*

Table 2.15 shows the areal distribution of land uses over each soil type, and Table 2.16 shows the proportions of different soil types under each land use type. Once again there are few informative associations. Soil type 6 is water so is expected to be

Table 2.11. Areal distribution of land cover types (listed in first column) within land uses (listed in header row). Values shown as proportions, rounded to 2 d.p. Cells shaded according to value; dark and light greens indicate high and low proportions respectively. Blank cells indicate no such combination; 0.00 indicates <0.005.

Land cover type	Commercial	Communications	Enclosed Land	Extractive	Horticulture	Industrial	Institutional	Ornamental, Parkland and Recreational	Residential	Unenclosed Land	Water Bodies	Woodland
1 Arable	0.01	0.00	0.14	0.06	0.06	0.01	0.05	0.02	0.01	0.00	0.01	0.01
2 Building	0.23	0.04	0.01	0.03	0.02	0.22	0.10	0.02	0.15	0.00	0.00	0.00
3 Garden	0.04	0.00	0.01	0.01	0.10	0.01	0.03	0.01	0.49	0.00	0.00	0.01
4 Grassland (imp.)	0.01	0.01	0.41	0.02	0.00	0.01	0.15	0.22	0.01	0.02	0.03	0.02
5 Grassland (rough)	0.02	0.08	0.24	0.07	0.30	0.02	0.14	0.26	0.02	0.33	0.07	0.03
6 Manmade surface	0.46	0.31	0.02	0.07	0.01	0.39	0.23	0.09	0.21	0.01	0.02	0.03
7 Moorland (bog)			0.00							0.17		
8 Moorland (heath)	0.00	0.00	0.01	0.00		0.00	0.00	0.00	0.00	0.38	0.00	0.01
9 Scrubland	0.01	0.03	0.02	0.06		0.02	0.00	0.02	0.00	0.02	0.00	0.01
10 Unkn. nat. surf.	0.13	0.23	0.02	0.37	0.51	0.15	0.16	0.11	0.06	0.01	0.01	0.01
11 Unknown surf.	0.01	0.02	0.00	0.00		0.01	0.02	0.01	0.01	0.00	0.00	0.00
12 Unveg. land	0.01	0.01	0.01	0.00		0.01	0.00	0.00	0.00	0.01	0.00	0.00
13 Water	0.01	0.02	0.00	0.02		0.04	0.00	0.01	0.00	0.00	0.76	0.01
14 Wood. (conif.)	0.00	0.01	0.01	0.00		0.00	0.00	0.00	0.00	0.00	0.02	0.26
15 Wood. (nc/m)	0.05	0.23	0.10	0.29	0.00	0.11	0.10	0.23	0.03	0.04	0.06	0.60
Sum	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*

associated with the water bodies land use; and the associations between soil type 63101 and enclosed land in Table 2.15, and between horticulture and soil type 71201 in Table 2.16, both occur because of very small sample sizes.

The association between ornamental, parkland and recreational land uses and soil type 101102 is driven by the fact that the bog and heath moorlands, which form part of the Peak District National Park, are included in this land use category. Unenclosed land is generally found over less well drained soil types i.e. 71201, 71301 (Table 2.16), and unsurprisingly the better drained soil types (54106, 54107, 54125) are more appropriate for land enclosures (Table 2.15). However, there do not appear to be any other strong and meaningful associations.

2.2.5.4. Synthesis

This section has analysed the degree of spatial coincidence of land use types, land cover types and soil types in Sheffield. The results suggest that some (but not all) land cover types are associated with particular land uses (Section 2.2.5.1). This is important

Table 2.12. Areal distribution of land use types (listed in first column) within land covers (listed in header row). Values shown as proportions, rounded to 2 d.p. Cells shaded according to value; dark and light greens indicate high and low proportions respectively. Blank cells indicate no such combination; 0.00 indicates <0.005.

Land use type	1 Arable	2 Building	3 Garden	4 Grassland (improved)	5 Grassland (rough)	6 Manmade surface	7 Moorland (bog)	8 Moorland (heath)	9 Scrubland	10 Unknown natural surface	11 Unknown surface	12 Unvegetated land	13 Water	14 Woodland (coniferous)	15 Woodland (non-coniferous/mixed)
Commercial	0.01	0.09	0.01	0.00	0.00	0.10		0.00	0.01	0.06	0.05	0.04	0.02	0.00	0.01
Communications	0.00	0.01	0.00	0.00	0.01	0.04		0.00	0.02	0.06	0.04	0.01	0.02	0.00	0.02
Enclosed Land	0.83	0.02	0.03	0.78	0.30	0.05	0.00	0.02	0.33	0.10	0.02	0.29	0.03	0.04	0.18
Extractive	0.01	0.00	0.00	0.00	0.00	0.01		0.00	0.03	0.06	0.00	0.00	0.01	0.00	0.02
Horticulture	0.00	0.00	0.00	0.00	0.00	0.00				0.00					0.00
Industrial	0.01	0.18	0.00	0.00	0.01	0.17		0.00	0.06	0.14	0.09	0.05	0.10	0.00	0.04
Institutional	0.04	0.06	0.01	0.04	0.02	0.07		0.00	0.01	0.10	0.09	0.02	0.00	0.00	0.03
Orn./Park./Rec.	0.03	0.02	0.01	0.10	0.08	0.05		0.00	0.07	0.13	0.07	0.01	0.03	0.01	0.11
Residential	0.03	0.61	0.94	0.01	0.02	0.46		0.00	0.03	0.29	0.55	0.01	0.01	0.01	0.06
Unenc. Land	0.02	0.00	0.00	0.04	0.53	0.02	1.00	0.97	0.38	0.05	0.07	0.56	0.03	0.04	0.09
Water Bodies	0.00	0.00	0.00	0.00	0.01	0.00		0.00	0.00	0.00	0.00	0.01	0.67	0.01	0.01
Woodland	0.02	0.00	0.01	0.01	0.02	0.02		0.01	0.05	0.02	0.00	0.01	0.08	0.88	0.45
Sum	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*

because, to some extent, the land use associated with a patch of a given land cover type will affect its management. Woodland in plantations, for example, will be managed primarily for productivity, whereas woodland in parks will be managed for aesthetics and safety. Likewise, grassland in enclosed land will often be used for pasture or hay; in contrast, grassy areas in parks and gardens are often managed as lawns. Therefore the ecosystem services provided by a patch of land will depend upon the way it is used by humans. In order words, the interaction between land use and land cover is critical to the ecosystem services provided by any patch of land (Verburg et al. 2009). Table 2.11 and Table 2.12 confirm the spatial disparities between land use and land cover distributions, and thus confirm that, where adequate data for parameterisation exist, understanding both land use and land cover should improve ecosystem service assessments.

The results also suggest that the physical environment imposes certain restrictions on what a given parcel of land can be used for. The most obvious case of this is for bog

Table 2.13. Areal distribution of land cover types (listed in first column) within soil types (listed in header row; described in Table 2.9). Values shown as proportions, rounded to 2 d.p. Cells shaded with dark/light greens indicate high/low proportions respectively. Blank cells indicate no such combination; 0.00 indicates <0.005.

Land cover type	6	54106	54107	54125	63101	65101	71201	71301	72103	81102	101102
1 Arable	0.00	0.11	0.04	0.04		0.02	0.06	0.03	0.01	0.02	
2 Building	0.00	0.06	0.01	0.03		0.00	0.08	0.12	0.00	0.20	0.00
3 Garden	0.00	0.16	0.02	0.07		0.00	0.19	0.26	0.00	0.02	
4 Grassland (imp.)	0.00	0.23	0.39	0.16		0.07	0.17	0.05	0.11	0.01	
5 Grassland (rough)	0.01	0.14	0.25	0.11	1.00	0.23	0.09	0.07	0.38	0.05	0.08
6 Manmade surface	0.01	0.10	0.02	0.09		0.01	0.14	0.21	0.01	0.40	0.00
7 Moorland (bog)						0.03			0.00		0.33
8 Moorland (heath)	0.00	0.01	0.01	0.00		0.20	0.00	0.00	0.24	0.00	0.58
9 Scrubland	0.02	0.01	0.02	0.01		0.03	0.01	0.01	0.03	0.01	0.00
10 Unkn. nat. surf.	0.00	0.04	0.02	0.05		0.01	0.07	0.12	0.00	0.14	0.00
11 Unknown surf.	0.00	0.00	0.00	0.00			0.01	0.02		0.01	
12 Unveg. land	0.00	0.01	0.02	0.00		0.00	0.01	0.00	0.01	0.01	0.00
13 Water	0.91	0.00	0.01	0.03		0.01	0.00	0.00	0.00	0.05	0.00
14 Wood. (conif.)	0.01	0.01	0.04	0.04		0.09	0.00	0.00	0.11	0.00	0.00
15 Wood. (nc/m)	0.03	0.12	0.15	0.36		0.29	0.16	0.10	0.09	0.08	0.00
Sum	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*

Table 2.14. Areal distribution of soil types (listed in first column; described in Table 2.9) within land covers (listed in header row). Values shown as proportions, rounded to 2 d.p. Cells shaded with dark/light greens indicate high/low proportions respectively. Blank cells indicate no such combination; 0.00 indicates <0.005.

Soil ID	1 Arable	2 Building	3 Garden	4 Grassland (improved)	5 Grassland (rough)	6 Manmade surface	7 Moorland (bog)	8 Moorland (heath)	9 Scrubland	10 Unknown natural surface	11 Unknown surface	12 Unvegetated land	13 Water	14 Woodland (coniferous)	15 Woodland (non-coniferous/mixed)
6	0.00	0.00	0.00	0.00	0.00	0.00		0.00	0.01	0.00	0.00	0.00	0.63	0.00	0.00
54106	0.33	0.13	0.17	0.23	0.12	0.12		0.01	0.11	0.11	0.09	0.19	0.02	0.04	0.12
54107	0.06	0.01	0.01	0.17	0.10	0.01		0.01	0.08	0.02	0.00	0.14	0.02	0.09	0.06
54125	0.02	0.01	0.01	0.03	0.02	0.02		0.00	0.02	0.02	0.01	0.00	0.04	0.04	0.06
63101					0.00										
65101	0.03	0.00	0.00	0.04	0.11	0.01	0.06	0.13	0.13	0.02		0.03	0.05	0.22	0.16
71201	0.35	0.36	0.40	0.33	0.16	0.35		0.00	0.21	0.32	0.33	0.19	0.08	0.04	0.33
71301	0.16	0.39	0.40	0.08	0.10	0.38		0.00	0.14	0.43	0.50	0.10	0.04	0.01	0.15
72103	0.03	0.00	0.00	0.12	0.35	0.01	0.00	0.32	0.27	0.00		0.29	0.03	0.53	0.09
81102	0.01	0.09	0.00	0.00	0.01	0.10		0.00	0.02	0.07	0.07	0.02	0.08	0.00	0.02
101102		0.00			0.05	0.00	0.94	0.53	0.00	0.00		0.03	0.00	0.02	0.00
Sum	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*

and heath moorlands, which occur over peaty soil types (especially bog moorlands). This soil type appears to limit usage as productive land (e.g. agriculture) or for urbanisation (unless it is drained, but even in this case it is unlikely that the land would be especially productive due to its highland nature); and most of these areas are either unenclosed or parkland/recreational land. There is also some indication that particular (well drained) soil types are especially useful as both arable and pasture land.

In conclusion, these patterns indicate that soil types to some extent restrict the types of land cover that can occur at a site, and that some land uses – especially those not associated with urbanisation – involve particular land cover types. These constraints may play a role in generating patterns in the production of the ecosystem services that are modelled in the subsequent chapters.

2.3. Spatial units of analysis

Investigation of the spatial properties of ecosystem services requires that maps are presented at an appropriate spatial level. The level that is appropriate depends to some

Table 2.15. Areal distribution of land use types (listed in first column) within soil types (listed in header row; described in Table 2.9). Values shown as proportions, rounded to 2 d.p. Cells shaded with dark/light greens indicate high/low proportions respectively. Blank cells indicate no such combination; 0.00 indicates <0.005.

Land use type	6	54106	54107	54125	63101	65101	71201	71301	72103	81102	101102
Commercial		0.00		0.00			0.01	0.04	0.00	0.08	
Communications		0.01		0.01			0.03	0.04		0.15	
Enclosed Land	0.00	0.47	0.74	0.41	1.00	0.22	0.28	0.05	0.24	0.01	0.00
Extractive	0.00		0.00	0.02		0.03	0.02	0.01	0.00	0.01	
Horticulture							0.00				
Industrial	0.04	0.01	0.01	0.10		0.00	0.05	0.08	0.00	0.56	
Institutional	0.00	0.05	0.00	0.01			0.05	0.07	0.00	0.01	
Orn./Park./Rec.	0.00	0.04	0.07	0.00		0.41	0.03	0.03	0.57	0.06	0.99
Residential	0.02	0.04	0.04	0.02		0.00	0.10	0.13	0.00	0.07	
Unenc. Land	0.00	0.32	0.03	0.13		0.00	0.36	0.51	0.01	0.05	0.00
Water Bodies	0.91	0.00	0.01	0.03		0.01	0.00		0.01		
Woodland	0.01	0.07	0.11	0.27		0.32	0.08	0.03	0.17	0.00	0.00
Sum	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*

Table 2.16. Areal distribution of soil types (listed in first column; described in Table 2.9) within land use types (listed in header row). Values shown as proportions, rounded to 2 d.p. Cells shaded with dark/light greens indicate high/low proportions respectively. Blank cells indicate no such combination; 0.00 indicates <0.005.

Soil ID	Commercial	Communications	Enclosed Land	Extractive	Horticulture	Industrial	Institutional	Ornamental, Parkland and Recreational	Residential	Unenclosed Land	Water Bodies	Woodland
6			0.00	0.00		0.01	0.00	0.00	0.00	0.00	0.70	0.00
54106	0.04	0.08	0.25			0.02	0.19	0.02	0.08	0.17	0.01	0.10
54107			0.17	0.02		0.01	0.00	0.02	0.04	0.01	0.03	0.07
54125	0.01	0.01	0.04	0.04		0.05	0.01	0.00	0.01	0.01	0.05	0.07
63101			0.00									
65101			0.06	0.21		0.01		0.13	0.00	0.00	0.06	0.23
71201	0.15	0.35	0.30	0.47	1.00	0.26	0.38	0.04	0.41	0.38	0.09	0.21
71301	0.62	0.38	0.04	0.23		0.32	0.41	0.02	0.43	0.41		0.07
72103	0.01		0.13	0.00		0.00	0.01	0.35	0.00	0.00	0.07	0.24
81102	0.17	0.18	0.00	0.02		0.33	0.01	0.01	0.03	0.01		0.00
101102			0.00					0.42		0.00		0.00
Sum	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*	*1.00*

extent on the question being asked. For example, in spatial ecology it is typical to impose an arbitrary grid, with the variables of interest being aggregated to or averaged over each grid square. In contrast, a social investigation would benefit from analysis over socially homogeneous spatial areas.

For the purposes of this thesis, three types of spatial units of analysis have been used. The use of particular spatial units for particular analyses is explained in the relevant chapters. Inclusion of multiple types of spatial unit also facilitates discussion of the implications of choosing a given spatial level of analysis.

2.3.1. 500m grid squares

Initially, ecosystem services were modelled over a 500 m^2 grid imposed within the boundaries of the unitary authority of Sheffield (derived from GIS data from the Office for National Statistics 2004). Figure 2.3 shows the 500 m^2 grid. In total, there are 1624 grid squares in the study area, of which 299 are partial squares located at the edge of the grid and therefore have an area of less than 0.25 square kilometers.

2.3.2. Historic Environment Character areas

To facilitate the analysis of the relationships between urban morphology and ecosystem services presented in Chapter 10, a GIS layer identifying morphologically homogeneous areas was desirable. The South Yorkshire Historic Environment Character (HEC) GIS dataset, also used to derive land use (see Section 2.2.1.6), was used to fulfil this purpose. It was reasoned that since each GIS polygon represents an area of land with a unique history, including building date (South Yorkshire Archaeology Service 2005), then these polygons would signify areas with a relatively homogeneous design (in comparison with other available GIS layers). Thus the second unit of analysis is the Historic Environment Character polygons, clipped to the boundaries of the Sheffield unitary authority, and with small fragments of polygons at the very edge of the area (presumed to be mostly outside of Sheffield) identified by eye and excluded from analyses.

Figure 2.4 shows the HEC polygons used as spatial units of analysis. There are in total 2347 included polygons. The mean polygon size is 0.154 square kilometers, although the distribution of sizes is heavily skewed towards smaller polygons and the median size is 0.037 square kilometers. The smallest polygon is only 2.44 square meters (note the units), while the largest is 35.734 square kilometers. The smallest polygons are generally located in built up areas where land uses are heterogeneous at smaller scales.

2.3.3. Output Areas

Finally, Chapter 11 involves a socioeconomic analysis of ecosystem service access. Chapter 11 therefore uses 2001 Census Output Areas (OAs) as units of analysis. OA boundaries were generated following analysis of the 2001 Census data, using automated clustering of maximally socially homogeneous adjacent postcode areas (except where postcodes lie over electoral ward boundaries, in which case the postcode was split into two), with social homogeneity determined from household tenure and dwelling type (Office for National Statistics 2007).

OAs are presently the lowest level at which census data is aggregated for statistics, with an average of 125 households and a minimum of 40 (Office for National Statistics 2007). Figure 2.5 shows Sheffield's 1744 OAs, which have a mean area of 0.211 square kilometers. Again the distribution of sizes is skewed towards smaller polygons and the median size is 0.055 square kilometers. The range of polygon sizes is 0.005 to 55.005 square kilometers. Because OA boundaries are built to consider population density, the polygons are smaller in built up areas with higher population density.

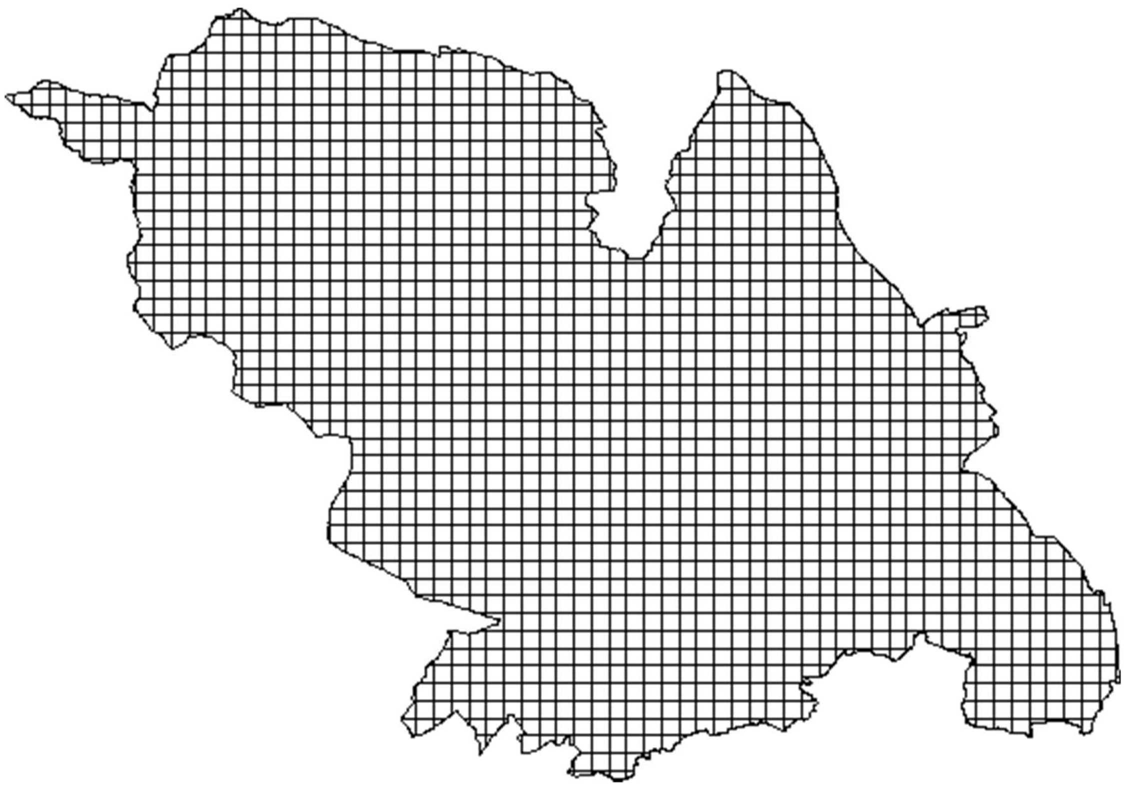


Figure 2.3. 500m^2 grid used as spatial units of analysis.

Figure 2.4. Historic Environment Character areas used as spatial units of analysis. The very small areas of red at the edge of the region show polygon fragments excluded from analysis (there are others that are not visible at this scale).

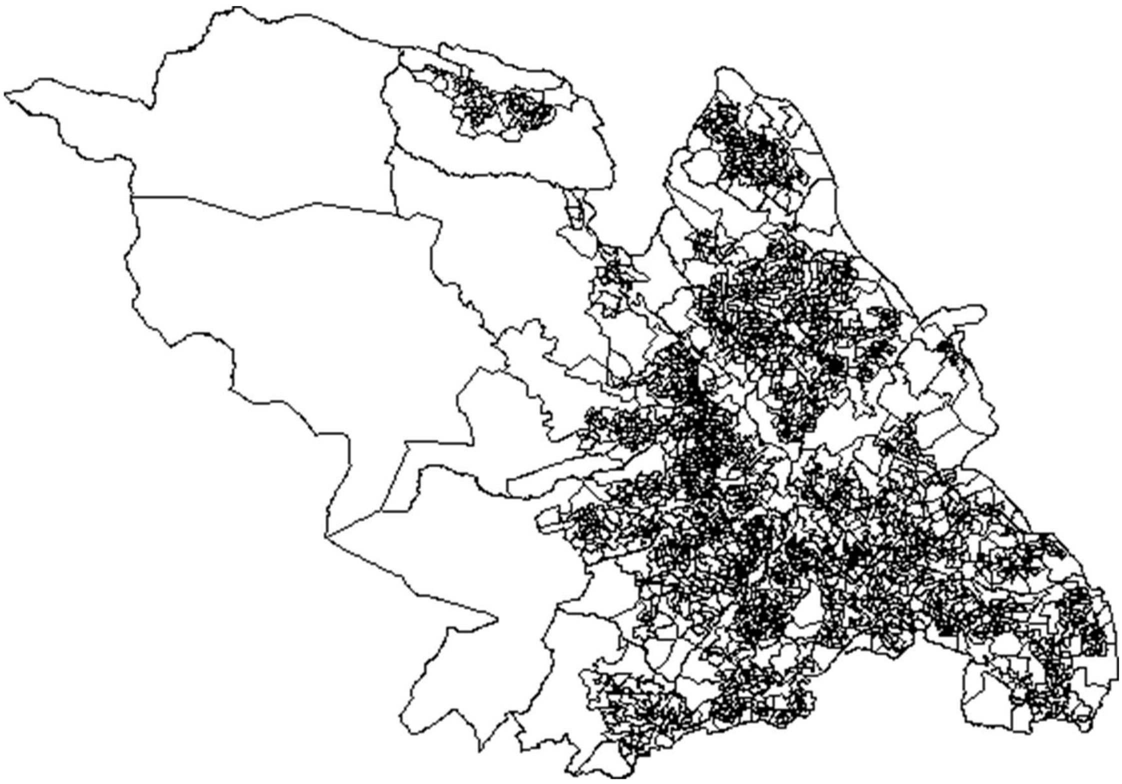


Figure 2.5. Output Areas used as spatial units of analysis.

2.4. Summary

This chapter documents the selection of the ecosystem services that will form the core focus of this thesis, and the creation and collation of key spatial data necessary for modelling these ecosystem services. It also describes the different spatial units used in the analysis of ecosystem services that will be presented in the following chapters.

The land cover, land use, soil type and air pollution maps introduced here provide necessary spatial data for ecosystem service modelling. A novel land cover map has been developed specifically for use in this study, which is especially appropriate for use in studies of urban areas. This land cover map, and the land use and soils maps, were analysed to identify differences and similarities in the spatial distribution of these three characteristics over the study area. The analysis illustrated that some land uses, land covers and soil types have characteristic patterns of co-occurrence, while others are more heterogeneous. As well as showing the importance of understanding spatial distributions of both land cover and land use in ecosystem service analysis, the results may help to provide some context for interpreting spatial patterns of ecosystem service production.

The following six chapters go on to describe and analyse each of the ecosystem service models in turn.

3. Reduction of air pollution

3.1. Introduction

Air pollution is a common environmental health hazard in cities (Brimblecombe 2001, Tunnicliffe and Ayres 2001, Matthias et al. 2006, Fenger 2009). High concentrations of airborne pollutants in cities usually originate primarily from combustion of fuels: as cities have high population densities they also have high levels of fuel use for purposes such as transport, industry and heating buildings (Brimblecombe 2001, Matthias et al. 2006, Fenger 2009). There are at least 3000 different types of anthropogenic airborne pollutant, of which several are recognised as major and widespread problems associated with centres of population: sulphur dioxide, particulate matter, nitrogen dioxide, carbon monoxide, lead and ozone (Fenger 2009).

Within Sheffield, nitrogen dioxide (NO_2) and fine particulate matter (with a particle diameter of <10 μm; PM_{10}) are particular problems (Sheffield City Council 2008a, Sheffield City Council 2008b, Elleker 2009). In some parts of the city, NO_2 levels exceed the annual mean target set in the UK by DEFRA, while PM_{10} levels are likely to exceed the 24 hour mean target unacceptably frequently (Sheffield City Council 2008a, Sheffield City Council 2008b, Elleker 2009). Maps of the distribution of these two pollutants are shown in Section 2.2.4.

Within a city, air pollution is emitted from point sources such as industrial sites and residences burning solid fuel, as well as diffuse sources such as road traffic (Elleker 2009). Following emission, pollutants can be transported by air currents across large areas, meaning it is possible for problems associated with air pollution to extend across and even beyond the urban area (Brimblecombe 2001, Matthias et al. 2006, Fenger 2009). Historically, air pollution in Sheffield was primarily the result of heavy industry, but following pollution control measures and later the collapse of these industries (Section 1.5.2.3), traffic has become the major source of NO_2 and PM_{10} pollution (Elleker 2009). Consequently, it is areas close to the major roads that are most likely to have unacceptably high levels of NO_2 pollution, and PM_{10} is a potential problem in large areas of the city (Sheffield City Council 2008b, Sheffield City Council 2008a).

Pollutants remain airborne until one of three events occurs: the pollutant reacts chemically and is transformed into a less damaging chemical species; the pollutant is deposited on a surface; or the pollutant is scavenged by water droplets in fog, cloud or precipitation (Zannetti 1990, Matthias et al. 2006, Fenger 2009). Urban greenspaces increase the rate of pollutant removal by providing more efficient deposition surfaces (and a greater surface area) than artificial surfaces, at least for all the major pollutant types listed above except lead (Zhang et al. 2001, Nowak and Crane 2002, Zhang et al. 2002, Zhang et al. 2003). The model of air pollution reduction therefore focuses on the amount of pollution removed over and above that which would be removed in the absence of the greenspace infrastructure for Sheffield's two most problematic pollutants: NO_2 and PM_{10}.

The purpose of this chapter is to introduce and describe the air pollution reduction model. The remainder of this introduction briefly describes the two pollutants and their human and environmental health implications, in order to demonstrate the importance of the service that greenspace is providing by reducing pollution levels; the process of pollutant deposition; and how the ecosystem service of air pollution reduction is generated. There is then an overview of the model, with further technical details found in Appendix A.1. Finally the model output is presented and discussed.

3.1.1. Nitrogen dioxide

Nitrogen oxides (NO_x), of which NO_2 is a component, are produced mainly by combustion in vehicle engines (Brimblecombe 2001, Matthias et al. 2006, Fenger 2009). It is difficult to isolate the effects of ambient NO_2 concentrations from those of other pollutants with which levels of NO_2 correlate (World Health Organization 2006). However, NO_2 is a contributor to photochemical smog, the chemicals in which can cause human health problems including reduced respiratory and circulatory system function, and increased infection risk (Tunnicliffe and Ayres 2001, Matthias et al. 2006). Long term exposure to NO_2 has also been associated with an increase in the severity of asthma and reduced lung growth (World Health Organization 2006).

The World Health Organization (WHO) has set guidelines for annual and one hour mean NO_2 exposure, which are also used as DEFRA air quality targets (World Health Organization 2006, Elleker 2009). It is the annual mean target that is exceeded in parts of Sheffield. Studies in Sheffield have shown that this has a significant identifiable effect on stroke and coronary heart disease mortality: rates of these mortalities are 37%

and 17% greater respectively in the part of Sheffield with the highest, compared to the lowest, ambient NO_x levels, after controlling for age, sex, socioeconomic deprivation and smoking prevalence (Maheswaran et al. 2005a, Maheswaran et al. 2005b).

With regard to environmental damage, high levels of smog chemicals damage plants either by entering stomata or settling on leaves, and causing foliage lesions and reduced growth (Matthias et al. 2006). Smog chemicals also contribute to the greenhouse effect (Matthias et al. 2006), with consequences for global warming.

3.1.2. Particulate matter

PM_{10} is a mixture of solid and liquid particles, including soot and smoke, dust, metals and other chemicals, which remain suspended in the air due to their small size (Maynard 2001, Matthias et al. 2006). Key sources of PM_{10} emissions are vehicle exhaust, high temperature industrial processes, and also agriculture, construction and fires (Matthias et al. 2006, Walworth and Pepper 2006). PM_{10} is responsible for atmospheric haze due to its ability to scatter light; alteration of radiation and thermal budgets; and fouling of buildings and other surfaces (Brimblecombe 2001, Matthias et al. 2006, Walworth and Pepper 2006, Fenger 2009).

PM_{10} enters the body via inhalation and is deposited inside the respiratory organs, with smaller particles penetrating deeper and being thought to cause more damage (Maynard 2001, Walworth and Pepper 2006). PM_{10} inhalation can reduce pulmonary function, cause chronic coughs and bronchitis, and exacerbate asthma (Maynard 2001, Matthias et al. 2006, Walworth and Pepper 2006).

DEFRA have also adopted the WHO's annual and 24 hour mean air quality guidelines as targets for maximum PM_{10} exposure (World Health Organization 2006, Elleker 2009). In the case of Sheffield, it is repeated short term exposure that exceeds the target (Elleker 2009). Studies in Sheffield indicate that there is again an increase in mortality associated with the levels of PM_{10} air pollution across the city, with excess risks of stroke and coronary heart disease mortalities of 33% and 8% respectively in areas with the highest PM_{10} levels compared to those with the lowest (Maheswaran et al. 2005a, Maheswaran et al. 2005b).

Other studies have found increased respiratory and cardiac mortality, hospitalisations and school absenteeism associated with $PM_{10} > 20\ \mu g\ m^{-3}$ (Vedal, 1995, in Walworth and Pepper 2006). The Sheffield Air Map live monitoring website suggests these levels do frequently occur for short periods of time (http://sheffieldairmap.org/; accessed 11/08/2009).

3.1.3. Deposition of air pollutants

Air pollutants are removed from the atmosphere by deposition to surfaces (Zannetti 1990), sometimes following chemical reaction (Brimblecombe 2001, Matthias et al. 2006, Fenger 2009). Dry deposition is deposition to land cover surfaces, and wet deposition is adsorption to water droplets that are subsequently precipitated or impacted to the surface (Zannetti 1990). The focus here is on dry deposition, as these rates are more directly influenced by land cover, whereas weather conditions are the main determinant of wet deposition rates.

The rate of pollutant removal, or pollutant deposition flux, is a product of the pollutant concentration and the effective velocity at which the pollutant is deposited – note that it is not a real velocity, as the deposition of most pollutants is not dominated by gravitational effects (Zannetti 1990). The dry deposition velocity of gaseous pollutants such as NO_2 is the inverse sum of resistances to deposition in three layers above and at the surface: the atmospheric surface boundary layer where turbulence is the dominant process; the thin layer just above the surface where diffusion processes are important; and the vegetation or other surface layer onto which pollutants are actually deposited, in which the solubility and oxidising capacity of the pollutant are important parameters (Zannetti 1990). Figure 3.1a shows how these resistances combine to determine the deposition rate. Meteorological conditions are critical to all three resistances, due to their influence over turbulent and chemical processes (Zannetti 1990, Zhang et al. 2002, Zhang et al. 2003, Environmental Protection Agency 2004). If there is a high resistance to deposition, the deposition velocity is low, and vice versa.
For particulate pollution, the dry deposition velocity is the inverse sum of resistances in the atmospheric surface boundary layer and vegetation/other surface layer (with the particulate size distribution being key to the efficiency of removal of pollutants by diffusion, interception and impaction), plus a gravitational settling term, which describes the influence of gravity in depositing pollutants (Zannetti 1990, Zhang et al. 2001, Environmental Protection Agency 2004). The pathway of particulate deposition is shown in Figure 3.1b. Again, meteorological conditions are highly important.

The canopy resistance (for NO_2) or surface resistance (for PM_{10}) is strongly influenced by land cover, or more specifically the physical and chemical properties of

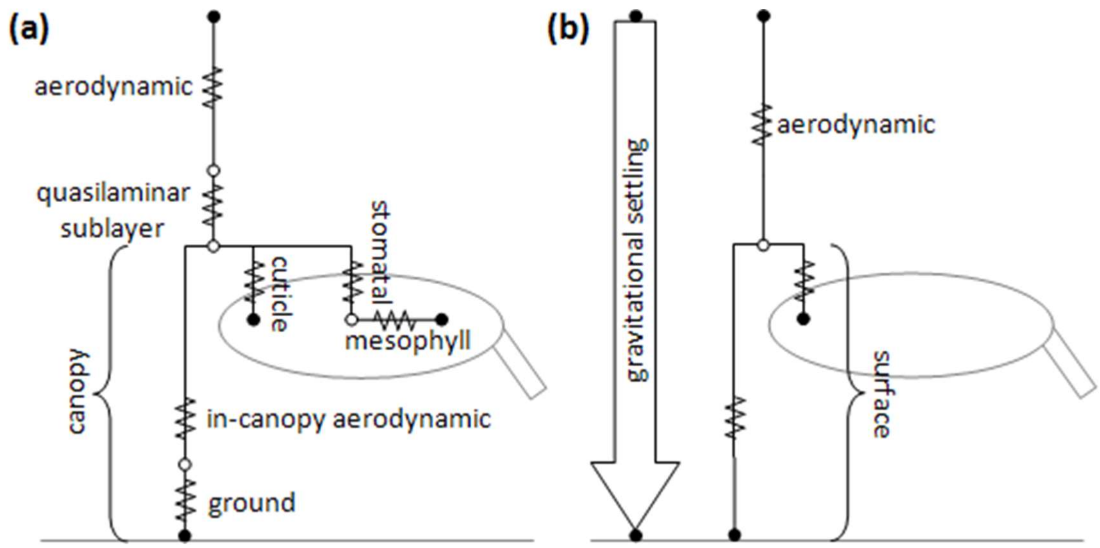


Figure 3.1. Resistance factors determining the rate of pollutant deposition to plant and land cover surfaces for (a) nitrogen dioxide and (b) <10μm particulate matter. Black dots represent the start and end points of pollutant particles; white dots represent 'junctions' between resistances; zig-zags represent the resistances. (a) after Hicks et al. (1987).

the ground and/or vegetation surfaces (Zannetti 1990, Zhang et al. 2001, Zhang et al. 2002). Deposition velocity is thought to be usually higher to vegetated surfaces than to artificial surface; although deposition to artificial surfaces is not zero (Zhang et al. 2001, Zhang et al. 2003). This is because plants take up gaseous pollutants such as NO_2 by molecular diffusion via the stomata or remove them from the air by chemical reaction with the plant surface, and because plant surfaces provide a large surface area for gravitational settling and collection of particles (Hicks et al. 1987, Zhang et al. 2001).

3.1.4. Generation of the ecosystem service

Table 2.1 in the previous chapter briefly summarised factors influencing the generation of the ecosystem service of air pollution reduction; this section provides more detail. The potential production (i.e. the capacity to produce the ecosystem service, according to the definitions given in Section 1.2.1) of air pollution reduction is the capacity of plant and ground surfaces to intercept pollutants or, more technically, the inverse of the resistance to deposition, or deposition velocity (Section 3.1.3). This value is strongly influenced by characteristics of the land cover and meteorological conditions (Zhang et al. 2001, Zhang et al. 2003). The actual production of the ecosystem service, however, is also dependent upon how the interception capacity is used, which is determined by the local levels of air pollution.

Deposition to non-natural surfaces is non-zero (Zhang et al. 2001, Zhang et al. 2003). Thus in order to capture the ecosystem service provided by the natural environment, it is necessary to calculate the additional deposition that occurs due to natural land covers, over and above what would occur if only artificial surfaces were present. This consideration is included in the modelling process.

The value of this ecosystem service derives from a reduction in ambient air pollution levels, and includes a reduction in the occurrence of ill health, reduced fouling of surfaces, and non-use benefits such as, for example, knowing that the air over the culturally valued Peak District National Park (to the west of the study area) is clean. These benefits have both economic value and value to human health and well-being.

The supply of these benefits depends on humans being at the sites where pollution is reduced. Due to movement of air in the atmosphere, airborne pollutants disperse from the point of emission and continue to disperse until they are removed from the atmosphere (Zannetti 1990). Consequently, the places where pollution levels are reduced are not necessarily the places where pollutant deposition occurs. However, it was not possible to create an integrated deposition and dispersion model within the scope of this study. Therefore the production, but not the supply or value, of the ecosystem service is modelled in this thesis.

Figure 3.2 shows a simplified conceptual model of how the prevailing meteorological conditions, the extent and types of land cover, and pollutant emissions interact in the environmental process of pollutant deposition, resulting in socioeconomic benefits to humans. The inputs to this figure are themselves the result of complex processes, but those processes are not modelled here.

3.2. Model overview

The pollutant deposition model was based on the deposition algorithms implemented in the Meteorological Service of Canada's multiple pollutant model ‘AURAMS - A Unified Regional Air-quality Modelling System’ (Zhang et al. 2001, Zhang et al. 2002, Zhang et al. 2003). Where parts of this formulation were inadequately reported, or the approach was unsuitable given the available input data, the formulation of the US EPA's regulatory AERMOD model was used (Environmental Protection Agency 2004).

The model implemented here is also broadly similar to the USDA's UFORE (Urban FORestry Effects) air pollutant removal model (Nowak et al. 1998), which has previously been used in several city-scale analyses (e.g. Escobedo et al. (2008), Nowak et al. (2002a) and references in Nowak et al. (2008)); although more recently developed formulations were used here.

Full technical details of the air pollution reduction model can be found in Appendix A.1. In brief, inputs to the model comprised land cover, meteorological and pollutant concentration data, as well as a number of empirical parameters such as process rates and constants. Further details about the sources and derivation of input data can be found in the Appendix A.1.2. These data were used to estimate the deposition velocity, V_d, of the pollutants to the different types of land cover found in the study area according to the schema illustrated in Figure 3.1. The estimates of V_d were then superimposed on the land cover map (Section 2.2.1), and an average calculated across each 500m grid square (Section 2.3.1). This was also the spatial resolution of the estimates of pollutant concentration, C, from the Sheffield City Council's AIRVIRO model (Section 2.2.4). These two inputs were used to calculate the total flux of pollutant deposition to the land covers, F, for each 500m grid square, as:

$$F = V_d C \tag{1}$$

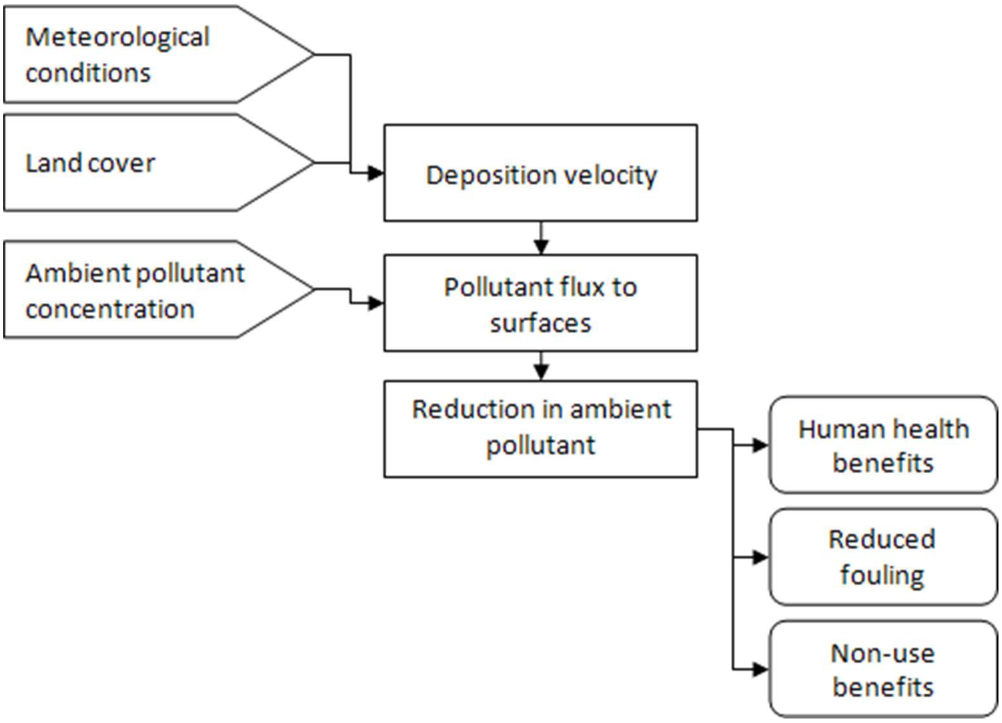


Figure 3.2. Conceptual representation of the pollutant deposition model. Input conditions are represented by pentagrams; ecosystem processes by rectangles; and socioeconomic benefits by ellipses.

Full technical details of the computation of F can be found in Appendix A.1.1. The spatial and temporal resolutions of the model were constrained by that of the AIRVIRO model input data, which were supplied as monthly averages per $500m^2$.

A second estimate of F assuming that all land cover was artificial, i.e. no ecosystem service was being produced, was also made (specifically, assuming that each $500m^2$ area was covered by 28% buildings and 72% manmade surfaces, which is the actual ratio of buildings to manmade surfaces in the study area). The first estimate (from the land cover map) was then divided by the second (assuming no natural land covers) to calculate the ecosystem service provided by the presence of natural land covers, and the monthly figures averaged to produce a single annual average for each pollutant. The figures were divided rather than subtracted in order to give equal weighting to the two pollutants, concentrations of which occur over different orders of magnitude; the frequency distribution of ecosystem service values for either pollutant singly is the same whether division or subtraction is used.

Finally, the mean of these two figures was taken to generate a single index of ecosystem service production for each $500m^2$ area. To produce maps at Output Area and Historic Environment Character area scales (Section 2.3), the pollutant concentration for each polygon was determined by area-weighted means from the 500m squares. The land cover composition of each polygon was used to calculate the deposition velocity, and was multiplied by the area-weighted mean pollutant concentration to calculate the pollutant flux according to Eqn. (1).

3.2.1. Sensitivity tests

As this model uses a large number of input variables, sensitivity tests were carried out to identify which parameters have the greatest effect on model output and whether the uncertainty associated with those variables might have implications for the reliability of the output. For full details of the sensitivity testing procedure and results, see Appendix A.1.4.

The NO_2 model was found only to have high sensitivity to an empirical constant that is believed to be accurately known, and several meteorological variables. The data for the meteorological variables are likely to be the greatest source of inaccuracy in the model, because the measurement error is unknown and more importantly because a single measurement is applied to the whole study area.

The PM_{10} model was found to be highly sensitive to one meteorological variable, with the same implications as for the NO_2 model. It is also sensitive to the size distribution and density of the particles; neither of these is empirically measured but use estimates from previous work. This means that the results of the model should be taken to refer to only possible type of PM_{10}, and that different particle size distributions or densities could give very different results.

3.2.2. Model limitations

The implementation of the air pollution model in this thesis has several limitations that may reduce the accuracy of the model results. Firstly, the meteorological data are not spatially resolved, but use a single data value across the whole study area. This means that factors such as the elevational gradient within the study area and smaller elevational effects, heat island effects, and terrain frictional forces are neglected. Additionally, wind direction effects are omitted entirely. Unfortunately there was no feasible way to improve this situation.

The model formulations here are mostly taken from Zhang et al. (2001, 2002, 2003). These formulations were specified for AURAMS, a regional scale unified meteorological driver, emissions processing and chemical transport model (Gong et al. 2006). The range of scales over which AURAMS has previously be applied range from 42km^2, 15 minute spatiotemporal resolution over eastern Canada and the eastern US for a period of six days; to 2km^2, three minute resolution over a 300km^2 area (Zhang et al. 2001, Gong et al. 2006, Cho et al. 2009). However, Zhang et al. (2001, 2002, 2003) do not suggest that the formulations are inapplicable at other scales, and the basic model type is the same as that used in within-city scale analyses by Nowak et al. (1998, 2002a) and Escobedo et al. (2008), but with more recent formulations. Therefore it is believed that this model is suitable for the present spatiotemporal resolution and scales, with the following exceptions.

This model implementation applies land cover specific parameters to areas that are often very small (e.g. single houses, gardens). However, many parameters are only valid for larger, uniform areas where meteorological conditions, especially turbulence, are able to develop. This is mostly likely to be problematic for the roughness length (z_0), which is related to the height of the structures interfering with turbulent processes at the land surface, and is only valid over uniform terrain and over areas large enough for the local roughness elements to dominate roughness effects on local turbulence

(Zannetti 1990). z_0, however, can also be applied to non-uniform terrain as an average value (Silva et al. 2007). It seems likely that higher z_0 values in nearby areas would increase the effective z_0 of small areas, but such effects could not be feasibly included in this model implementation. Boundary and heterogeneity effects are also not considered by Nowak et al. (1998, 2002a) or Escobedo et al. (2008), whose model is not spatially resolved but operates only on total areas of vegetated land. Therefore the output of the present model should be acceptably accurate, despite this limitation.

Although it is not unsuitable for this modelling method, the temporal resolution of this model is a limitation. While the deposition velocity (V_d) can be estimated for each hour, hourly values are averaged to produce a monthly value for each land cover to match the resolution of pollutant concentration (C) data. This may be significant because both V_d for gaseous pollutants and C of pollutants both commonly show diurnal cycles (Mayer 1999, Zhang et al. 2002, Zhang et al. 2003). If the cycles of C and V_d coincide, use of a monthly average for C will not accurately capture the average value of F. However, no higher resolution pollution data source was available.

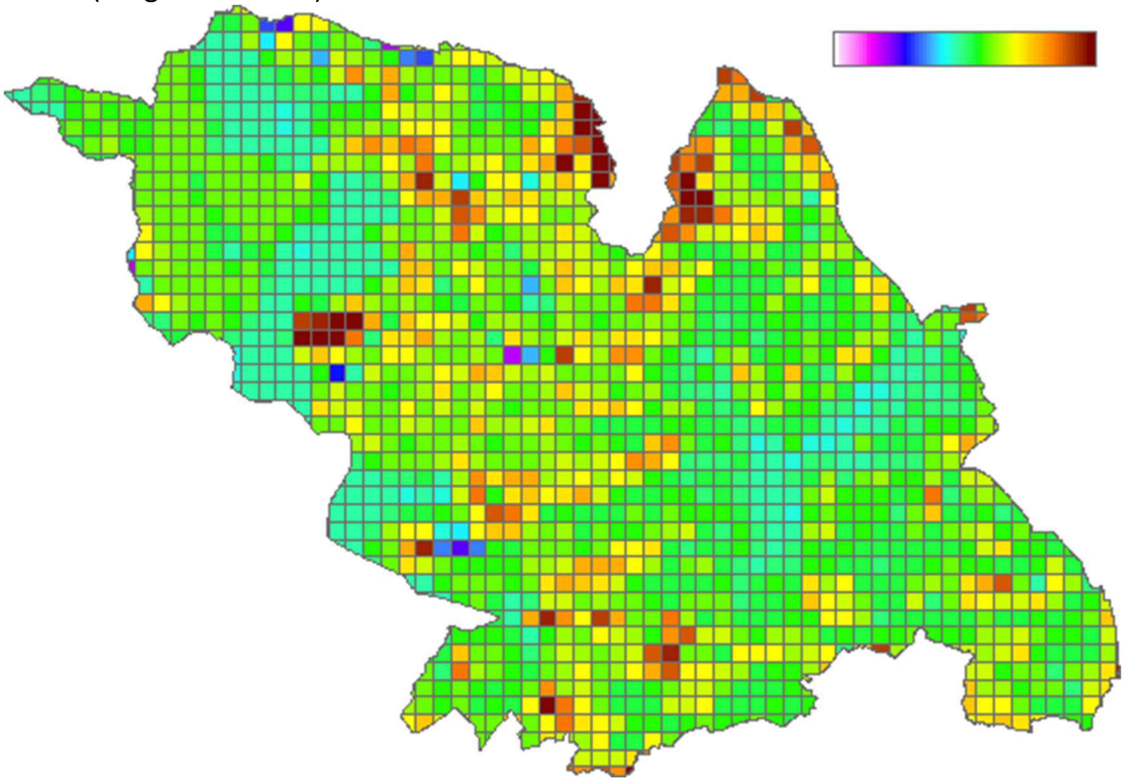
3.3. Results

3.3.1. Model output

Figure 3.3 shows the modelled standardised pollutant fluxes, i.e. the amount of pollutant deposition that occurs standardised by that which would occur if there were no natural land covers, for NO_2 and PM_{10}. The patterns are similar between the two pollutants, but over different numerical scales. The highest rates of deposition for both pollutants are in areas with a high proportion of woodland cover; however, for PM_{10} coniferous woodlands are far better at removing the pollutant than any other land cover type, accentuating the difference between these other areas; whereas for NO_2 non-coniferous and mixed woodlands are similarly good and there is less of a difference between woodlands and other land cover types. In general, deposition is greatest in the agricultural belt in the middle of the study area, which is interspersed with woodlands; and lower in the moorlands and urbanised area to either side.

These patterns are a consequence of the rates of deposition velocity for different land covers, shown in Figure 3.4. Figure 3.4 also shows that some land cover types have slower deposition velocities than manmade surfaces and buildings. This is most

a. NO_2 (range: 0.27-1.92)



b. PM_{10} (range: 0.98-3.13)

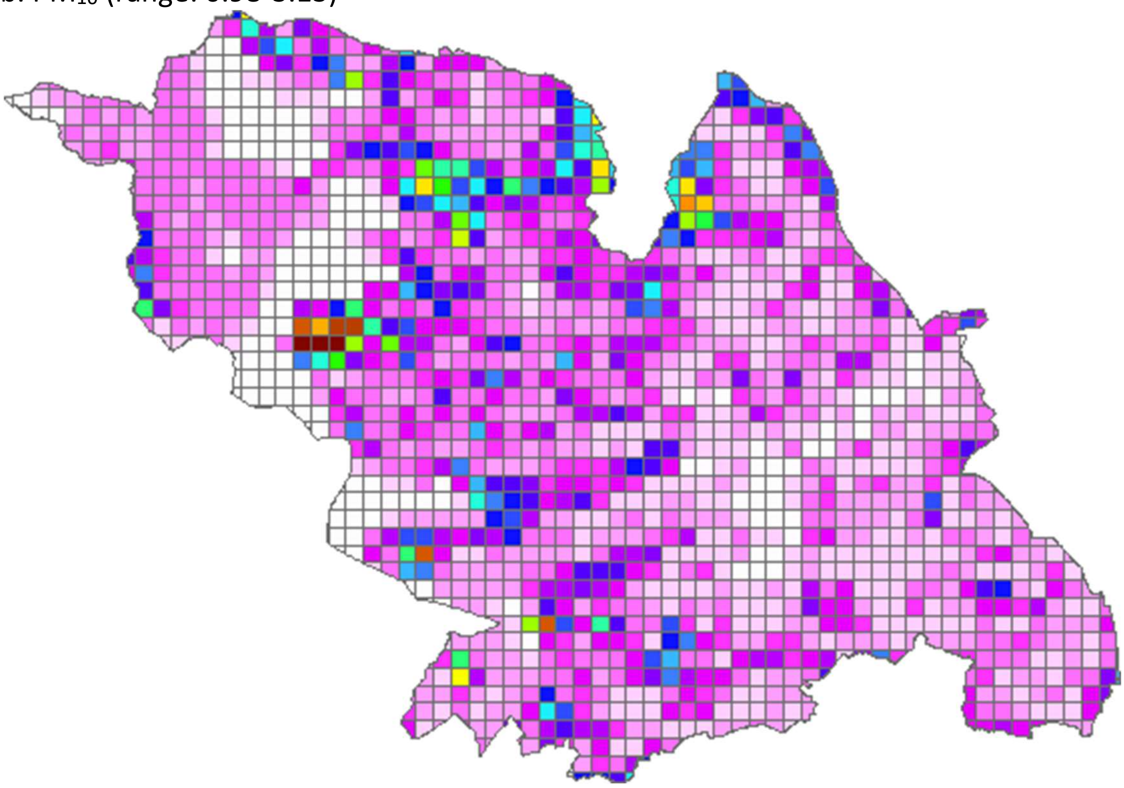


Figure 3.3. Rates of removal of air pollutants by deposition to ground surfaces, standardised by the rates that would occur if no greenspace land covers were present, for (a) NO_2 and (b) PM_{10}. Numbers in brackets are ranges of values; the maps both use the pseudo-continuous linear colour ramp shown in (a), where white and deep red represent the lowest and highest values in each graph respectively.

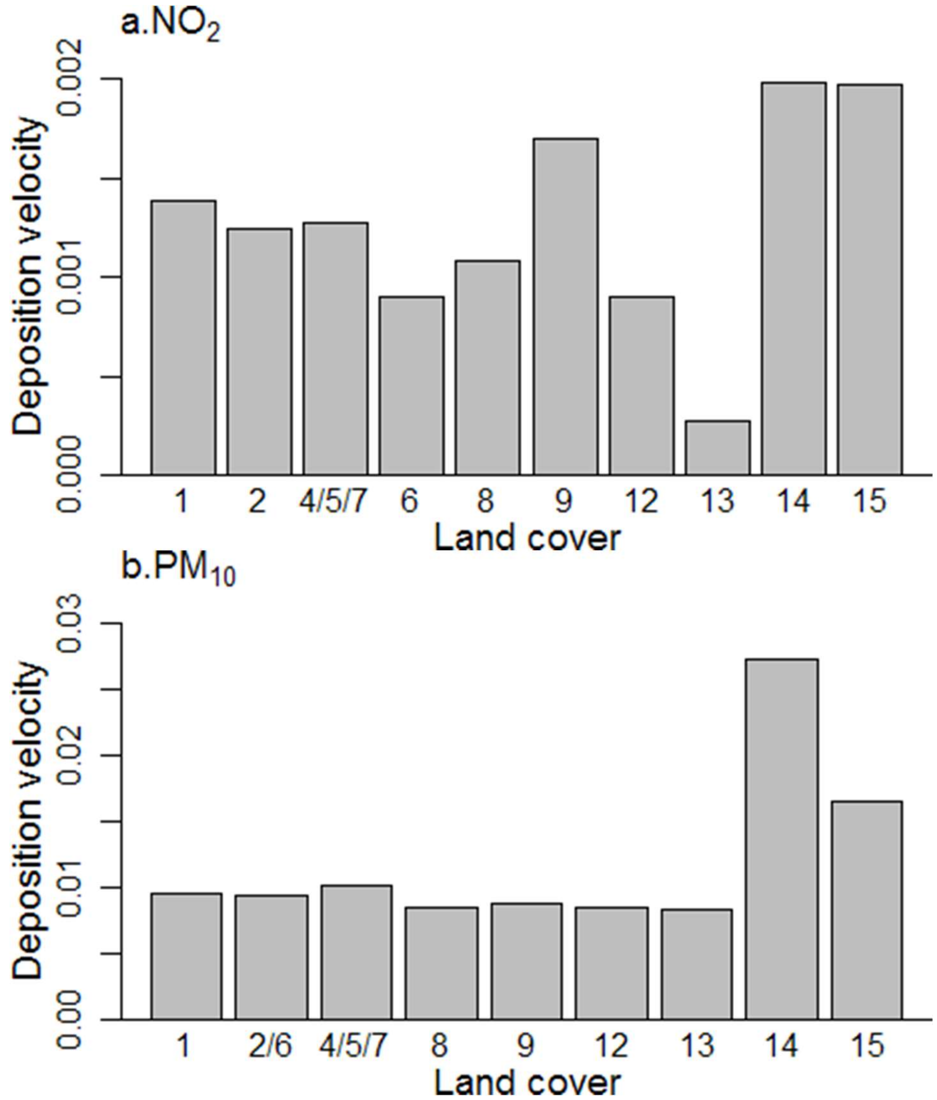


Figure 3.4. Annual average rates of deposition velocity (V_d) for different land cover types, for (a) NO_2 and (b) PM_{10}. Deposition velocity units are m s^{-1}. Land cover codes: 1 Arable; 2 Building; 4 Grassland (improved); 5 Grassland (rough); 6 Manmade surface; 7 Moorland (bog); 8 Moorland (heath); 9 Scrubland; 12 Unvegetated land; 13 Water; 14 Woodland (coniferous); 15 Woodland (non-coniferous and mixed). (Rates for other land cover classes calculated from combinations of classes shown here.)

notable for NO_2 deposition to water, but is also true of other land covers, highlighting that urban areas will not necessarily have lower deposition velocities than natural ecosystems. This explains why the moorland and urban areas have similar levels of pollutant removal.

3.3.2. Ecosystem service maps

The index used to quantify the ecosystem service, i.e. the average of the two standardised fluxes, is shown in Map 5 for the 500m^2 grid, Map 6 for Output Areas, and

Map 7 for Historic Environment Character areas (all maps found in Appendix D). Map 5 clearly shows the same patterns as Figure 3.3: the ecosystem service generated is generally greatest in the belt of mixed farmland and woodland in the centre of the study area, and lower in the moorland to the west and the urban area to the east; within the central belt there are however local areas of low deposition fluxes due to water bodies and areas of unvegetated land. Across most of the area (86% of polygons), the averaged pollutant deposition flux is only between 1.0-1.5 times the flux that would occur in the absence of natural ecosystems, and indeed the moorlands have similar values to the urban centre, suggesting that the urban matrix can provide a similar level of this ecosystem service to extensively managed ecosystems.

When ecosystem service generation is aggregated to Output Area boundaries (Map 6), much of the detail found in Map 5 is lost. However, because Output Areas have approximately equal populations Map 6 can be interpreted in terms of areas where people are located relative to ecosystem services. This interpretation would be more meaningful if Map 6 showed the supply, rather than the generation of an ecosystem service, but nevertheless provides additional insight.

Map 7 again has a different but informative interpretation, by facilitating comparison of ecosystem service generation between land uses. Although statistical analysis is not performed here, comparison with the land use map shown in Map 3 shows, for example, that in the urban centre the ecosystem service is provided mainly in residential areas and by parks, while industrial areas have lower values; and in contrast that farmland areas can provide different levels of the ecosystem service depending on the exact land cover matrix.

3.4. Discussion

Figure 3.4 shows that woodlands are by far the most effective deposition surfaces for NO_2 and especially PM_{10}. Even small amounts of woodland in the urban matrix can have a significant impact on air pollution: a previous modelling study in the West Midlands that was able to connect deposition to the effects on pollutant concentrations found that woodland reduces ambient PM_{10} concentrations by 4% despite covering less than 1% of the total area (McDonald et al. 2007). In Sheffield, where trees cover 14.7% of the land (according to the land cover map; the land use map classes 8.9% as woodland), this figure is likely greater.

Planting more trees is clearly an effective way to improve the ecosystem service of air pollution reduction, and this is indeed the plan of Sheffield City Council (Sheffield City Council, no date). A study in Santiago, Chile found that maintaining urban forests is a cost-effective air pollution control measure (Escobedo et al. 2008). In particular, pollution resistant trees planted in heavily polluted areas have the potential to make a large difference (Jim and Chen 2008). Coniferous trees are more effective than deciduous trees, as they have a year-round effect (Nowak et al. 2002a). However, the range of many coniferous trees is likely to decrease as a consequence of climate change (Thuiller et al. 2006), which may reduce the potential use of trees for air pollution mitigation.

Individual species also vary in effectiveness as deposition surfaces (Nowak et al. 2002a). Other factors to be taken into consideration in choice of tree species include lifespan and maintenance needs, and cultural and aesthetic preferences (Nowak et al. 2002a). Ideally, trees should be managed for good health, large size and large leaf surface area in order to maximise deposition effectiveness, by means such as timely pruning, watering and pest control (Jim and Chen 2008).

A final important consideration in choice of tree species is the emission of volatile organic compounds, or VOC (Nowak et al. 2002a). VOCs are produced by some tree species for attracting pollinators, controlling pests and/or protection against high temperatures (Nowak et al. 2002a). Although biogenic VOC emissions are relatively small compared to emissions from fuel combustion, they play a role in contributing to the formation of ozone, carbon monoxide and photochemical smog (Nowak et al. 2002a, Fenger 2009). These pollutants are associated with health issues, even though biogenic VOC in themselves are not harmful (Nowak et al. 2002a, Matthias et al. 2006, World Health Organization 2006). On the other hand, trees reduce air temperatures (see Chapter 4), which reduces the rate of formation of these pollutants (Nowak et al. 2002a, Matthias et al. 2006). To date, there do not appear to have been many studies of the overall consequences of trees for the formation of pollutants, although it is generally thought that the benefits of trees significantly outweigh the costs (Nowak et al. 2002a).

Planting trees is not the only option for improving levels of air pollution reduction. Water bodies, unvegetated land and, to a lesser extent, heath moorland, have lower rates of deposition than buildings and manmade surfaces (Figure 3.4); although, as is seen in the next chapters, these land covers do contribute significantly to other ecosystem services and so are not negative for ecosystem services overall. Other types of land cover, particularly arable, grass and scrub, are able to increase air pollution reduction relative to artificial surfaces (Figure 3.4). Some of these vegetation types are suitable for green roofs, which have been suggested as a way to increase levels of multiple ecosystem services in urban areas (Oberndorfer et al. 2007). However, building green roofs does not appear to be a cost-effective way of reducing air pollution, at least not without the consideration of other environmental benefits (Yang et al. 2008).

Although pollutant concentrations have clearly changed historically – with the service being more in demand at times when pollutant levels were highest – the ability of the land to act as a pollutant deposition surface and thus provide this ecosystem service has probably been relatively unaffected. This is because the proportional cover of woodland land use has remained nearly constant since 1700 (Figure 3.5). Built land uses have expanded considerably, especially in the past century, at the cost of enclosed and unenclosed land uses. However, with the exception of water bodies, which have only a small percentage cover anyway (Table 2.8), these land uses have deposition surfaces with more similar deposition velocities (Figure 3.4). Thus the overall capacity

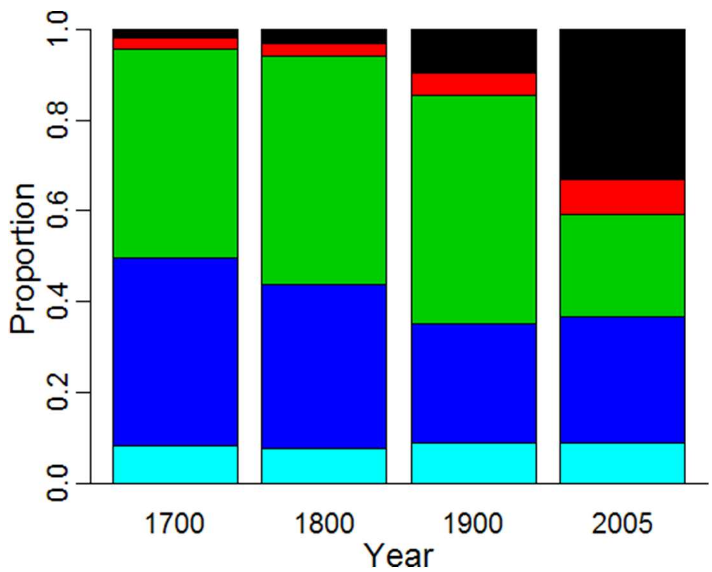


Figure 3.5. Proportion of Sheffield under different land uses from 1700 to 2005. Light blue: woodland. Dark blue: unenclosed land. Green: enclosed land. Red: other non-built land uses (extractive uses, horticulture, ornamental/parkland/recreation, water bodies). Black: built land uses (communications, commercial, industrial, institutional, residential). See Table 2.11 for land cover composition of land use types. Source: Historic Environment Character dataset.

of the land in Sheffield to remove pollutants from the air has probably remained relatively unchanged over the past 300 years.

The implications of the output of this model are discussed further in Chapter 9, alongside the results of the other ecosystem service models.

4. Mitigation of the heat island effect

4.1. Introduction

It is well documented that the climate of urban areas differs from that of the surrounding countryside, with the effect on temperature being especially prominent (Oke 1987, Pickett et al. 2001, Jenerette et al. 2007). This "heat island effect", whereby urban temperatures can be several degrees higher than the surrounding countryside, is a consequence of an altered energy exchange regime; especially absorption of heat from sunlight by the built environment that is later reradiated as thermal radiation, and reduced cooling from evapotranspiration due to diminished vegetative cover (Oke 1987, Frumkin 2002, Gartland 2008). Thus the areas within a city that have the greatest proportional cover by artificial surfaces – typically the urban core – tend also to have the highest temperatures (Pickett et al. 2001, Jenerette et al. 2007).

Increased temperatures are of concern because of the risks posed to human health (Frumkin 2002, Gartland 2008). Mild health consequences of heat exposure include cramps, heat syncope (fainting), oedema, and tetany (involuntary muscle contraction induced in this case by hyperventilation); whereas heat exhaustion and especially heat stroke are more serious consequences and have a high fatality rate (Frumkin 2002, Gartland 2008). Heat waves exacerbate the effects of the urban heat island, and are associated with large increases in death rates (Gartland 2008). The 2003 European summer heat wave is estimated to have caused 52,000 deaths, for example; and the extremity and frequency of high temperatures will increase as the climate warms (Gill 2006, Larsen 2006). High temperatures also increase the use of air conditioning systems and electric fans, with an associated fuel cost (Gartland 2008).

An indirect health issue associated with high temperatures in urban areas that have significant air pollution problems is increased production of ozone (Gartland 2008). Ozone is a pollutant that is formed by chemical reaction between nitrogen oxides (NO_x), emitted by fuel combustion, and volatile organic compounds, which have many sources, including incomplete combustion, industrial processes and biogenesis by certain types of trees and vegetation (Gartland 2008, Fenger 2009). Ozone causes health problems by inflaming the airways and reducing lung function (Frumkin 2002, World Health Organization 2006). Hospital admissions and mortality rates have been linked to ambient ozone concentrations, especially amongst asthmatics and other groups susceptible to respiratory problems (Frumkin 2002, World Health Organization 2006, Fenger 2009).

The question of heat island effects on urban ecology does not appear to have been extensively studied, although there is evidence that growing seasons start earlier and finish later in urban areas than in the surrounding rural regions (Roetzer et al. 2000, White et al. 2002). Research on warming more generally suggests that such phenological changes have the potential to disrupt ecological interactions, such as plant-pollinator relationships and availability of food for nesting birds (Visser et al. 1998, Memmott et al. 2007). Global warming research also indicates that species with low temperature tolerance may be unable to persist in cities with a strong heat island effect, while new species – including some pest species – with a preference for higher temperatures may become more common (Bisgrove and Hadley 2002, Wilby and Perry 2006).

A further likely effect of heat islands, again based on global warming research, is to increase plant water stress in the summer months (Gates 2002). Increased water stress would reduce evapotranspiration and its coincident cooling effect and, eventually, may kill vegetation through drought (Gill 2006). This would reduce the ability of the ecosystem to counteract the heat island effect, creating a positive feedback loop.

Urban temperatures are therefore an issue of increasing environmental as well as socioeconomic concern. This chapter presents the model used to map the ecosystem service of heat island mitigation.

4.1.1. Heat island causation and mitigation

Radiation fluxes in urban areas can be represented by the following equation (after Tso et al. 1991, Offerle et al. 2005):

$$Q^* + Q_F = Q_H + Q_E + \Delta Q_S + S \tag{2}$$

Where Q^* is the net all-wave radiation, Q_F is the anthropogenic heat flux (i.e. waste heat from fuel use), Q_H is the sensible (convective energy) heat flux, Q_E is the latent heat of evaporation (energy storage in water vapour), ΔQ_S is heat storage in the built environment, and S represents conduction to the soil and other sources and sinks of energy (such as rainwater, which removes heat from the urban area via sewers).

The urban environment contributes to a number of changes in the relative importance of these fluxes (Oke 1987, Gartland 2008). The anthropogenic heat flux and heat storage in the built environment are only important in built-up areas (Oke 1987, Tso et al. 1991). Reduction of the latent heat flux due to the abundance of sealed surfaces and relative lack of vegetative cover is also important (Oke 1987, Gartland 2008). Other contributing factors are that urban surfaces reduce reflection of short-wave radiation; air pollution and increased cloud formation increase long-wave radiation re-emission to the ground; and the canyon geometry of cities reduces long-wave radiative loss and turbulent heat transport, due to reduced 'sky view' and wind speeds respectively (Oke 1987, Gartland 2008). The net effects of these factors are summarised in Figure 4.1.

The heat island effect has not been measured directly in Sheffield. An array of temperature loggers located around the urbanised region reveal spatial differences of several degrees Celsius, although no loggers were placed in the surrounding rural regions for comparison (Fuller et al. 2010). Mean temperature surfaces for summer and winter, created from these loggers, are shown in Figure 4.2. The heat island effect occurs in settlements of only one thousand inhabitants, with the intensity of the effect increasing with city size: a city such as Sheffield, with a population of just over half a million, might be expected to have a thermal modification of 7°C at certain times of day

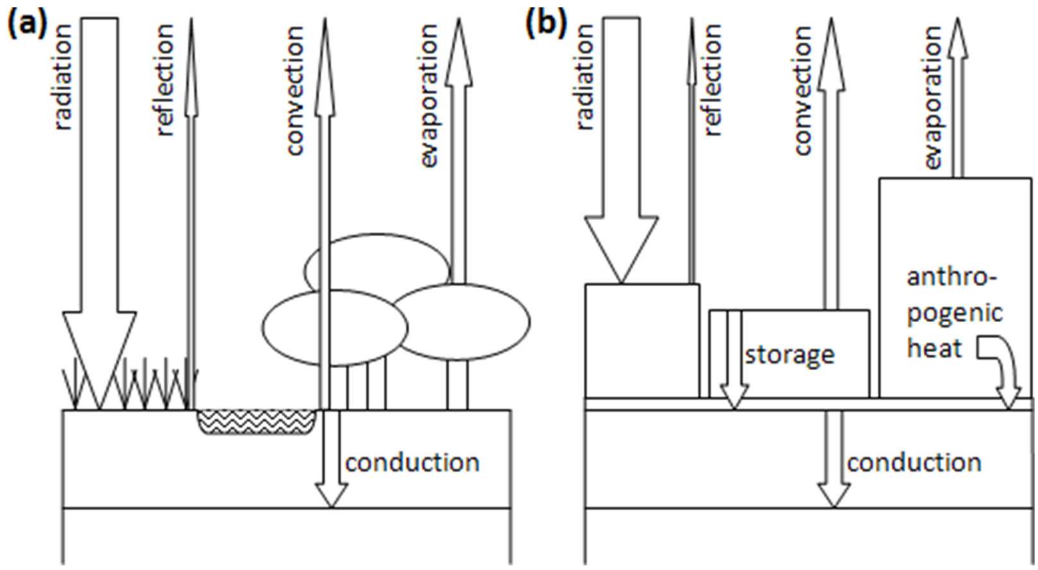


Figure 4.1. The relative importance of energy transfers in (a) rural areas and (b) urban areas, represented by width of arrows. The urban heat island effect occurs because of a reduction in reflection and evaporation, the production of heat from fuel, and storage of heat that is re-radiated at night. Adapted from Whitford et al. (2001).

compared to the surrounding countryside (Oke 1987). Although the intensity of the heat island depends upon meteorological, urban morphological and topographical characteristics, the effect typically increases towards the urban centre (Oke 1987, Pickett et al. 2001, Arnfield 2003). Land use also has an effect on a smaller scale, with vegetated areas and water being cooler, and densely built environments being hotter (Oke 1987, Jenerette et al. 2007). These patterns are reflected in Figure 4.2, which shows the highest temperatures in the city centre as well as other hotspots further out.

There is a diurnal cycle to heat island intensity that results from the insulating effect of heat storage in the built environment: the urban environment is both slower to warm up in the morning, and slower to cool at night (Oke 1987). The degree of thermal modification is typically greatest in the evening, shortly after sunset, although the temperature is highest in the afternoon (Oke 1987, Pickett et al. 2001).

There are several practical measures that can be implemented to reduce the heat island effect. Pavements and other surfaces can be constructed from alternative materials that increase reflection of incoming radiation, store less heat and/or are permeable to facilitate cooling through evaporation (Cambridge Systematics 2005, Gartland 2008). Roofs can similarly be constructed to increase reflectance, or can be built as green or living roofs, i.e. with a porous substrate and vegetation, which increase the latent heat flux through evapotranspiration (Oberndorfer et al. 2007, Gartland 2008).

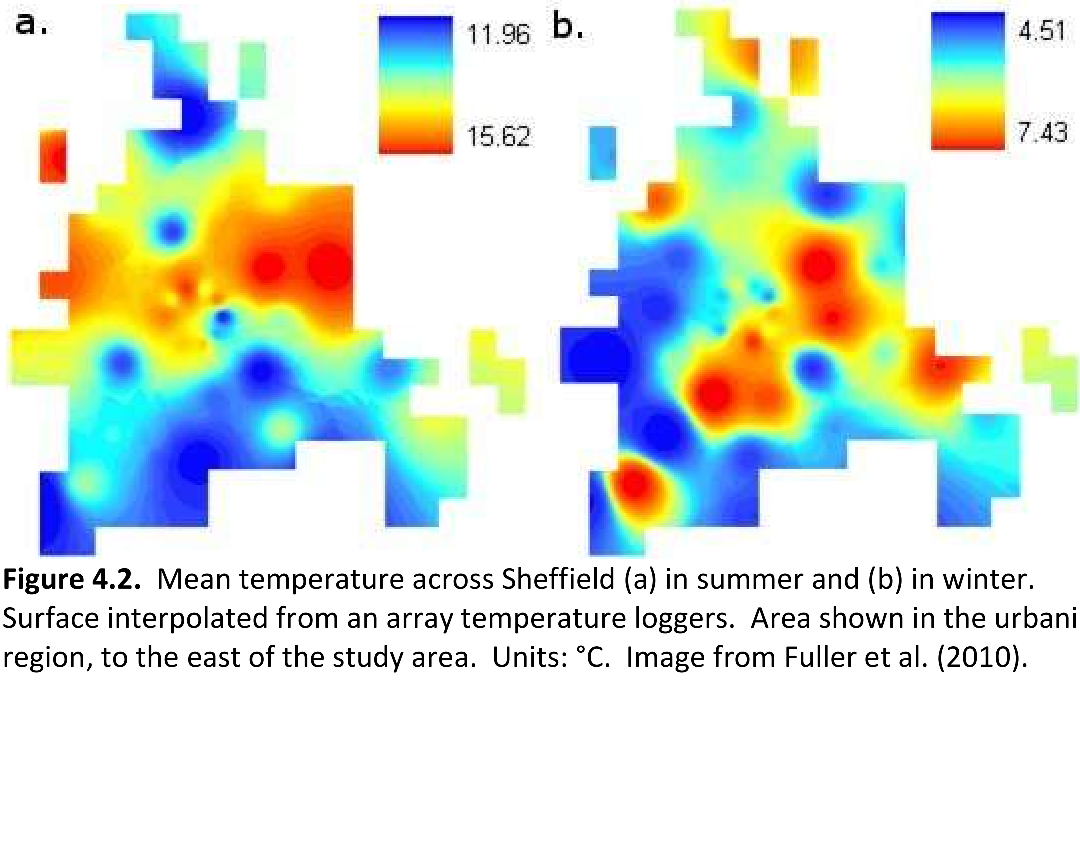


Figure 4.2. Mean temperature across Sheffield (a) in summer and (b) in winter. Surface interpolated from an array temperature loggers. Area shown in the urbanised region, to the east of the study area. Units: °C. Image from Fuller et al. (2010).

Alternatively, trees and vegetation can be used to provide cooling through more natural energy budgets (Figure 4.1) – that is, more evapotranspiration and convective cooling, increased reflection of incoming radiation, less storage of heat, and also shading of the ground (Oke 1987, Gartland 2008, Nowak et al. 2008). Observational studies have confirmed the contribution of urban greenspaces to heat island mitigation (Jenerette et al. 2007). As well as large areas such as parks, there is also an important role for small patches of greenspace within the matrix of the built environment, such as verges and gardens (Gill 2006). It is this mitigation effect by small and large patches of urban greenspace that is relevant for the current model.

4.1.2. Generation of the ecosystem service

All permeable surfaces and vegetation can contribute to heat island mitigation by evapotranspiration. However, different types of land cover mitigate the effect by varying amounts, with woodland being the most effective (Tso et al. 1991, Gill 2006). The thermal properties of the particular soil substrate also play a role in determining surface temperature (Tso et al. 1991, Gill 2006).

The production of the ecosystem service of heat island mitigation is quantified here as the reduction in surface temperature that results from the presence of greenspace in the land cover matrix (see Section 2.1.1). Surface temperatures are modelled both for the actual land cover matrix, and for a hypothetical land cover of 28% buildings and 72% artificial surfaces (i.e. an extrapolation of the actual ratio of buildings:artificial surfaces in the study area). The reduction in surface temperature is then the difference between these temperatures.

The reduction in surface temperatures provides benefits to humans at the site of reduction. This means that the reduction in temperature can be considered to be both the production and the supply of the ecosystem service, provided there are people at that site (see Section 1.2.1). The value of the ecosystem service of heat island mitigation arises from the reduction of human health impacts and the economic costs of air conditioning and other climate control measures. A conceptual diagram of the production of these values is shown in Figure 4.3.

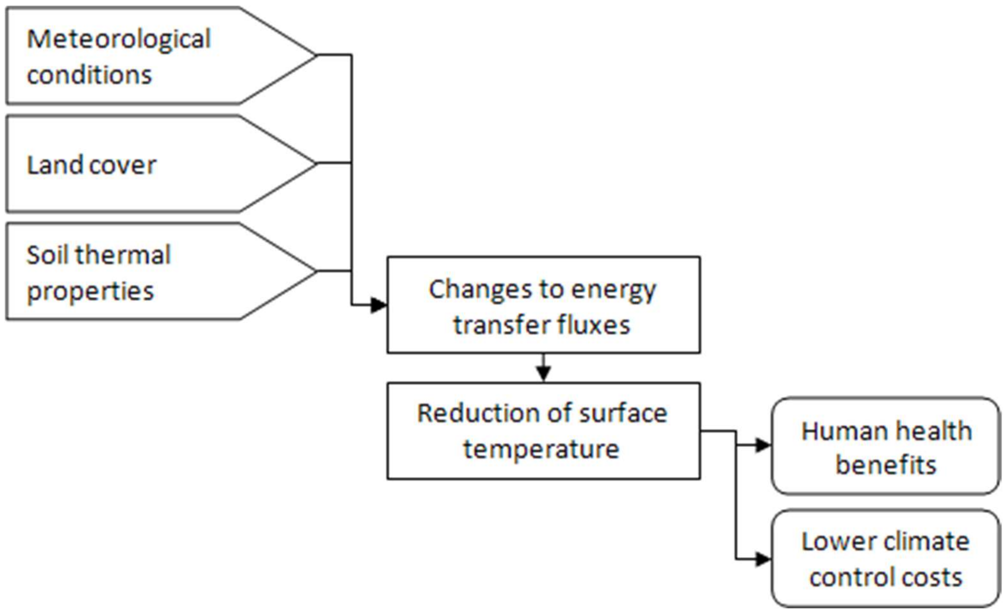


Figure 4.3. Conceptual representation of the heat island mitigation model. Input conditions are represented by pentagrams; ecosystem processes by rectangles; and socioeconomic benefits by ellipses.

4.2. Model overview

The model implemented in this project is the urban climate model developed by Tso et al. (1991) for modelling surface temperature in Kuala Lumpur, and later used by Whitford et al. (2001), Gill (2006) and Tratalos et al. (2007) for various towns and cities in the UK. Although the above discussion related to air temperature, it is actually thesurface temperature that is modelled here. The two temperatures are correlated both temporally and spatially, although with some differences: surface temperature is more strongly affected by shading than air temperature, while air temperature is dependent on the temperatures of the surrounding surfaces and also on advective currents (Arnfield 2003). Surface temperature is therefore easier to model because wind effects do not need to be taken into account (Whitford et al. 2001). Furthermore, surface temperatures determine the mean radiant temperature, which is a dominant factor contributing to human comfort levels (Matzarakis et al. 1999, Gill 2006); indeed, the mean radiant temperature is quite closely correlated with an index of the total effect of meteorological conditions on the human energy balance, at least in unshaded areas (Matzarakis et al. 1999). This model is also suited to estimating temperatures from a known surface composition, over scales larger than the individual ‘urban canyons’ that are the focus of other models (Whitford et al. 2001).

Full details of the model can be found in Appendix A.2; this section provides an overview. The model estimates the maximum daytime surface temperature for a given set of meteorological parameters. In this case, the meteorological parameters used are customised to represent an extremely hot summer day in Sheffield. Spatially variable model inputs include land cover and soil properties. There are also a number of parameters for constants and process rates. Details of the sources of input data are given in Appendix A.2.2.

The model is based on a traditional energy exchange equation, with an additional term to represent heat storage in buildings. This equation relates the heat storage in buildings, M, to the net radiation flux, R; the sensible heat flux to the air, H; the latent heat of water, L; the evaporation rate, E; and the heat flux to the soil substrate, G (Tso et al. 1991):

$$M = R - H - LE - G \tag{3}$$

These fluxes, and the vertical layers across which the energy exchanges are modelled, are illustrated in Figure 4.4. The model uses a pair of simultaneous, time dependent, linear, first-order differential equations to estimate the temperature at ground level and in the soil layer during the course of 24 hours (Gill 2006). The full mathematical derivation is given in Appendix A.2.1.

The model was used to estimate the temperature for each cell in a 500m^2 grid superimposed over the study area (Figure 2.3). The maximum temperature during the modelled 24 hour period was recorded, and from this value was subtracted the estimated temperature for a hypothetical land cover of 28% buildings and 72% manmade surfaces. The result of this subtraction quantifies the production of the ecosystem service of heat island mitigation. Area-weighted means of these 500m squares were used to produce maps at Output Area and Historic Environment Character area scales (Section 2.3).

4.2.1. Sensitivity tests

There are a large number of variables as input to this model, and it is not immediately clear which variables have the greatest impact on the results. Therefore sensitivity testing was performed in order to identify if there were any variables with a high degree of uncertainty that have a large influence over the model results, and thus

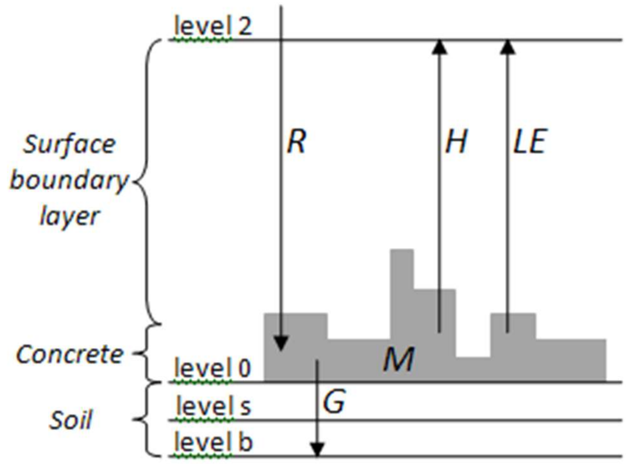


Figure 4.4. Framework of the urban energy exchange model, showing modelled energy balance components and the vertical layers across which energy exchange occur. Level 2 is the top of the stable boundary layer; level 0 is ground level; and levels s and b are the middle and bottom of the soil layer respectively. *R* is the net radiation flux; *H* the sensible heat flux to the air; *LE* the latent heat loss through evaporation; *M* the heat storage in buildings; and *G* the heat flux to the soil substrate. Adapted from Tso et al. (1991).

might introduce unacceptable levels of unreliability. Further details of the sensitivity tests are given in Appendix A.2.3.2.

The sensitivity tests found that model results were not highly dependent on any single parameter. Furthermore, the parameter to which the model is most sensitive is the Von Kármán constant, of which the value is known with confidence (Kantha and Clayson 2000). Uncertainty associated with any single parameter in this model should not therefore cause unacceptable uncertainty in the model results.

4.2.2. Model limitations

Gill (2006) lists the key assumptions of this model. These are listed below, with appropriate modifications for the present model implementation, i.e. using land cover- and soil-specific parameters.

1. Meteorological properties remain constant horizontally.
2. Soil properties remain horizontally constant within NATMAP soil polygons and between polygons of the same soil type.

3. The eddy diffusivities for heat and water vapour are given by the near neutral value for momentum (Eqn. (70), Appendix A.2.1).
4. The turbulent fluxes of heat and water vapour are constant over the planetary boundary layer.
5. The temperature, wind speed and specific humidity are assumed to be constant at the height of the planetary boundary layer.
6. The study area is assumed to have a unique roughness length (a parameter related to the height of the structures interfering with turbulent processes at the land surface).
7. Anthropogenic heat sources have been neglected.

Discussion of assumptions 3 to 5 is beyond the scope of this study. However, assumption 1 is especially relevant: it is plain that meteorological conditions will not be constant across the study area, which covers both upland moorland and lower altitude, densely urbanised areas. The model is slightly sensitive to the values of several meteorological parameters (Appendix A.2.3.2). Of these, most are dependent on the reference temperature, which itself is a scenario value. The others are wind speed and peak insolation. Temperature and wind speed in particular would be expected to vary across Sheffield. Although the effects of variation in these parameters individually are quite small, the combined effects may be significant. The model output should therefore be regarded as showing the heat island response to given local meteorological conditions, since the actual variation in meteorological conditions that would occur across Sheffield on any given day cannot be taken into account.

The soil properties (assumption 2) are also unlikely to be constant within NATMAP polygons given the natural variability of soils and the range of land covers/uses, and thus soil management regimes, occurring over each soil type. The model is insensitive to the precise values of soil parameters, however, so violation of this assumption should not be a problem (Appendix A.2.3.2).

The assumption of a constant roughness length over the urban area is also not valid, again because of the range of different land covers and uses that the study area encompasses. Indeed, the use of a single roughness length is in direct contrast to the range of values used in the air pollution reduction model (Appendix A.1.2.3); but this approach was necessary for the present model implementation (see Appendix A.2.2.1

and Section A.2.3). The heat island mitigation model is relatively insensitive to small changes of the roughness length value, so this should not pose too great a problem.

The omission of anthropogenic heat sources may be significant, as research suggests that the anthropogenic heat flux can be large (Arnfield 2003, Offerle et al. 2005), and it would also be expected to vary across the study area. It is unclear how violation of this assumption might impact on the model results, and the issue has not been addressed by previous authors who have used this model (e.g. Whitford et al. 2001, Pauleit et al. 2005, Gill et al. 2007, Tratalos et al. 2007). However, simulation studies have indicated that, during the summer in temperate climates, the influence of anthropogenic heat on the energy balance is quite small even in large cities (Ichinose et al. 1999). When models do include the anthropogenic heat flux in urban climate models, it is usually included as a simple additive component to the net radiation (Arnfield 2003). It therefore increases the amount of energy that must be dissipated and increases the surface temperature. It can be reasoned that areas with the greatest anthropogenic heat flux are likely to be those already suffering from the greatest heat island, i.e. those areas with little vegetation and a high density of building mass for heat storage. The effect of anthropogenic heat is probably therefore fairly linear, meaning that it should not invalidate the model results as long as it is the relative, rather than the absolute values of the output that are emphasised.

4.3. Results

4.3.1. Model output

Figure 4.5a shows the output of the model for the actual land cover composition of Sheffield, i.e. the estimated maximum surface temperature reached in each grid square on a hot summer's day. The grid squares reaching the highest temperature are generally in the city centre, with the exception of one square in the centre of the study area that is composed almost entirely of unvegetated land. Unvegetated land reaches the highest temperatures of any land cover due to a lack of evapotranspirative cooling, combined with less heat storage capacity than built environments. There is a fairly steady decrease in temperature with increasing distance from the urban centre, with localised variation resulting from roads and suburban and rural built-up areas, and parks and gardens introducing cooler spots within developed areas.

This pattern of decreasing temperature is in spite of the temperature variations that would occur in the absence of natural land covers, shown in Figure 4.5b: in this situation, the surface temperature would be greatest in the west of the study area, due to variation in soil properties. It should again be emphasised that these are the temperatures that would be seen if exactly the same meteorological conditions were applied to each 500m grid square: the lowest temperatures would probably be seen in the high altitudes in the west of the study area if the model input data reflected realistic meteorological variation.

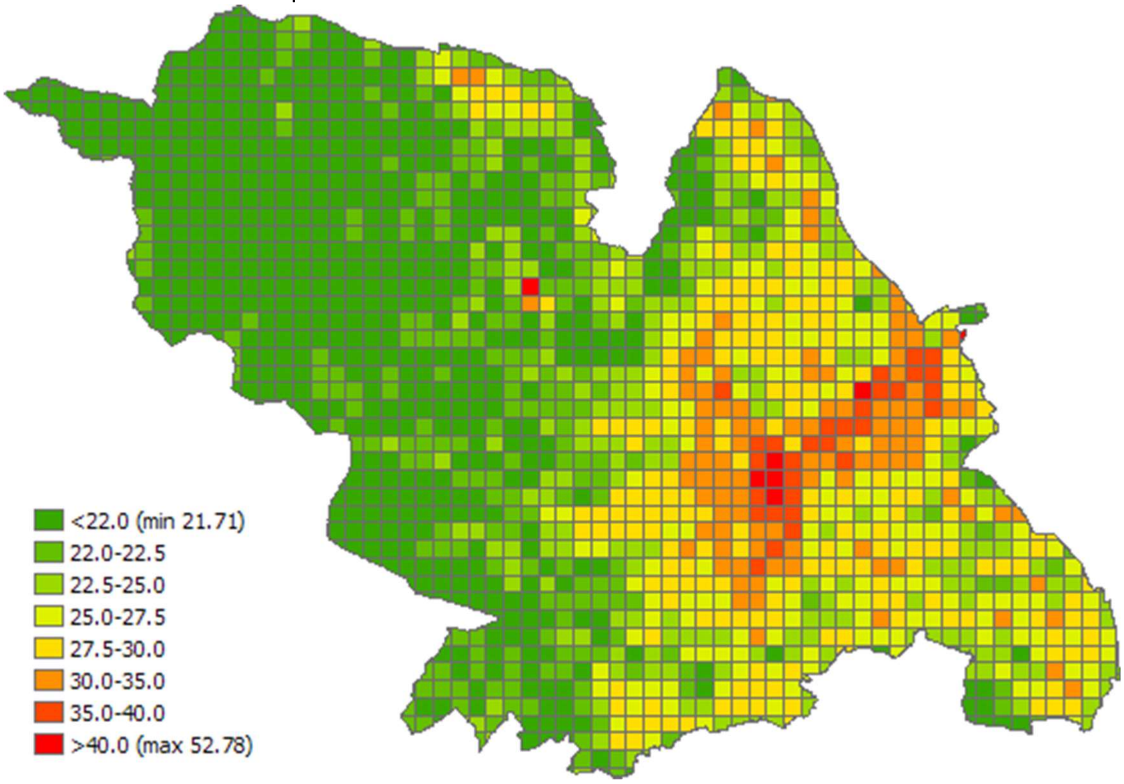
4.3.2. Ecosystem service maps

The contrasting patterns illustrated in Figure 4.5 mean that the actual ecosystem service, i.e. the difference between the actual maximum surface temperature and that which would be reached in the absence of the ecosystems, is very large in the western part of Sheffield: across much of the moorland, the modelled reduction in surface temperature is greater than 25°C (Map 8; all maps shown in Appendix D). The reduction is much smaller in the urbanised area, especially in the commercial and industrial core. Typical reductions in the mainly residential areas are 15-25°C, with greater reductions in more distant suburban regions. However, the square that mostly consists of unvegetated land actually has a higher estimated temperature than would occur if only artificial surfaces were present.

Map 9 and Map 10 show the reduction in surface temperature for Output Areas and the Historic Environment Character polygons respectively. These maps are less informative than those for the air pollution reduction service (Section 3.3.2) because the ecosystem service values could not be independently calculated but were rescaled from the 500m grid. Nevertheless, these maps remain valuable for showing alternative population-based and land use-based perspectives.

Map 9 suggests an interesting relationship between heat island mitigation and resident population density. Areas of very low population density in the moorlands have consistently high levels of ecosystem service production. Neighbourhoods with the highest population density, however, have better levels of ecosystem service production than those with lower density in the urban core. This is an artefact of the land cover composition of different types of land use: residential areas, which have the highest population density, also tend to have higher levels of greenspace provision than

a. Maximum surface temperature



b. Maximum temperature without natural surfaces

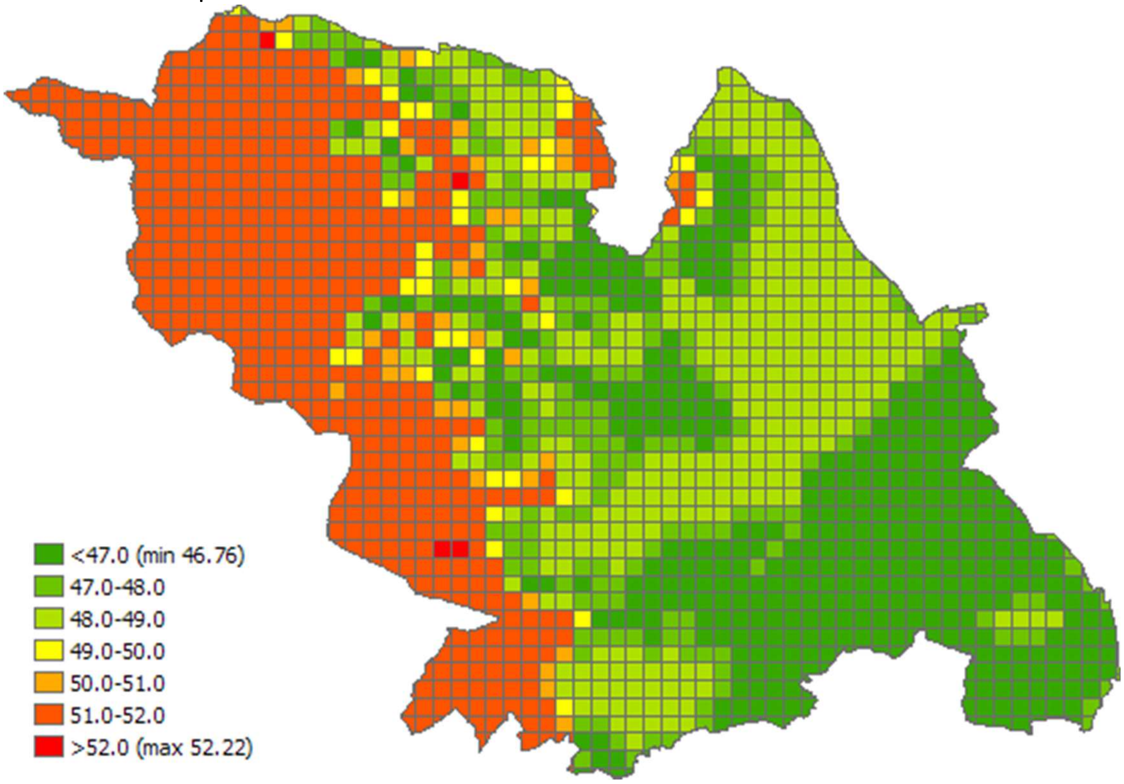


Figure 4.5. Estimated maximum surface temperature (a) and estimated maximum surface temperature if only manmade surfaces and buildings were present (b). Note different legends. Units: °C.

the industrial and commercial units common in the very centre of the city (Map 2, Map 3, Map 10).

Comparison of Map 10 with the land use map (Map 3) indicates that, within the urbanised area, land uses such as parks and unenclosed land tend to provide greater ecosystem service levels than residential and institutional areas. There is, however, also some variation between different residential and institutional polygons, reflecting variation in land cover composition.

4.4. Discussion

The supply of the ecosystem service of heat island mitigation is very much dependent on individuals being present at sites with low surface temperatures. This presents something of a paradox for urban planning, since it is the presence of large numbers of people that necessitates the replacement of vegetation with the buildings and manmade surfaces that cause the heat island effect in the first place (Alexandri and Jones 2008). This can clearly be seen by comparing Map 10 to the land use map in Map 3: land use types where population densities are relatively high (residential, commercial, industrial) typically have high surface temperatures.

One obvious solution to this problem is to design cities with lower population density in order to increase the proportion of land covered by greenspace. Studies of cities in England have, however, suggested that lower population densities are associated with greater social inequity and segregation, poorer public transport services and access to facilities, greater energy consumption and dependence on automobiles, and greater emissions of pollutants (Burton 2000, Newton 2000, Simmonds and Coombe 2000). It is therefore important to focus on optimising the amount of greenspace within the urban matrix, in preference to reducing population density.

One approach to this would be to convert all non-essential artificial surfaces to vegetated, unsealed land covers. Where finding such space is an issue, constructing green roofs can also help to mitigate local temperatures. Green roofs are building-top constructions including vegetation in a growing medium. Green roofs may be intensive, with a substantial soil layer and potentially large plants; these types of roofs are typically built into the building design due to support requirements (Oberndorfer et al. 2007, Santamouris 2009). Extensive greenroofs, which can be retrofitted, typically have a small layer of lightweight growing medium and small plants such as grasses and *Sedum* (stonecrop) species (Oberndorfer et al. 2007, Santamouris 2009).

Green roofs are widely recognised to insulate buildings, making it easier to keep the internal air temperature within a comfortable range during hot or cold weather (Del Barrio 1998, Oberndorfer et al. 2007, Santamouris 2009). They also cool the air above the roof: an empirical study in Singapore found that the air temperature 30cm above a green roof was up to 4.2°C lower than that above a conventional roof, with the cooling effect decreasing with height above the roof but still present to at least 1m (Wong et al. 2003). The air cooling effect is a result chiefly of less convective heat exchange to the air from the cooler roof (Wong et al. 2003, Alexandri and Jones 2008). As air turbulence moves air masses from above roofs into urban canyons (i.e. streets), air temperatures can be cooler at ground level (Alexandri and Jones 2008). Modelling exercises indicate that if all roofs were greened, the air temperature across the region could be significantly reduced (Alexandri and Jones 2008). For example, if half of the buildings in the city of Toronto were fitted with green roofs, the air temperatures across the whole geographic region would be reduced by 1°C, or more if the green roofs were irrigated (Bass et al. 2003).

In contrast to the air pollution reduction model, the ability of the land cover in Sheffield to mitigate the heat island effect has probably been drastically reduced in the past few centuries. This is because a far larger proportion of land is under built land uses today than it was historically (Figure 3.5). Since the value of the ecosystem service is linked to the cause of the heat island – i.e. greater population size necessitating replacement of natural land covers with artificial surfaces – the value to the residents of Sheffield has presumably increased in line with the land cover change.

Sheffield's population is set to continue to increase (Sheffield City Council 2009b), and the intensity of the urban heat island is typically seen to increase with population and city size (Oke 1987, Arnfield 2003). Thus the value of this ecosystem service is likely to continue to increase. Moreover, climate change is predicted to cause an average summer temperature increase of 0.5-1.3°C by the 2020s, rising to 2.3-4.6°C by the 2080s (UKCIP 2002). There is also likely to be an increase in the frequency of very hot days (UKCIP 2002), when mitigation of the heat island effect is especially important.

A study in Greater Manchester found that increasing greenspace coverage by 10% in high density residential and town centre areas would keep maximum surface temperatures at the same level seen in 1961-1990 until 2080; green roofing was also suggested as an alternative mitigation measure (Gill 2006). Increasing greenspace cover therefore has the potential to play a key role in mitigating temperature increases due to both increases in city size and global warming.

5. Reduction of storm water runoff

5.1. Introduction

One of three things will happen to water falling as precipitation: it may be intercepted by vegetation or another structure that can hold water; it may be abstracted from the ground surface by infiltration to a permeable substrate, especially soil; or it may remain in small depressions on the ground surface (USDA-NRCS 1986, Arnold and Gibbons 1996). Abstracted water is likely to become part of a subsurface flow, whereas water remaining on the surface will form a surface flow if there is a topographical gradient.

Urbanisation inherently involves the replacement of natural land covers with developed and often impermeable land covers such as buildings and roads (USDA-NRCS 1986, Kline 2006). Two consequences of this are reductions in the amount of precipitation that is intercepted and later evapotranspired by vegetation, and in the amount that infiltrates and is stored in the soil, with the result that the surface flow is greatly increased (Whitford et al. 2001, Booth et al. 2006). This phenomenon is known as storm runoff, and is illustrated for rural and urban environments in Figure 5.1. This chapter describes the model used to map reduction of storm water runoff and discusses the spatial patterns of runoff and runoff reduction by greenspaces.

5.1.1. Consequences of storm runoff

High levels of storm runoff result in damage to human property, human health risks, and negative effects on ecosystems; these problems are reviewed here. The most obvious consequence is flooding. There are two types of flooding that can be caused by heavy rainfall: fluvial flooding, which occurs when rivers overtop their banks; and surface water flooding (Wheater 2006). Surface water flooding is a problem particularly in urban areas because replacement of natural land covers with impervious manmade surfaces increases both the amount of surface flow and the peak discharge rates following storm events (Paul and Meyer 2001, Booth et al. 2006, Wheater 2006); see Figure 5.1. The surface water flows down-catchment, flooding any topographical

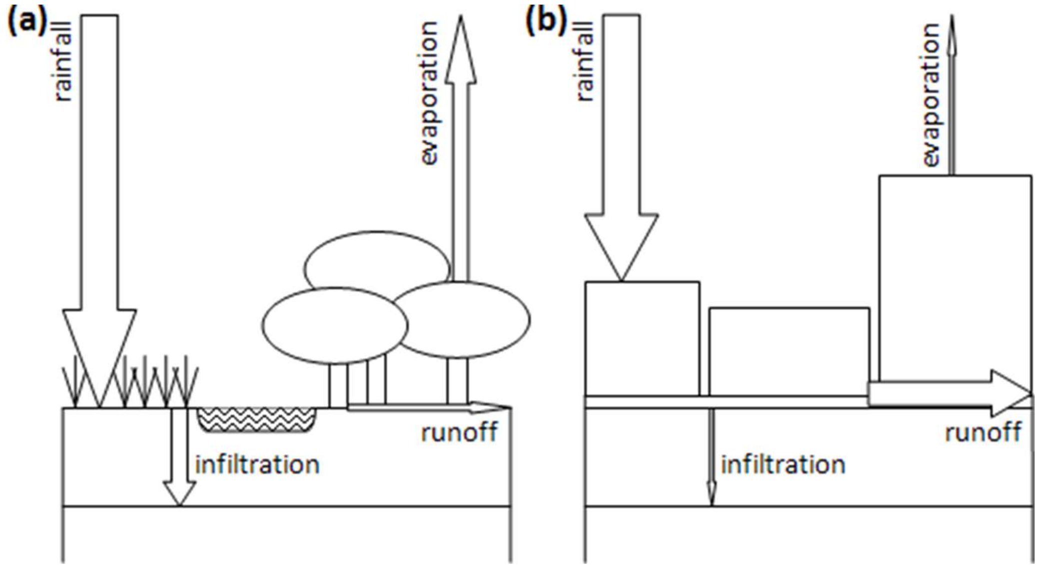


Figure 5.1. The relative importance of hydrological processes following rainfall in (a) rural areas and (b) urban areas, represented by arrow widths. Increased runoff in urban areas results from less evapotranspiring land cover and introduction of impermeable ground covers. Runoff flows down-catchment towards a river. Adapted from Whitford et al. (2001).

depressions, such as subways that are not adequately drained, before continuing until it flows into a river (Wheater 2006). Increased peak discharge rates make it more likely that the capacity of urban drainage systems – which can themselves be expensive to maintain – will be increased (Booth et al. 2006). Both types of flooding have occurred in Sheffield in recent years (BBC News 2007, Crabtree 2009).

The economic costs of floods are large, with extensive fluvial flooding in Sheffield in June 2007 causing an estimated £30m of damage, and the average home insurance claim following UK-wide flooding at this time being close to £52,000 (BBC News 2007, Guardian.co.uk 2008). Just as important are the human health costs, both physical and psychological, resulting from flooding of either type (Ohl and Tapsell 2000). Physical health risks are posed directly by rising waters (for example due to drowning, hypothermia, electrocution) as well as by contaminants within the water, which may cause allergic reactions or harbour contagious disease (Howard et al. 1996, Centers for Disease Control and Prevention 2000, Ohl and Tapsell 2000). There is also strong evidence for longer-term, psychological disturbance caused by the trauma of flooding and its consequences (Bennet 1970, Abrahams et al. 1976, Ohl and Tapsell 2000).

Contaminated storm water runoff presents a further problem if it comes into contact with drinking water supplies. Runoff may become contaminated by harmful levels of dissolved or suspended pollutants, including high loads of particulate matter, heavy metals, fertilisers and carcinogenic polyaromatic hydrocarbons (Tsihrintzis and Hamid 1997, Paul and Meyer 2001, Gaffield et al. 2003, Sansalone 2003, Booth et al. 2006). Runoff can also harbour outbreak-causing levels of pathogens including faecal coliforms, *Giardia* and *Cryptosporidium* (Gaffield et al. 2003). This has obvious consequences for drinking water quality (Ohl and Tapsell 2000, Gaffield et al. 2003).

Storm water runoff also has ecological consequences, regardless of whether or not flooding occurs. The changes to a catchment's hydrological regime associated with urbanisation typically cause a loss of biodiversity and compromise ecological functioning (Arnold and Gibbons 1996, Paul and Meyer 2001, Chadwick et al. 2006, Paul et al. 2006). Contaminants can harm biodiversity as well as humans, and the increased frequency and size of storm discharges can disturb biotic communities (Paul and Meyer 2001, Walsh et al. 2005, Booth et al. 2006).

The ecosystem services that are typically degraded as a consequence of these ecological effects include the capacities to attenuate pollutants, retain excess nutrients and prevent soil erosion; and nutrient cycling (Paul and Meyer 2001, Chadwick et al. 2006, Paul et al. 2006, Hogan and Walbridge 2007). In addition, water quality degradation may reduce the value of water bodies for recreational ecosystem services: for example, canoeing and fishing are common on Sheffield's rivers, but are likely to be less attractive pastimes at degraded sites, as Johnstone & Markandya (2006) found to be the case for angling.

5.1.2. Generation of the ecosystem service

Permeable surfaces allow abstraction of precipitation into soils, and vegetation also intercepts some rainfall; hence the presence of both permeable and vegetated land covers in the urban matrix increase the capacity of the land to reduce surface runoff, i.e. the potential for production of the ecosystem service. The actual production of the ecosystem service, or the amount of surface runoff that does in fact occur, is dependent on the specific rainfall event. The amount and rate of rainfall, and also the antecedent moisture conditions – for example, whether the soils are already saturated due to previous recent rainfall – affect the actual production (USDA-NRCS 1986).

Completely impermeable surfaces intercept a small amount of runoff, for example in small pits or holes, unless they are entirely flat (USDA-NRCS 1986). Water is stored in, and later evaporates from, these topographical depressions. The ecosystem service is therefore provided by the interception/abstraction of precipitation over and above what would occur from impermeable surfaces, due to the presence of permeable and vegetated land covers (see Section 2.1.1).

The spatiotemporal characteristics of this ecosystem service mean that benefits of this service are supplied down-catchment, due to the reduction of accumulated surface flow through the catchment to lower areas. The benefits of the ecosystem service include reduced occurrence and severity of floods, improved water quality for drinking and recreational uses, and non-use values such as lower impacts on biodiversity and lower damage to natural habitats. The components of the generation of this ecosystem service are illustrated in Figure 5.2.

Modelling surface water routing, flow accumulation and flooding is complex, requiring a large time investment. The aim of this project was to derive an understanding of the patterns in several ecosystem services across Sheffield, and achieving this precluded the level of in depth work that would have been required to effectively model the spatial flow of surface waters across the urban area. For this reason only the production of this service was modelled, not the supply.

5.2. Model overview

The model used to estimate surface runoff following a storm event was the USDA-NRCS Soil Conservancy Service's curve number (CN) method. This method assigns a “curve number” to each area according to its vegetation cover and soil type. The curve number describes the capacity of the soil and vegetation to intercept and abstract precipitation, i.e. to prevent runoff; when applied to a specified precipitation scenario, the curve number can calculate the proportion of precipitation that runs off as surface flow (USDA-NRCS 1986). Curve numbers range from zero to 100, with higher numbers assigned to areas with greater runoff; for example, the curve number for urban impermeable surfaces is 98, while that for woodlands ranges from 30 to 77 depending on the soil type. The curve number method is suitable for application in urban areas (USDA-NRCS 1986), and has previously been implemented in similar studies by Whitford et al. (2001) and Tratalos et al. (2007). Full details of the model are given in Appendix A.3.

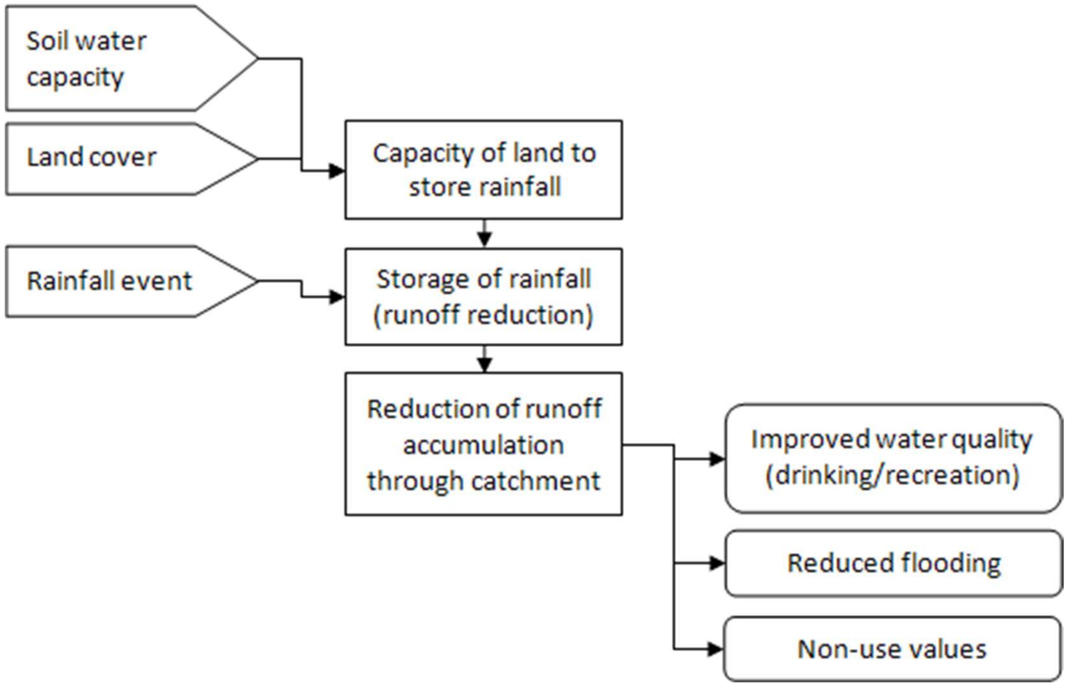


Figure 5.2. Conceptual representation of the storm runoff reduction model. Input conditions are represented by pentagrams; ecosystem processes by rectangles; and socioeconomic benefits by ellipses.

The model uses the land cover map and soils map (and associated soil texture data) as input. Curve numbers were assigned from lists given in a USDA-NRCS technical report (USDA-NRCS 1986), using previous implementations by Whitford et al. (2001) and Tratalos et al. (2007) as guidance. Two rainfall event scenarios were designed: a 'typical heavy rainfall' scenario, representing a fairly common event in Sheffield, with 1.2cm rainfall and soils not especially wet due to recent rainfall (nor very dry); and an 'extreme rainfall' event, based on the June 2007 rainfall that caused extensive flooding in Sheffield, with 6cm rainfall onto already saturated soils. Further details of input data can be found in Appendix A.3.2.

For each scenario, the runoff volume per m^2 was spatially assigned according to the land cover and soils maps, and from this value was subtracted the runoff that would have occurred if each m^2 was covered by artificial, impervious surfaces. This calculated the reduction in runoff due to natural land covers (see Section 2.1.1). The average reduction in runoff from the two scenarios was used as the quantification of the ecosystem service (further details are given in Appendix A.3.3).

In contrast to the air pollution reduction and heat island mitigation models (Chapter 3 and Chapter 4), sensitivity testing was not considered necessary. This is because there are only two input parameters, i.e. the curve number and the amount of rainfall. It is

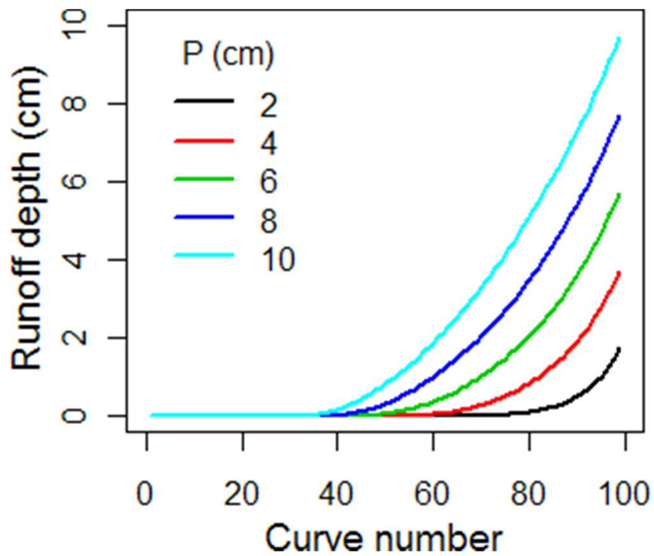


Figure 5.3. Relationships between curve number (*CN*), precipitation depth (*P*) and runoff depth (*Q*) in the storm water runoff reduction model.

clear from the model formulation that the runoff depth increases exponentially with increasing curve number; and that increasing precipitation depth increases the runoff depth that will occur for a given curve number. These relationships are illustrated in Figure 5.3.

5.2.1. Model limitations

USDA-NRCS (1986) lists several relevant limitations of the curve number method. In brief, these are as follows. Curve numbers describe average conditions, and as such disregard spatial heterogeneity within land covers and temporal heterogeneity such as that between seasons. The method also does not describe temporal aspects of any single precipitation event, such as rainfall duration; nor does it include contribution to runoff from subsurface flow or high water tables. There is also an assumption that initial abstraction of rain, before runoff occurs, is a direct function of the curve number; this assumption may not hold if there are surface depressions that hold water. USDA-NRCS (1986) states that the equations are less accurate when the runoff depth is less than 1cm.

Finally, USDA-NRCS (1986) suggests the use of alternative methods to calculate runoff when curve numbers are less than 40. Despite the presence of low curve numbers for a limited number of land covers under normal moisture conditions for the most freely draining soils (see Table 0.18), this method is still used for simplicity. As these areas contribute a comparatively small amount of runoff, this should not be a significant problem.

The usefulness of this model is also limited by the absence of flood modelling. This is especially the case because the urban centre, where runoff is very high, is located at low altitude in the floodplain (Figure 1.3) and thus receives the runoff from higher altitude areas. However, as explained previously, flood modelling was not feasible within this project.

5.3. Results

5.3.1. Model output

The curve numbers and runoff depths for the two scenarios are shown imposed on the land cover/soil map in Figure 5.4 and Figure 5.5 respectively. There is a strong general east-west gradient in curve numbers and consequently in runoff for the normal heavy rainfall scenario (Figure 5.4a, Figure 5.5a): the moorland and some of the agricultural soils and land covers are able to intercept this moderate amount of precipitation (1.2cm), whereas the high proportion of impervious land cover in the urban centre means that almost all rainfall runs off. However, during periods of extremely heavy rainfall, bogs and water bodies become full to capacity. This results in patches of high runoff levels in the moorlands/agricultural lands that are similar to those in heavily developed areas (Figure 5.5b).

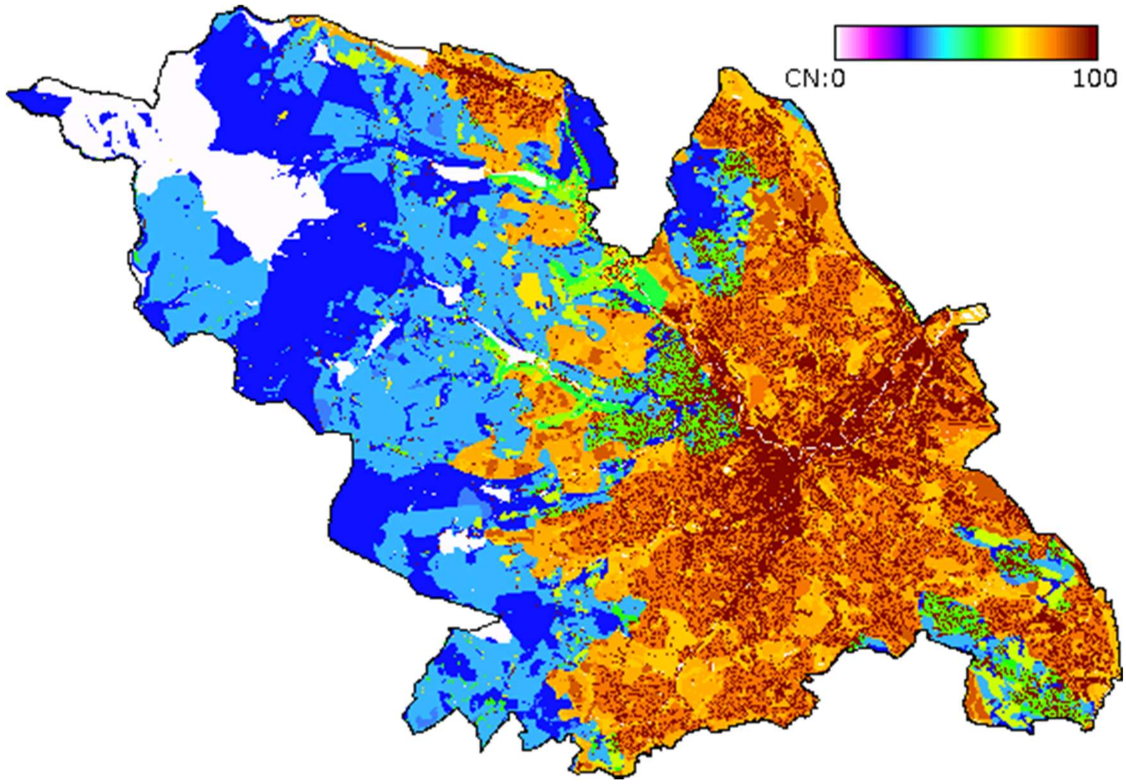
In the absence of natural land covers, curve numbers and runoff levels would be the same everywhere. (Variation in soil type does not affect curve numbers for impervious surfaces because the sealed surface makes soil properties irrelevant.) Runoff levels in the absence of pervious surfaces are high for both scenarios: 0.74cm (of 1.2cm) for the normal heavy rainfall event, and 5.81cm (of 6cm) for the extreme rainfall event.

5.3.2. Ecosystem service maps

The quantified ecosystem service of reduction of storm water runoff is shown in Map 11 for 500m grid squares, Map 12 for Output Areas and Map 13 for Historic Environment Character polygons (all maps shown in Appendix D). Note that higher values in these maps mean that more runoff is prevented by natural surfaces, i.e. a greater ecosystem service is being produced.

The method of ecosystem service quantification (Section A.3.3.1) causes the influence of the extreme rainfall scenario to outweigh that of the normal rainfall

a. Normal antecedent moisture conditions



b. Saturated antecedent moisture conditions

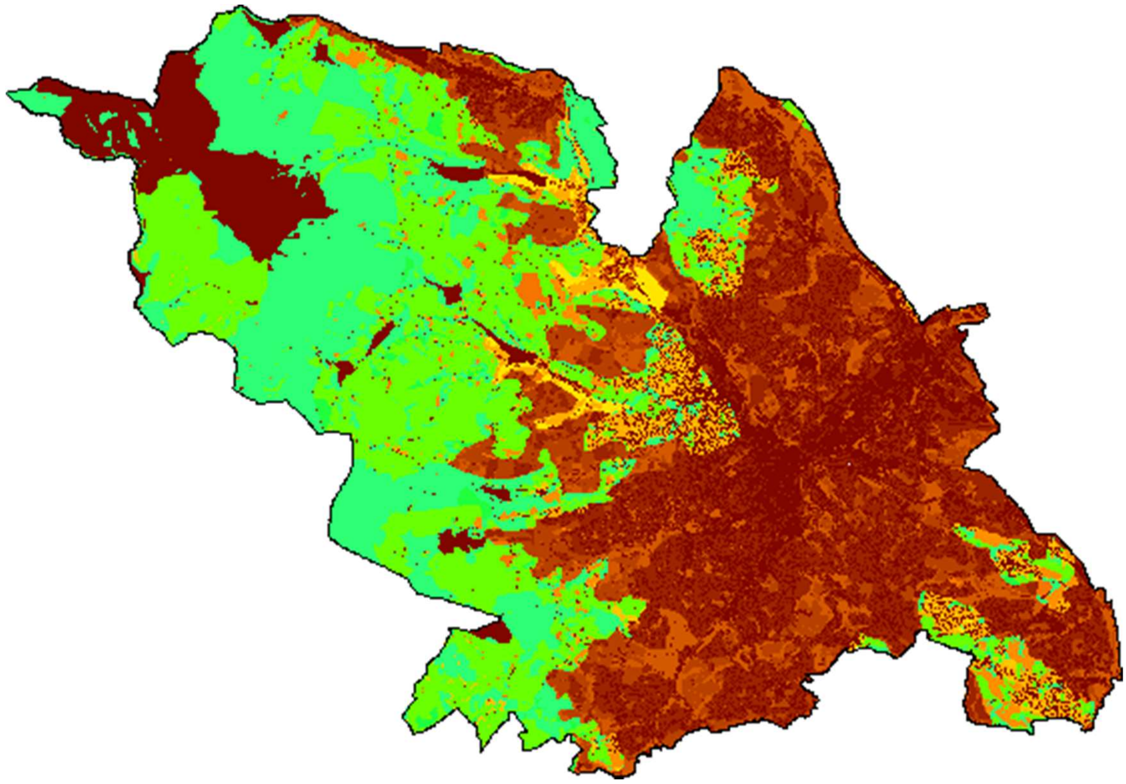
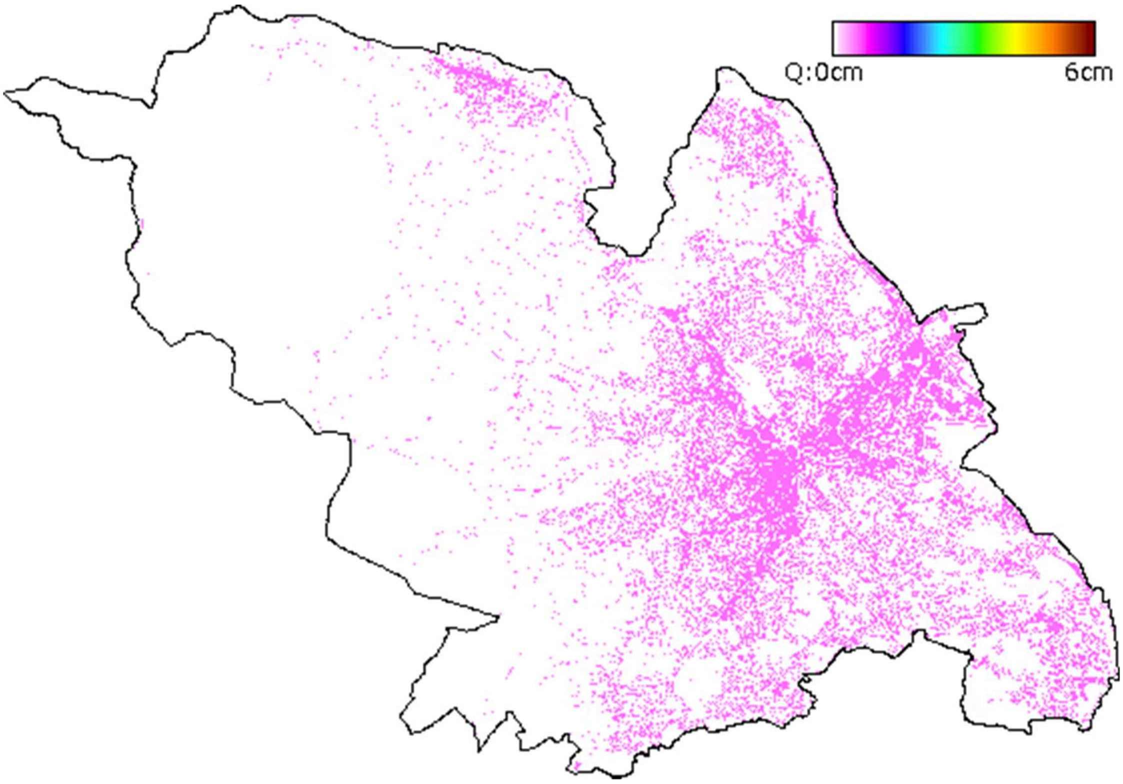


Figure 5.4. Curve numbers (*CN*) under (a) normal and (b) saturated antecedent soil moisture conditions. Both maps use the pseudo-continuous linear colour ramp shown in the upper right of the figure, where white is *CN* = 0 and deep red is *CN* = 100.

a. Normal rainfall scenario



b. Extreme rainfall scenario

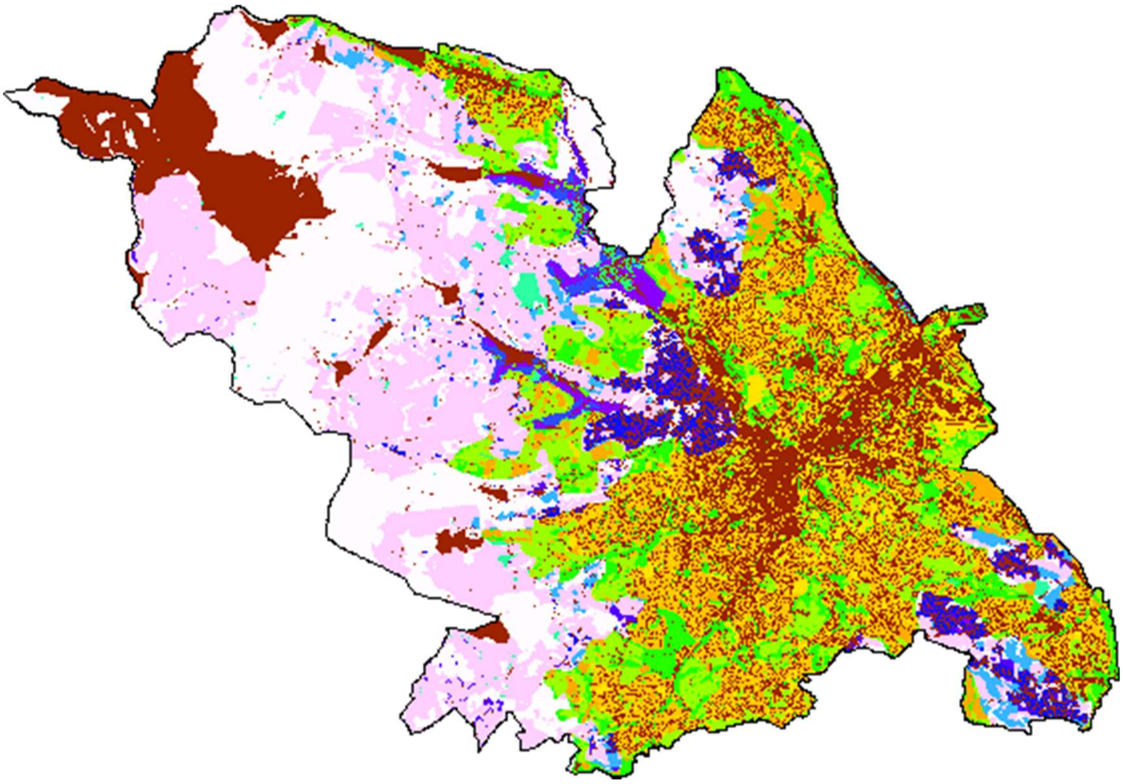


Figure 5.5. Runoff depth (*Q*, in cm) for (a) the normal rainfall scenario with normal antecedent soil moisture conditions and (b) the extreme rainfall scenario with saturated soil. Numbers in brackets are ranges of values; the maps both use the pseudo-continuous linear colour ramp shown in the upper right of the figure, where white is 0cm runoff and deep red is 6cm runoff.

scenario considerably, and this can be seen in Map 11, where the area of bog moorland in the northwest corner of the study area has a similar ecosystem service values as the urban centre. Given that the amount of damage done is far greater in extreme rainfall events, however (see the discussion in Section 5.1.1), this is appropriate.

Levels of ecosystem service production are lowest in the bog moorland and the commercial/industrial urban area, and many times higher in the heath moorland and the western part of the agricultural belt (Map 11). When the model is aggregated to other spatial units, however, the poor levels of ecosystem service provision due to the bog moorland become less visible; indeed, this area now appears to be a relatively good provider of the ecosystem service at the Output Area scale (Map 12). This is because of differences in polygon size between the three types of spatial unit. In the western part of the study area, the Output Areas and Historic Environment Character areas are large and are more likely to contain a wide variety of land covers providing different levels of the ecosystem service. The small 500m grid squares are able to pick out finer details such as the bog moorland. In contrast, Output Areas and Historic Environment Character areas in the city centre are generally small and thus pick out more variation in ecosystem service production in this part of the study area than the 500m grid squares.

Despite the general gradient, as with the other ecosystem services, there is clearly considerable local variation in ecosystem service production. This is most obvious with the Historic Environment Character area map (Map 13), presumably because the boundaries delineate areas with different land uses such as industry and parks. It can also be seen that Output Areas of similar size, and thus with similar population density, can have quite different levels of ecosystem service production (Map 12). These patterns suggest that urban design can have a strong influence on the generation of this ecosystem production.

5.4. Discussion

Although it was not possible to translate the estimates of surface runoff to flood modelling within this study, it is clear that the more upland regions outside of the urbanised area (with the exception of the bog moorland) have relatively low levels of runoff (Map 11). This means that a significant proportion of surface runoff that would otherwise flow down-catchment into the urbanised area is prevented, reducing the risk of flooding in these heavily populated areas. Thus the value of the ecosystem service produced in the moorlands and much of the agricultural belt to residents of the heavily urbanised areas is high.

Historically, there would have been even less surface flow accumulating to the river basins, but throughout the centuries Sheffield has encroached on the surrounding countryside and more of the area within the city itself has been sealed by impervious surfaces (Hey 1998). Today the remaining moorlands are designated as Sites of Special Scientific Interest, giving them a degree of protection from development and land management pressures (Natural England 2009). Much of the valuable agricultural land and woodland is not protected in this way, however; although the Sheffield Development Framework states that “the city’s rural setting will be safeguarded and enhanced. Most of the countryside will remain protected as Green Belt to support urban and rural objectives” (Sheffield City Council 2009a), indicating that these areas are valued. Green Belt designations are a policy designed to control urban growth, but the results from this model suggest that the designation has benefits in terms of ecosystem services also.

Climate change predictions indicate that the ecosystem service of storm water runoff reduction will become more valuable as the frequency of high intensity rainfall increases (UKCIP 2002). Indeed it is recognised by Sheffield City Council that it is important to protect the remaining areas able to intercept a large amount of precipitation and to consider flood risks when planning urban development (Sheffield City Council 2009a). Map 11 shows that the highest surface runoff levels occur from the bog moorland and the city centre. The bog moorland is a valuable habitat for other reasons, however, and it is unclear how climate change will affect this habitat type (UK Biodiversity Group 1999). Therefore it seems more sensible to focus on improving the surface water interception/abstraction capacity of urbanised areas.

Steps being taken by Sheffield City Council to minimise flood risks to residents include avoiding developing high-risk areas and, where possible, designating these areas for open space uses; ensuring developments have minimal surface runoff; developing “water-compatible uses” in the functional floodplain and providing open space at riversides to minimise damage from overflowing rivers; avoiding culverting or building over watercourses, and removing culverts where possible; and implementing sustainable drainage techniques, such as green roofs (Sheffield City Council 2009a). In other words, the City Council recognises the need to spatially plan land use around flood risk, and to use land cover patterning to reduce surface runoff.

A popular approach to introducing pervious land covers into areas at risk of storm water flooding is the rain garden (Holman-Dodds et al. 2003, Shuster et al. 2007). As the name suggests, rain gardens are open spaces with horticultural land covers sited to intercept accumulated surface water (Shuster et al. 2007). Rain gardens are especially effective in lower intensity rainfall events, but also help to reduce runoff accumulation during heavier rainfall (Holman-Dodds et al. 2003). Rain gardens, however, require free space for implementation, which can be a problem in more heavily urbanised areas.

Green roofs are cited by many authors as an alternative or complement to the use of natural land cover to reduce surface runoff, and can be implemented in heavily developed areas (VanWoert et al. 2005, Villarreal and Bengtsson 2005, Getter et al. 2007, Oberndorfer et al. 2007). Studies have found that, depending on the construction and slope, green roofs can intercept a considerable amount of rainfall (e.g. 12mm) before runoff begins, and continue to intercept a substantial proportion of rainfall even in extended and heavy rainfall events (VanWoert et al. 2005, Villarreal and Bengtsson 2005, Getter et al. 2007). Furthermore, green roofs delay and extend the period over which runoff occurs, reducing the total surface flow at any one time (VanWoert et al. 2005, Getter et al. 2007). Green roofs may therefore be a more appropriate method of runoff reduction in the most heavily urbanised areas with little remaining open space.

6. Carbon storage

6.1. Introduction

One recent estimate suggests that, globally, thirty to forty percent of anthropogenic greenhouse gas emissions are produced in cities; and that due to the import of energy and goods, city dwellers are responsible for at least sixty to seventy percent of all emissions (Satterthwaite 2008). The likely consequences of projected climate change due to greenhouse gas emissions for biodiversity, water availability, food production, extreme weather events and floods, and human health are now well known (IPCC 2007). Greenhouse gas and climate change management are therefore increasingly being incorporated into urban planning and development: Sheffield Development Framework's core strategy, for example, includes plans to reduce the city's impact on, and to adapt to, climate change (Sheffield City Council 2009a).

One service that ecosystems provide with respect to climate change is reduction of carbon dioxide levels, by the sequestration and storage of carbon. Carbon storage in plants and soils is an important aspect of global carbon regulation. Climate change is a global phenomenon, meaning that the consequences of greenhouse gas production are felt worldwide regardless of the site of production, and that the location of removal of carbon dioxide from the atmosphere does not need to be close to the site of production (IPCC 2007). It is nonetheless important to understand the contributions of different sorts of environment to this process. This chapter focuses on assessing the capacity for, and spatial pattern of, carbon storage in urban greenspace.

6.1.1. Carbon dioxide and urban greenspace

Urban forests in the United States have been estimated to store over 700 billion tonnes of carbon – approximately 25 tonnes per hectare – and to sequester more than twenty million tonnes of carbon each year (Nowak and Crane 2002). In four cities in Korea, sequestration by trees offsets carbon emissions by 0.5-2.2% (Jo 2002). Urban gardens can in some situations store more carbon per area than the surrounding ecosystems, such as Colorado's native grasslands or agricultural fields (Golubiewski 2006), and urban residential soils can contain more organic carbon per area than forest soils (Pouyat et al. 2006). These examples indicate that urban greenspace can play an important role in reducing atmospheric carbon dioxide levels, especially where management regimes are designed to maximise benefits (Nowak and Crane 2002, Nowak et al. 2002b).

Sheffield in particular is likely to store a large amount of carbon, due to peaty soils and natural vegetation in the Peak District National Park as well as a relatively high proportion of greenspace within the urbanised area. Furthermore, the City Council intends to enhance the greenspace network and preserve the surrounding countryside (Sheffield City Council 2009a); and although the link between greenspace and carbon management is not explicitly made in the Sheffield Development Framework core strategy, estimation of carbon storage will add further weight to greenspace management and planning.

6.1.2. Generation of the ecosystem service

The actual ecosystem service provided by carbon storage is prevention of anthropogenic climate change. However, because of the global nature of the carbon cycle, carbon storage in Sheffield has negligible impact on anthropogenic climate change unless the same patterns are occurring worldwide. Therefore it is not possible to quantitatively link the production of this ecosystem service to the supply, or to the benefits of that supply.

Nevertheless, the value of climate stability is indisputable. The most recent report of the Intergovernmental Panel on Climate Change suggests that climate changes are likely already happening, and that changes of greater magnitude will occur in the future (IPCC 2007). The probable consequences of these changes are numerous and many are severe: for example, many ecological processes will be disrupted with the result that ecosystems may undergo state changes; crop productivity may increase but then decrease with greater warming; freshwater availability will increase in some areas but decrease in others; and sea level rise will increase the frequency and severity of floods (IPCC 2007).

6.2. Model overview

The carbon storage model is simple, using land cover based estimates of carbon biomass in different types of vegetation, and estimates of the organic carbon content of soils from the NATMAP soils map. Soils under manmade surfaces and buildings were assumed to have half the carbon content that they would otherwise, because the development process generally causes large losses of carbon (Pouyat et al. 2006). Soil carbon losses can occur during development regardless of whether the soil is directly disturbed or not, due to the loss of plant, microbial and earthworm biomass, which reduces inputs of organic matter (the source of carbon) to the soil (Byrne et al. 2008). Direct disturbance can also expose deeper soil layer to conditions in which carbon is likely to oxidise to the atmosphere (Jandl et al. 2007).

In order to quantify the ecosystem service provided by natural land covers, an estimate was also made of the carbon content of the different soil types when under sealed surfaces, and this second estimate was subtracted from the first estimate of actual carbon storage. Further details of the methods used to model carbon storage can be found in Appendix A.4.

Sensitivity testing was not performed for the carbon storage model because there are few parameters and it is obvious how they relate to each other. Carbon in vegetation biomass is a single parameter, and soil organic carbon is calculated by a simple equation relating soil bulk density and percent carbon by weight, summed over each soil horizon.

6.2.1. Model limitations

As with the other ecosystem service models, the carbon storage model treats all land cover types as homogeneous; in addition, the soils within each NATMAP mapping unit are treated as homogeneous. The carbon content of soils even within a soil type is known to be exceptionally variable, however, due to its dependence on factors such as local land cover, climate, management and disturbance history (Guo and Gifford 2002, Pouyat et al. 2006, De Deyn et al. 2008). The carbon content of vegetation will also depend upon factors such as the species composition, vegetation age and management at a site (Cruickshank et al. 2000, Nowak et al. 2002a). Unfortunately there is no way to overcome this limitation without undertaking fieldwork beyond the feasibility of this study.

6.3. Results

6.3.1. Model output

Figure 6.1 shows estimates of vegetation and soil carbon imposed on an intersection of the land cover and NATMAP soil maps. Vegetation carbon estimates (Figure 6.1a) show a spatially complex matrix, with levels ranging from 0-1500 g m^{-2}. In the urban centre this complexity is due to buildings and roads with no carbon being interspersed with gardens, parks and other green areas that often include vegetation types with high carbon levels. In the less developed areas, the matrix arises from the relatively low carbon content of arable and grassland (and zero carbon in water bodies), as compared to the higher levels in woodland and scrub.

The estimated carbon content of soils is up to two orders of magnitude greater than that of vegetation (Figure 6.1b). Average levels are far lower than this, but still considerably higher than that of vegetation. The map shows that the soils at higher altitudes contain more carbon, and that riparian soils generally contain more carbon than other nearby soils. The pattern of urban development is also apparent in Figure 6.1b from the fine pattern of lower soil carbon levels.

The ecosystem service of carbon storage is quantified from estimates of levels that would occur in the absence of natural ecosystems subtracted from the spatial data shown in Figure 6.1. For storage in vegetation, this value is zero. Soil storage estimates are shown in the inset map in Figure 6.1b.

Table 6.1 shows the estimated total carbon storage and average areal carbon density across Sheffield.

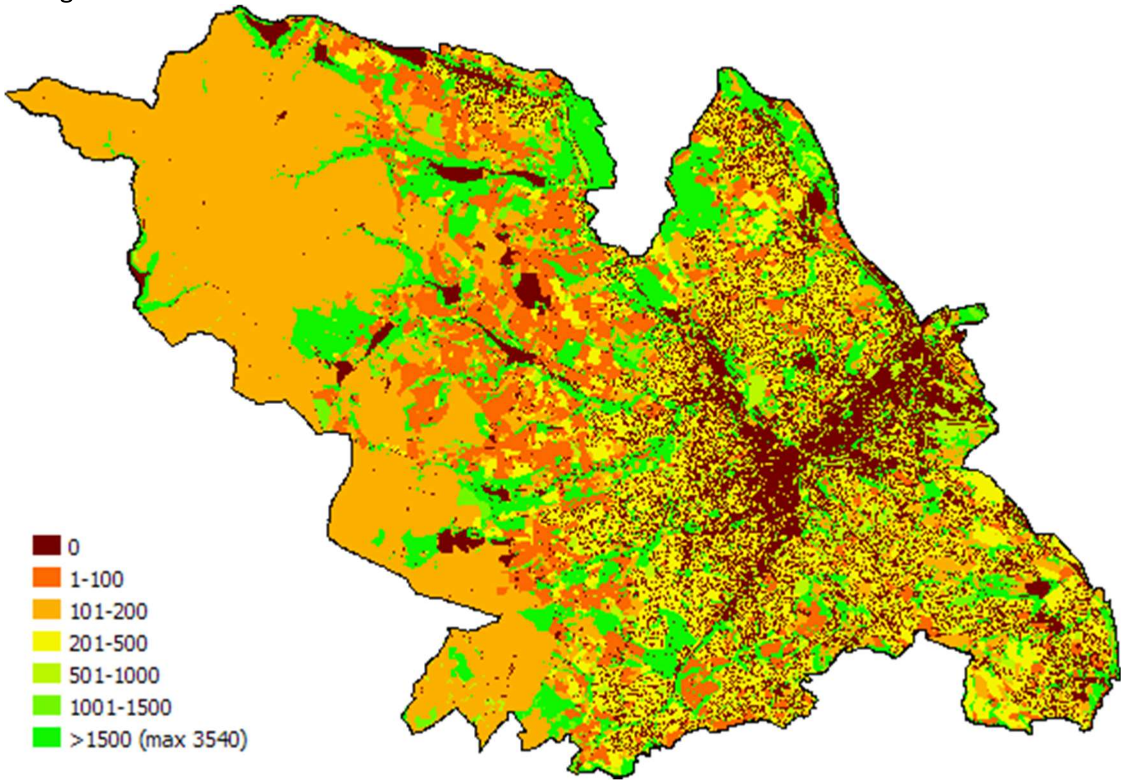
6.3.2. Ecosystem service maps

Estimates of the ecosystem service of carbon storage in soils and vegetation aggregated to 500m grid squares are shown in Map 14 (all maps shown in Appendix D).

Table 6.1. Carbon storage statistics for Sheffield.

	Total storage (Mg)	Areal average (g m^{-2})
Soil	1.59×10^7	4.32×10^4
Vegetation	2.49×10^5	6.78×10^2
Total	1.62×10^7	4.39×10^4

a. Vegetation carbon



b. Soil carbon

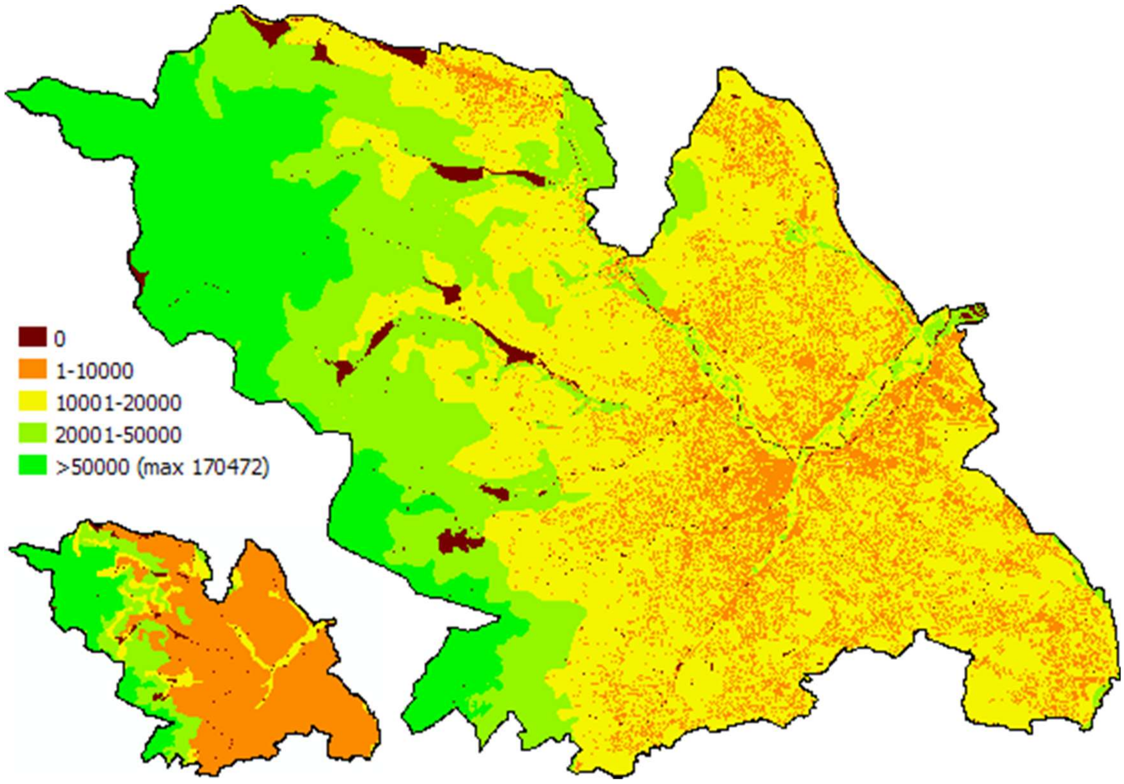


Figure 6.1. Estimates of carbon stored in (a) vegetation and (b) soils in the study area. Inset in map (b) shows estimates of soil carbon that would be occur if only impervious surfaces existed. Units: g m^{-2}.

Unsurprisingly, levels of carbon storage are lowest in the city centre, due to a lack of vegetation, and soils with relatively low carbon contents even prior to development. There are also some more distant areas that appear to have very low levels of carbon storage, which is caused by the presence of water bodies: the NATMAP dataset does not provide estimates of the carbon content of soils under water bodies. The suburban and agricultural areas store more carbon, due to a higher proportion of vegetated land covers. It is the moorlands, however, that provide the greatest levels of carbon storage, due to the peat soils in this region.

Map 15 shows carbon storage at the Output Area level. The patterns are essentially the same as for the 500m grid squares (Map 14), because there is little variation in carbon storage levels to the west of the study area where Output Areas are large. This means that there is not much detail to be lost during aggregation to larger polygons. However, additional detail can be seen at the Historic Environment Character area level (Map 16). This is likely because the polygons represent areas with a consistent land use, and which are therefore likely to have a characteristic land cover pattern. Small developed areas (e.g. industrial estates) and water bodies are therefore individually delineated and are shown to have low carbon storage levels; similarly, individual parks and greenspaces in the heavily urbanised area can be seen, showing that carbon storage is not uniformly low, as might be assumed from Map 14.

6.4. Discussion

Sheffield is located in a region of naturally very high carbon storage levels due to the prevalence of soils of high organic carbon content (Cranfield University 2009). As the city of Sheffield has grown, developed land uses and agriculture have encroached upon the edge of the moorlands, where soil carbon densities are highest (Hey 1998, Cranfield University 2009). Such land use conversion is very frequently associated with loss of soil carbon (Pouyat et al. 2006). Nevertheless, large areas of moorland remain relatively undisturbed.

This model was unable to address carbon sequestration, as estimates of carbon fluxes for local soils were not available and sequestration by vegetation depends on factors not represented by the land use/cover maps and for which data could not be found (e.g. annual vs. perennial vegetation, land management). However, the present and future flux of carbon from UK peat soils, such as those found in Sheffield's moorlands, is a subject of current climate research interest due to the fact that they represent a very large proportion of the UK's total carbon reserve (Moors for the Future 2007).

A recent estimate of the carbon flux from an upland peat catchment in the North Pennines (Sheffield's moorlands are located in the South Pennines) indicates that these soils are presently a sink for atmospheric carbon gases but a net source of carbon (including soil respiration, methane emission, dissolved and particulate organic carbon and excess dissolved carbon dioxide); and that the rate of total carbon loss is likely to increase due to predicted changes in rainfall and temperature, with the gas sink reducing in size (Worrall et al. 2007). Nevertheless the fact remains that peatlands in good condition in a suitable climate can sequester large amounts of carbon (UK Biodiversity Group 1999). However, some areas of the Peak District moorlands are presently in poor condition, causing the soils to erode and lose carbon (Moors for the Future 2007). In order to minimise the increasing risks of carbon loss from moorlands due to climate change, it is important that they be maintained in pristine condition (UK Biodiversity Group 1999).

The areal averages of carbon stored in Sheffield are shown in Table 6.1. Pouyat et al. (2006) estimated the belowground and aboveground carbon storage for six US cities; the results are shown alongside those for Sheffield in Table 6.2. It is clear that the soil carbon density estimates for the US cities are far lower than that for Sheffield, which is likely because none of the US cities contains a high proportion of high-carbon wetland soils within its borders. Sheffield also has a far lower proportional impervious cover (15.5%) than the US cities (39.8-53.9%), meaning that less of the soil has lost carbon through disturbance and intensive management. In addition, the study by Pouyat et al. (2006) includes only soil carbon to a depth of 1m, whereas the present study includes all soils to the bedrock. This is particularly significant for the peaty soils, which can be very deep.

In contrast, the storage of carbon in vegetation is relatively low in Sheffield (Table 6.1). This is because of the prevalence of land cover types that do not store large amounts of carbon in vegetation, especially moorland and agricultural land (arable and improved grassland). In the US cities, there is a higher proportional cover by forests and residential gardens, which store far more carbon in vegetation (Pouyat et al. 2006). Nevertheless, the huge amount of carbon stored in its soils means that Sheffield has by far the highest areal mean carbon storage.

Table 6.2. Average carbon storage (kg m^{-2}) for six US cities and Sheffield. Source for US cities: Pouyat et al. (2006).

City	Soil	Vegetation
Atlanta	7.8	3.6
Baltimore	6.3	2.5
Boston	5.9	2.0
Chicago	5.5	1.4
Oakland	5.9	1.1
Syracuse	7.1	2.4
Average of above	*6.3*	*2.1*
Sheffield	43.9	0.6

The fact that carbon storage in Sheffield is so heavily skewed towards soil, rather than vegetation, suggests that management actions to improve total carbon storage in Sheffield would have a far greater potential if focused on soil. The estimates of soil storage made by this model are only approximations, because of the complex impacts of both regional climate variables that determine Net Primary Productivity and rates of organic matter decomposition; and smaller scale effects of soil drainage (which is determined by topography and soil texture), litter quality, and the presence of nitrogen fixing plants (Johnson and Curtis 2001, Pouyat et al. 2006, Jandl et al. 2007). Specifics of land use history also influence present carbon storage (Guo and Gifford 2002, Golubiewski 2006, Takahashi et al. 2008). Further factors complicating soil carbon prediction in urban environments are human disturbances such as the introduction of exotic plants, fertilisation, irrigation, clipping; and urban climate modifications, including the heat island effect and locally elevated carbon dioxide concentrations (Golubiewski 2006, Pouyat et al. 2006).

It is nevertheless certain that, under the right conditions, soils in urban soils and greenspaces can store large amounts of carbon – sometimes even more than undisturbed native land uses (Jo and McPherson 1995, Golubiewski 2006, Byrne et al. 2008, Takahashi et al. 2008). Management actions that improve plant productivity, such as irrigation and fertilisation, and leaving plant litter in situ, can help to increase soil carbon contents (Golubiewski 2006, Pouyat et al. 2006, Takahashi et al. 2008). Establishing tree cover also seems to increase levels of soil carbon storage compared to grassland, as well as storing more carbon within its biomass (Pouyat et al. 2006, Takahashi et al. 2008).

A final point to note is that Output Areas (Map 15) of similar size can have quite different levels of carbon storage. Since many of these Output Areas are over the same soil type (Map 4), this indicates that there is a potential for appropriate urban planning to increase levels of carbon storage.

7. Opportunities for cultural ecosystem services

7.1. Introduction

There now exists a considerable body of evidence showing the importance of publicly accessible greenspace in urban areas to both the physical and psychological health of residents. Tzoulas et al. (2007) recently reviewed this evidence, concluding that "ecosystem services provided by a Green Infrastructure [urban greenspace] can provide healthy environments and physical and psychological health benefits to the people residing in them. Healthy environments can contribute to improved socio-economic benefits as well". Consequently there have been calls to use interactions between nature and humans (i.e. cultural ecosystem services), and improvement of the provision of nature within the living environment, as a means to promote public health (St Leger 2003, Stokols et al. 2003, Payne et al. 2005); and Irvine et al. (2010) suggest that appropriate urban planning might be one way to achieve this.

Although it is difficult to define precisely the ecosystem service that is being provided to people by greenspace (Wallace 2007), it is clear that either active or passive use of greenspace improves physical and psychological health (Section 7.1.1). It therefore follows that accessible greenspace is an important and valuable contributor to quality of life (Handley et al. 2003). This chapter presents and discusses the index used to quantify the production and supply of health benefits provided by cultural ecosystem services, i.e. interactions between humans and nature.

7.1.1. Health benefits of cultural ecosystem services provided by urban greenspace

Simply viewing greenspace is sufficient to provide some health benefits. For example, hospital patients recover from surgery faster and require fewer analgesics if their room has a view of a natural landscape (Ulrich 1984). Studies from public housing blocks indicate that residents have greater effectiveness in facing major life issues and suffer less intra-family aggression due to an improvement in attentional functioning and reduced mental fatigue if they have a view of a green landscape (Kuo 2001, Kuo and Sullivan 2001a). There have also been experimental studies showing that viewing a natural environment after watching a stressor video increases the rate of recovery from stress, as assessed by self-reporting and by physiological measures such as blood pressure and skin conductance (Ulrich et al. 1991, Parsons et al. 1998). Moreover, viewing a natural environmental can reduce stress experienced due to subsequent events (Parsons et al. 1998).

The active use of greenspace, i.e. actually being in an urban greenspace, provides further psychological benefits. Many studies have found that being in a green place improves attentional functioning, and play in green areas has been shown to improve symptoms in children with attention deficit hyperactivity disorder (Wells 2000, Taylor et al. 2001, Hartig et al. 2003, Kuo and Taylor 2004). Visiting greenspace also provides psychological restoration in the form of stress relief, mood improvement, reduction of negative feelings such as anger, opportunities for reflection, and recovery from mental fatigue (Herzog et al. 1997, Kuo 2001, Hartig et al. 2003, Payne et al. 2005). Greenspace is also associated with improvements in self-discipline, at least in girls (Taylor et al. 2002).

There is evidence that people use urban greenspace as a form of self-regulation. Many studies have found that natural places are commonly cited as being amongst a person's favourite places, especially for people with more negative moods or worse perceived health (Korpela and Hartig 1996, Korpela 2003, Korpela and Ylén 2007). People think of their favourite places as having restorative qualities such as increasing positive feelings and decreasing negative feelings, providing opportunities for reflection and relaxation, recovering attention focus, and providing a good environment in which to face matters on one's mind (Korpela 1989, Korpela 1992, Korpela et al. 2001). It is clear that people implicitly value the benefits they obtain from being in greenspace.

Visiting public greenspaces such as parks usually coincides with physical activity such as walking, and indeed people are more likely to be physically active and choose to participate in sports and other physical activity if they live close to a park (Booth et al. 2000, Pikora et al. 2003, Payne et al. 2005). These higher activity levels are very likely to translate into improved health; one study has found that senior citizens who live close to a lot of greenspace suitable for walking in have greater longevity (Takano et al. 2002).

Several other studies have looked specifically at the amount of greenspace in a neighbourhood, finding that more greenspace correlates with better self reported physical and psychological health in the community (de Vries et al. 2003, Payne et al. 2005, Maas et al. 2006). Furthermore, public greenspace can also improve the community itself: there is evidence that green landscaping in public areas encourages social interactions and increases residents' sense of community, and is also associated with lower levels of criminal, anti-social and aggressive behaviour (Coley et al. 1997, Kuo and Sullivan 2001b, Kim and Kaplan 2004).

7.1.2. Recommendations for urban greenspace provision

This fact that urban greenspace provides large benefits to human health is recognised by English Nature's recommendations for the provision of greenspace in the urban environment. Specifically, these standards are (Handley et al. 2003):

> "That no person should live more than 300m from their nearest area of natural greenspace of at least 2ha in size; provision of at least 1ha of Local Nature Reserve per 1,000 population; that there should be at least one accessible 20ha site within 2km from home; that there should be one accessible 100ha site within 5km; that there should be one accessible 500ha site within 10km."

Regarding the definition of "natural greenspace", English Nature state that this means "areas naturally colonised by plants and animals" (Handley et al. 2003). In practice, however, in urban areas greenspace does not need to meet this strict definition to provide benefits to people (Handley et al. 2003, references in Tzoulas et al. 2007), as long as it is publicly accessible.

7.1.3. Generation of ecosystem services

The evidence presented in Section 7.1.1 and the recommendations for provision listed in Section 7.1.2 indicate that merely the existence of urban greenspace does not provide psychological and physical health benefits to people. It is also important that the greenspace is accessible for use, whether active (as in visiting the greenspace) or passive (viewing the greenspace from a distance). This is because the supply of the health benefits identified in Section 7.1.1 is dependent on people travelling to (or being near to, in the case of passive use) the public greenspace. This is in contrast to the production of the potential to receive health benefits, which is provided by the presence of suitable types of greenspace. Put another way, the existence of greenspace produces opportunities for people to receive cultural ecosystem services, but the provision of these opportunities to people depends on people being able to travel to the greenspace.

In the case of opportunities for cultural ecosystem services, it was possible to model both the production (existence of greenspace) and supply (proximity of greenspace to people). However, it is not possible at present to quantitatively relate access to service-providing greenspace to the actual health benefits obtained in a way that is generalisable between studies (Tzoulas et al. 2007). Furthermore, the benefits to any individual will depend upon the way that person chooses to use any accessible greenspace. Therefore the benefits provided by this ecosystem service could not be modelled.

7.2. Index overview

The model of access to opportunities for cultural ecosystem service in greenspace is, more strictly, an index, as unlike the other models discussed so far the output does not directly describe something tangible. Rather, it ordinally describes the spatial availability of greenspace infrastructure to the general public, such that members of the public have the opportunity to visit greenspace and obtain benefits of the kind described in Section 7.1.1, should they choose to take those opportunities.

The model calculates the production of opportunities as the proportion of an area of interest that is covered by land uses that are considered to provide such opportunities (e.g. public parks, moorland, woodlands).

The supply of opportunities was previously modelled by Barbosa et al. (2007), using GIS layers of unique addresses, the transport network and public greenspaces to calculate the travelling distance from each address to the nearest accessible greenspace. Unfortunately, the GIS layer of addresses was not available for the present study and consequently an alternative approach had to be developed. The supply of opportunities is therefore quantified using the four distance-related greenspace provision standards given by English Nature, and listed above. The standard detailing a requirement for Local Nature Reserve provision is not included as this is not spatially explicit. Actual use of greenspace, which is only indirectly related to the provision of greenspace, is not considered here due to the difficulty of obtaining data. Areas of publicly accessible greenspace were identified from the Historic Environment Character area dataset.

7.2.1. Index calculation

The index makes use of the Sheffield land use map described in Section 2.2.2. The land use legend was studied in order to identify whether areas of each category were likely to fulfil two requirements: firstly, that greenspace is a major component of that land use; and secondly, that the greenspace is freely publicly accessible. Table 7.1 shows the land use categories that were considered to meet these requirements.

These areas were identified on the land use map and used to generate a map of publicly accessible greenspace. The land use map does not extend to cover a reservoir at the western edge of Sheffield. When generating the map, this area was included as publicly accessible greenspace.

The proportion of each area of interest meeting each of the recommendations described in Section 7.1.2 was calculated, i.e. within 300m of a 2ha greenspace, 2km of a 10ha greenspace, 5km of a 100ha greenspace and 10km of a 500ha greenspace; Handley et al. (2003). These proportions were summed and divided by four in order to calculate an index quantifying this ecosystem service. Areas of greenspace located only partially within, or nearby to the boundaries of the study area (e.g. the Rother Valley Country Park) were also included in generating these proportions.

7.2.2. Index limitations

The main limitation of this index is that it assesses only opportunities for the provision of cultural ecosystem services. Although accessing urban greenspace has

Table 7.1. Land use categories considered to be likely to contain freely publicly accessible greenspace.

Broad type	Land use
Institutional	Cemetary
Unenclosed land	Commons and greens Moorland Regenerated scrubland
Ornamental, parkland and recreation	Inner city farm Deer park Playing fields/recreation ground Public park
Water bodies	Reservoir
Woodland	Ancient woodland Plantation Semi natural woodland Spring wood

been related to a large number of physical and psychological benefits (see Section 7.1.1), proximity to greenspace does not necessarily mean that an individual will make use of that greenspace. For example, greenspaces that do not appear safe or well maintained are less likely to be used (Coles and Bussey 2000, Jorgensen et al. 2002). Such factors could not be included in this index.

Nevertheless, some benefits have shown to accrue from passive use; for example, visible vegetation is related to improved attentional functioning and reduced aggressional levels in public housing (for more details and further examples see Section 7.1.1). In addition, studies of adolescents and adults have shown positive relationships between physical activity and the number and variety of nearby parks (Giles-Corti et al. 2005, Cohen et al. 2006), suggesting that physical health benefits may also be related to proximity-based measures of park accessibility.

7.3. Results

7.3.1. Model output

A total of 15,174 ha of publicly accessible greenspace was identified within the study area, and an additional 2,138 ha from outside the study area boundary was included in calculation of the supply index in order to accurately capture the delivery of benefits (Figure 7.1). High proportions of the study area meet each of the distance-related criteria (Figure 7.2): 79.9% of the area is within 300 m of a 2 ha site; 99.2% is within 2 km of a 10 ha site; 98.4% is within 5 km of a 100 ha site; and 78.2% is within 10 km of a 500 ha site. Much of Sheffield meets at least three of the criteria.

Interestingly, Figure 7.2a suggests that provision of small, local greenspaces is better in the city centre than in some of the less urban areas. Indeed much of the agricultural belt is poorly provided by these local greenspaces (agricultural land is not considered here to be "publicly accessible greenspace").

7.3.2. Ecosystem service maps

In contrast to the ecosystem services previously discussed, separate maps have been generated for the indices of production, i.e. existence of publicly accessible greenspace, and supply, i.e. proximity of publicly accessible greenspace to a given location. The maps of production, which are used in further analyses, are shown in Map 17 (500m

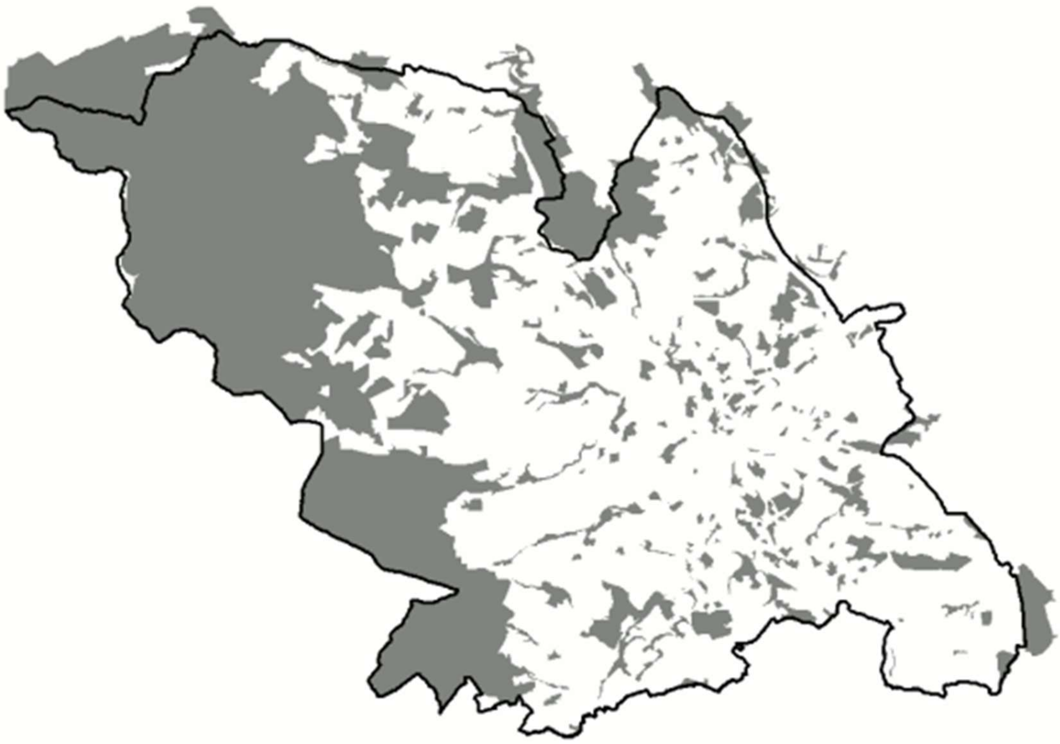


Figure 7.1. Areas of publicly accessible greenspace (shown in grey), including areas outside the study area that are used in calculating the index of accessibility.

grid squares), Map 18 (Output Areas) and Map 19 (Historic Environment Character areas; all maps shown in Appendix D). The maps of supply are shown below in Figure 7.3.

As would be expected from the distribution of publicly accessible greenspace (Section 7.3.1), the production of the ecosystem service is high in the moorland to the west of the study area, and low in the agricultural belt and urban centre with but localised areas of high production (Map 17, Map 18, Map 19). Map 19, which aggregates the index to Historic Environment Character area polygons, simply shows a matrix of polygons that are either entirely or not at all publicly accessible greenspace, because it is the Historic Environment Character area polygons from which these areas were identified (see Section 7.2).

Despite the locally poor production of this ecosystem service in many places, a large proportion of the study area is well supplied by both large and small local greenspaces (Figure 7.3a). The absence of any very large greenspaces in the east of the study area means that only the parts of Sheffield within 10km of the moorlands can meet all four

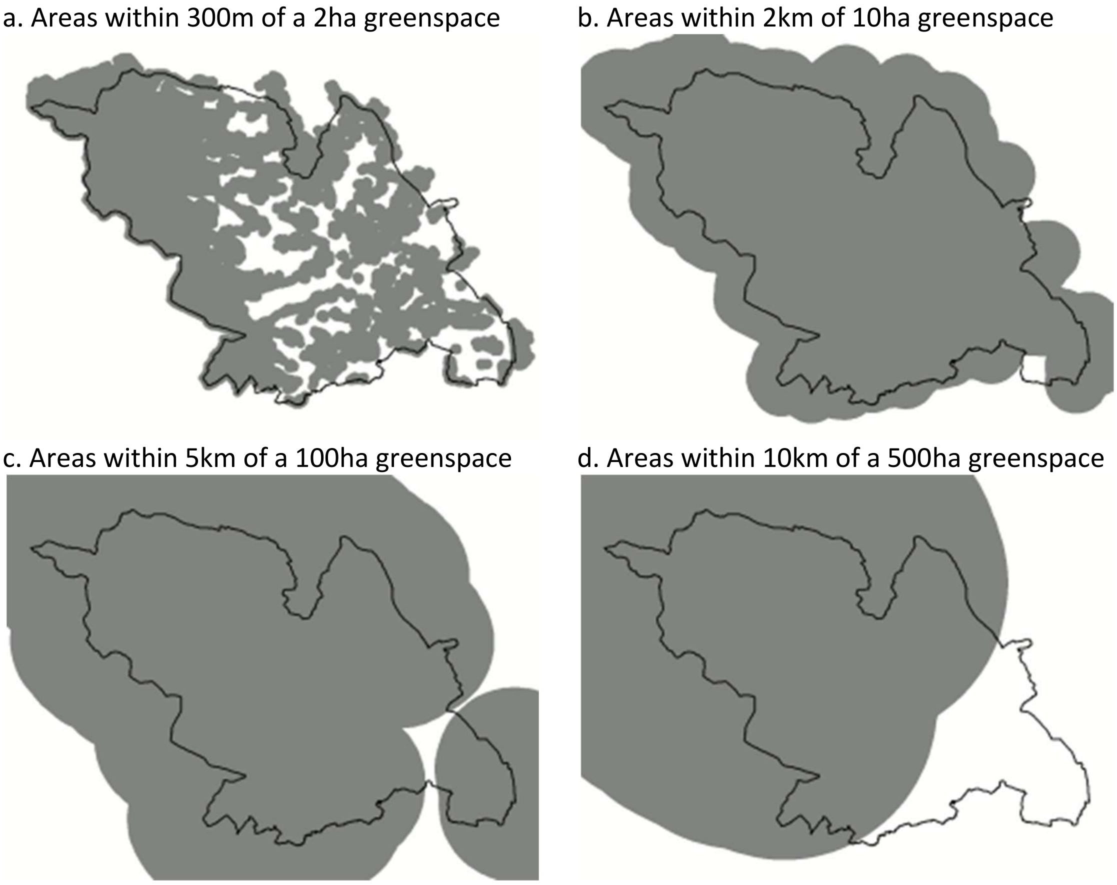


Figure 7.2. Areas satisfying the four distance-related criteria (shaded grey) used to calculate the index of publicly accessible greenspace accessibility.

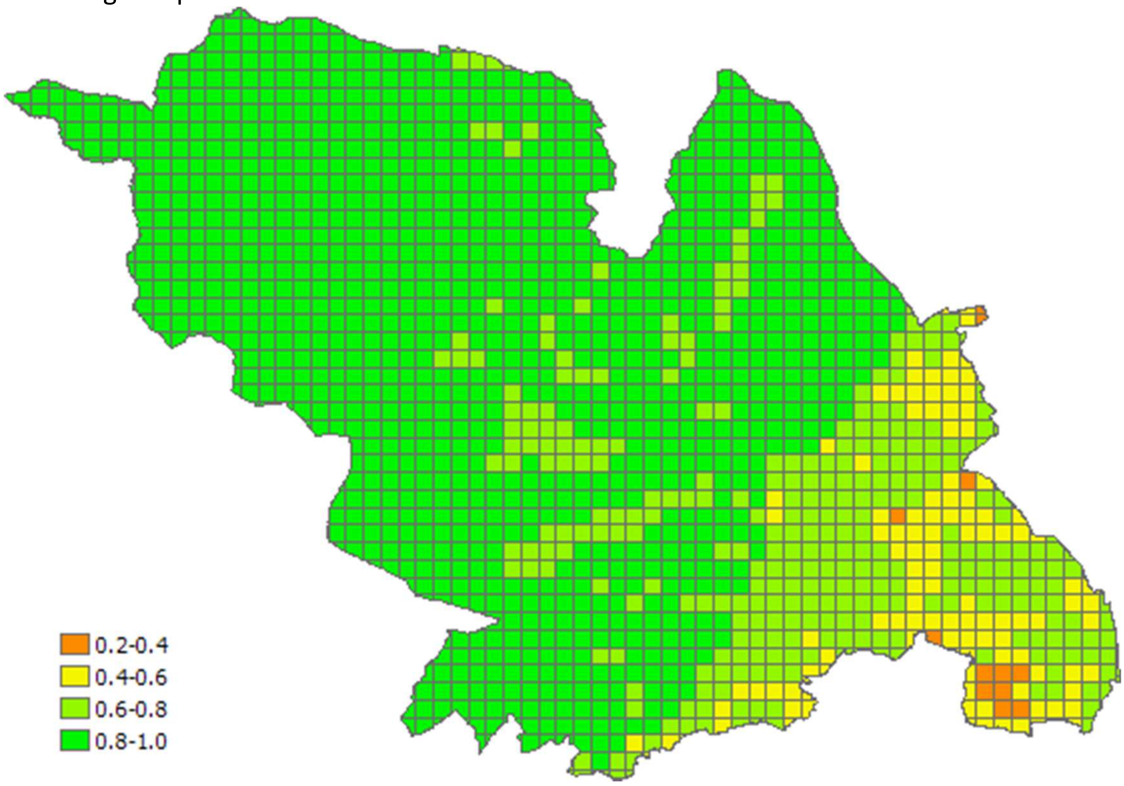
criteria listed in Section 7.1.2 (Figure 7.1, Figure 7.2d). The 10km buffer of the moorlands is thus very apparent on the supply maps.

Some suburban areas are also poorly supplied by small local greenspaces (Figure 7.2a), and this is the major cause of local variation in provision in the eastern half of the study area (Figure 7.3a). Only small parts of Sheffield are not within easy reach of intermediate sized greenspaces (Figure 7.2b,c). However, the regions not meeting the two intermediate distance criteria are also not within 10km of the moorlands, causing overall supply of the ecosystem service to be very low in some parts of southeastern Sheffield (Figure 7.3a).

The Output Area map of supply, which reflects population densities, shows a clear northwest/southeast divide (Figure 7.3b). Residents to the northwest of the urban core mostly have very good access to greenspace. However, residents in the southeast have only poor to moderate access.

In the case of the supply of this ecosystem service, there is a very similar pattern observed across the study area regardless of the spatial units used (Figure 7.3). This is because the moorland areas, where Output Areas and Historic Environment Character

a. 500m grid squares



b. Output areas

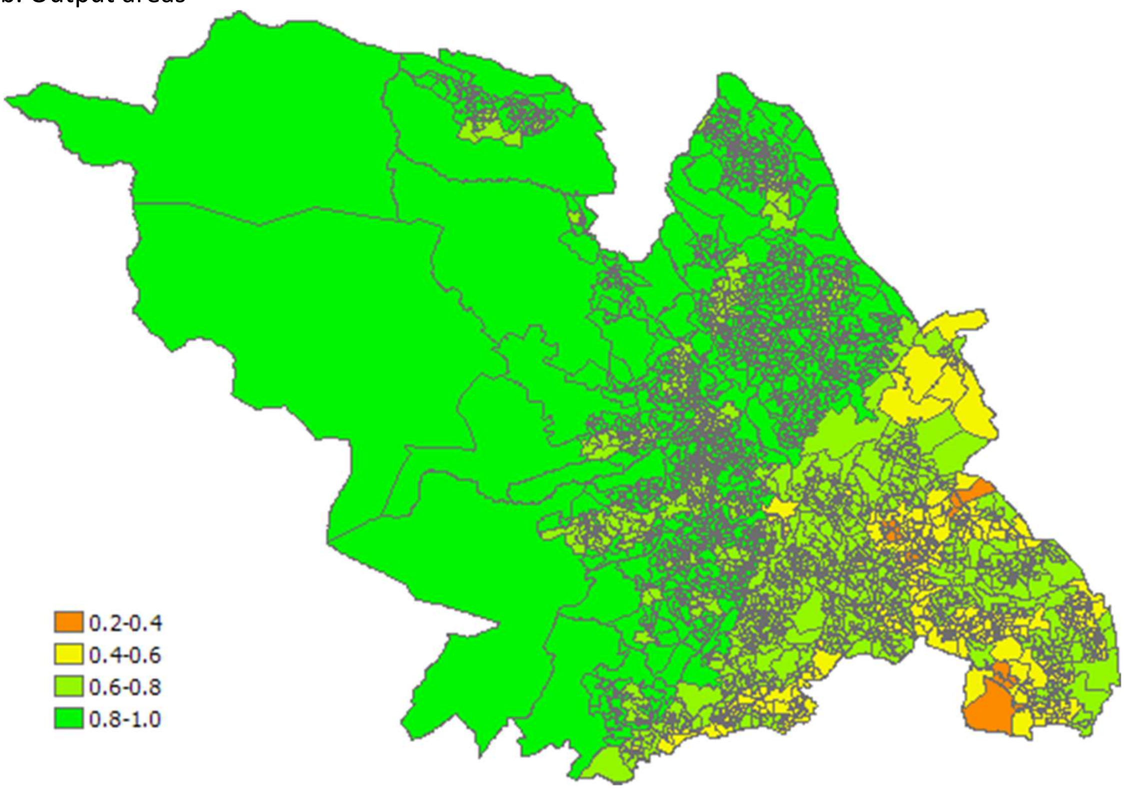


Figure 7.3. Index of supply of opportunities for cultural ecosystem services from greenspace (i.e. proportion of area consisting of publicly accessible greenspace), aggregated over different spatial units of analysis.

c. Historic Environment Character areas

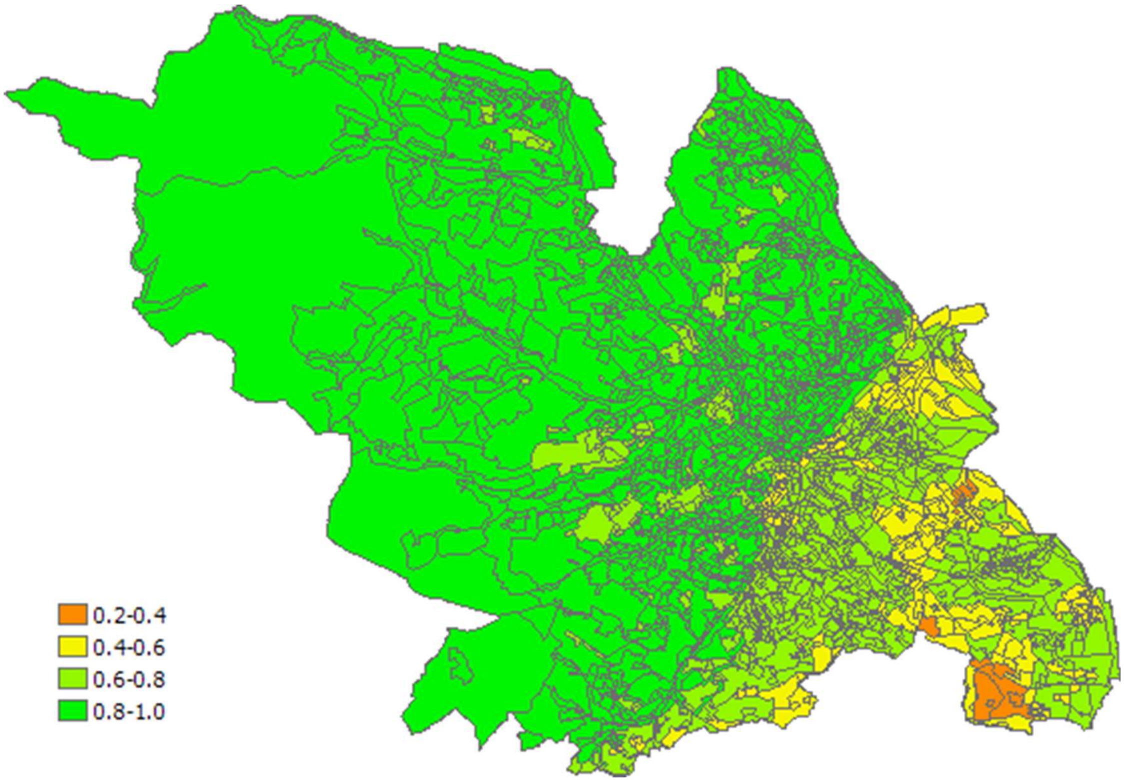


Figure 7.3 continued.

area polygons are large, have uniformly high levels of supply. There is no smaller scale detail that is lost within these large polygons compared to the 500m grid squares. Small-scale variation in supply only occurs further east, where polygons are smaller.

7.4. Discussion

This analysis has found a spatial division in the accessibility of opportunities for cultural ecosystem services in Sheffield, whereby residents in the northwest have better access than those in the southeast. This work complements a previous case study of access to greenspace in Sheffield by Barbosa et al. (2007). Barbosa et al. used maps of unique addresses and the transport network to determine the distance from each household in urban Sheffield to both the nearest greenspace and the nearest municipal park. The calculated distance was then compared to the English Nature recommendation that no household should be more than 300m (i.e. a five minute walk) away from their nearest greeenspace (similar to the first recommendation used in this study), and a similar European Environment Agency recommendation suggesting 900m as the maximum distance. This is in contrast to the present study, which uses as-the-crow-flies distances and four greenspace access criteria relating to greenspaces of different sizes.

Barbosa et al. found that only 36.5% of households in the urbanised area of Sheffield are within 300m of their nearest greenspace. This is in contrast to the finding of this study that 79.9% of Sheffield is within 300m of a 2ha greenspace (Section 7.3.1, Figure 7.2). In addition to the use of different spatial boundaries (the present study also includes the more rural parts of Sheffield) and different methods to measure distance, this may partly be attributable to differences in methods used to identify areas of publicly accessible greenspace.

The implication of these differences is that estimations of the supply of cultural ecosystem services from greenspace (Figure 7.3) may be over-estimates. Thus Figure 7.3b, which shows the supply of opportunities for cultural ecosystem services from greenspace at the Output Area scale, may suggest that more people have good access to cultural ecosystem service opportunities than is actually the case.

The differences between the results of the two studies also highlights the importance of ensuring that public greenspaces are well connected to the local transport network so that they are accessible both to pedestrians and to road users. The disparity suggests that, at present, routes from residences to public greenspaces are often not very direct; or in other words, that the shortest route via roads and paths is often considerably greater than the direct distances.

The results here suggest that, at present, routes from residences to public greenspaces are often not very direct. Nevertheless, 95.6% of Sheffield households meet the European Environment Agency's less conservative recommendation of a maximum distance of 900m to the nearest greenspace when pedestrian routes are followed (Barbosa et al. 2007).

Barbosa et al. (2007) also addressed the question of whether private gardens act as a replacement for publicly accessible greenspace in areas where provision of the latter is low. A negative relationship between areas of public and private greenspace was indeed found ($r_s = -0.36, p < 0.001$); but decreases in the area of public greenspace are met by smaller increases in the area of private gardens, meaning that the total area available is still less than in areas with lots of publicly accessible greenspace. Moreover, the two types of greenspace cannot be considered as substitutes for one another, as they are used for different practical and social purposes and activities (Kellett 1982, Barbosa et al. 2007). Furthermore, there is evidence that the use of public parks is in fact positively correlated with access to a private garden (Handley et al. 2003). It appears therefore that access to public greenspace is important regardless of whether or not a private garden is available to a household.

Section 7.2.2 stated that the index developed in this chapter quantifies only opportunities, rather than actual use, as a major limitation. The use of urban greenspaces is in part determined by its design and condition. It has been found that signs of neglect or poor maintenance, such as littering or vandalism, can discourage visitation (Coles and Bussey 2000). It is also important that parks have an open design, such that visitors are at no point “enclosed” by trees or thick vegetation (Jorgensen et al. 2002). Both signs of neglect and enclosed designs can cause greenspace users to feel unsafe and consequently to visit less frequently (Coles and Bussey 2000, Jorgensen et al. 2002, Barbosa et al. 2007). In addition, there is evidence that the actual benefits to individuals who do use greenspace depends on the perception of the greenspace quality; for example, a study has found that perceived biodiversity in city parks is positively related to psychological benefits (Fuller et al. 2007).

In conclusion, the index used in this chapter has found the provision of accessible greenspace for cultural ecosystem services to be good in some parts of the city, but poor in others. Significantly, it is likely that factors not included in this index – for example. how far residents have to travel on the ground to get to a greenspace, or the effect of the design and conditions of individual greenspaces on visitation frequencies – would show the present results to be overestimations. Thus although Sheffield is cited as being a green city (Beer 2003), the availability of cultural ecosystem service-providing greenspaces to residents is far from universal.

8. Habitat for flora and fauna

8.1. Introduction

Biodiversity is thought to be critical to the production of many ecosystem services, and to their continued production in the face of pressures on the environment (Walker 1992, Naeem 1998, Folke et al. 2004, Millennium Ecosystem Assessment 2005a, Balvanera et al. 2006). The evidence supporting this statement was reviewed in Section 1.2.6. To summarise that evidence, each species contributes to a variety of different environmental processes and functions; for example, some species contribute to pollination, others to decomposition and nutrient cycling, and others to the primary productivity of an ecosystem (Naeem 1998, Schlesinger et al. 2008). It is these functions that underpin the production of ecosystem services (Haines-Young and Potschin 2008), as shown in Figure 1.1.

Some species perform similar functions, and if there are multiple species in an ecosystem performing the same function, then that function – and any ecosystem services it produces – will continue if one of those species becomes locally extinct (Naeem 1998, Folke et al. 2004). Once this redundancy is lost, however, further pressures causing loss of biodiversity may cause ecosystem services to be lost (Naeem 1998, Folke et al. 2004).

Rates of local extinctions are exceptionally high in urban environments, due to habitat loss and fragmentation and other anthropogenic impacts (Ehrenfeld 2004, McKinney 2006, Heckmann et al. 2008, Schlesinger et al. 2008, Evans et al. 2009). This is not to say that biodiversity is necessarily low in urban areas, as there are some species that thrive in association with humans; and the fact that these species are so common in developed areas means that biota can be unexpectedly similar between cities (McKinney 2006, Pickett et al. 2009). Nevertheless, changes to the composition of the biological community means that ecosystem functioning becomes liable to change, with the potential to degrade ecosystem service provision (Folke et al. 2004, McKinney 2006).

There is evidence that biodiversity also has direct value to humans through provision of psychological benefits, similar to those provided by being in a natural setting (Section 7.1.1). Fuller et al. (2007) surveyed users of a number of parks in Sheffield and found that the species richness of plants and birds in the park, and also the diversity of habitat types, were associated with self-reported psychological benefits. These benefits include opportunity for reflection, and senses of identity and continuity of identity across time established through association with a particular greenspace (Fuller et al. 2007). Biodiversity is also valued by some in and of itself, irrespective of its human or ecological benefits; and in some situations particular taxa are valued for a specific instrumental use such as food or medicine production or recreation attraction (Sandler 2010).

This chapter introduces the index used in this thesis to spatially quantify the provision of habitat for biodiversity in the study area, and discusses spatial patterns across the study area in relation to evidence about the value of biodiversity.

8.1.1. Generation of ecosystem services

Figure 8.1 shows a conceptual model of how the provision of habitat diversity in urban areas, through variety in the land cover matrix, results in benefits to humans. The dotted lines indicate that the exact nature of the relationships is uncertain. Nevertheless, the evidence presented in Section 1.2.6 shows that, in general, changes to the matrix of natural land covers (in terms of amount, diversity and layout) resulting from urbanisation lead to a change in biological community composition, which can in turn affect ecosystem service production in an unpredictable way.

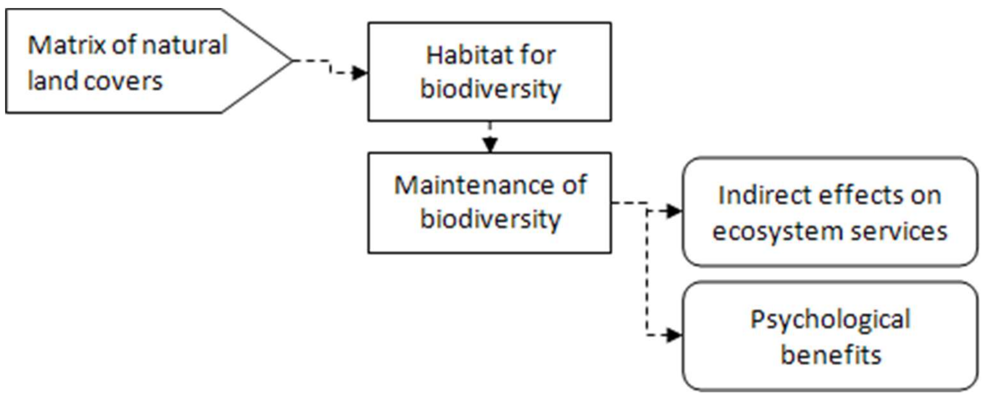


Figure 8.1. Conceptual representation of the ecosystem services provided by habitat for flora and fauna. Input conditions are represented by pentagrams; ecosystem processes by rectangles; and socioeconomic benefits by ellipses. The dotted lines

represent uncertainty in the relationships.

It is not the ecosystem services themselves there are modelled here. Rather, the index described in Section 8.2 attempts to quantify the ability of the land cover matrix to provide habitat for biodiversity. This is done by taking into consideration the amount of natural habitat, the diversity of habitat types, and the size of habitat patches.

8.2. Index overview

Ideally, biodiversity would be assessed directly from observational records. Unfortunately, consistent and reliable records of biodiversity at the scale required for this study are not available for the study area. However, the fact that habitat homogenisation is associated with loss of biodiversity suggests that metrics describing the degree of urbanisation and the variety of remnant natural habitats might provide an indication of the potential of an urban area to support ecosystem service-providing biodiversity (Whitford et al. 2001, Tratalos et al. 2007, Schlesinger et al. 2008, Sanford et al. 2009). For example, Whitford et al. (2001) used an index combining the percent of an area covered by natural habitat and the Shannon diversity index of habitat types as an indicator of biodiversity potential, while Tratalos et al. (2007) used selection of natural habitat area and cover measures. A land cover-based approach similar to that of Whitford et al. (2001) is developed here.

Urban development has multiple consequences for the natural vegetation that is being replaced, as well as other types of biodiversity that use vegetation as habitat. As urbanising areas are covered with impermeable surfaces, the natural vegetation is split into smaller patches that may have low connectivity (Bolger et al. 2000, Crooks 2002). Greenspace management by gardeners and landscape architects means that the remaining natural vegetation may also be converted to different types of land cover, changing the availability of some types of habitat (McKinney 2006).

Multiple metrics of the biodiversity potential of the landscape were chosen to reflect different components of these complex effects, and then combined to produce a single indicator measure. The first metric is the proportion of the area of interest that comprises natural land covers. This metric represents the well known species-area relationship, which has been observed in urban areas (Clergeau et al. 2001, McKinney 2008). The second metric is the Shannon diversity index, which represents the diversity of habitat types and evenness in the amount of habitat types. Habitat diversity has been observed to correlate with species diversity (Norderhaug et al. 2000, Pino et al. 2000). The Shannon diversity index in particular is sensitive to the presence of rare habitat types, which is important because different habitats support different species, and thus rare habitat types may be the last remaining areas capable of support some species. It is for this reason that the Shannon diversity index has been recommended as a metric of landscape diversity (Nagendra 2002). The final metric is the natural land cover correlation length, which is used to represent the connectivity of individual patches of natural land cover through their average size. High connectivity (i.e. large patches) is important because habitat fragmentation is thought to cause the loss of some species from urban areas (Bolger et al. 2000, Clergeau et al. 2001, Crooks 2002, McKinney 2008, Pickett et al. 2009).

The first two metrics are used following Whitford et al. (2001); but the third, and the method of metric combination, do not appear to have been used in this context previously. The individual metrics all have numerical ranges of zero to one. The final index was computed as the sum of the three metrics, as no empirical evidence about the relative importance of the metrics to biodiversity could be found. Complete details of the metrics and index formulation can be found in Appendix A.5.

8.2.1. Index limitations

As has already been mentioned, this index quantifies potential biodiversity, because it was not possible to obtain data on observed biodiversity. This fact needs to be borne in mind when interpreting the results, especially as there has been no opportunity to ground-truth the association between the metrics and levels of biodiversity in the study area. Nevertheless, there is empirical evidence from other locations that the amount, variety and connectivity of natural land cover in urban areas are each related to species richness.

8.3. Results

8.3.1. Model output

Maps of each of the metrics are shown for 500m grid squares in Figure 8.2. Each metric shows a considerably different pattern across the study area. The proportion of natural land cover (Figure 8.2a) is, as expected, very high in the moorland and agricultural areas and increasingly low towards the centre of the developed urban area.

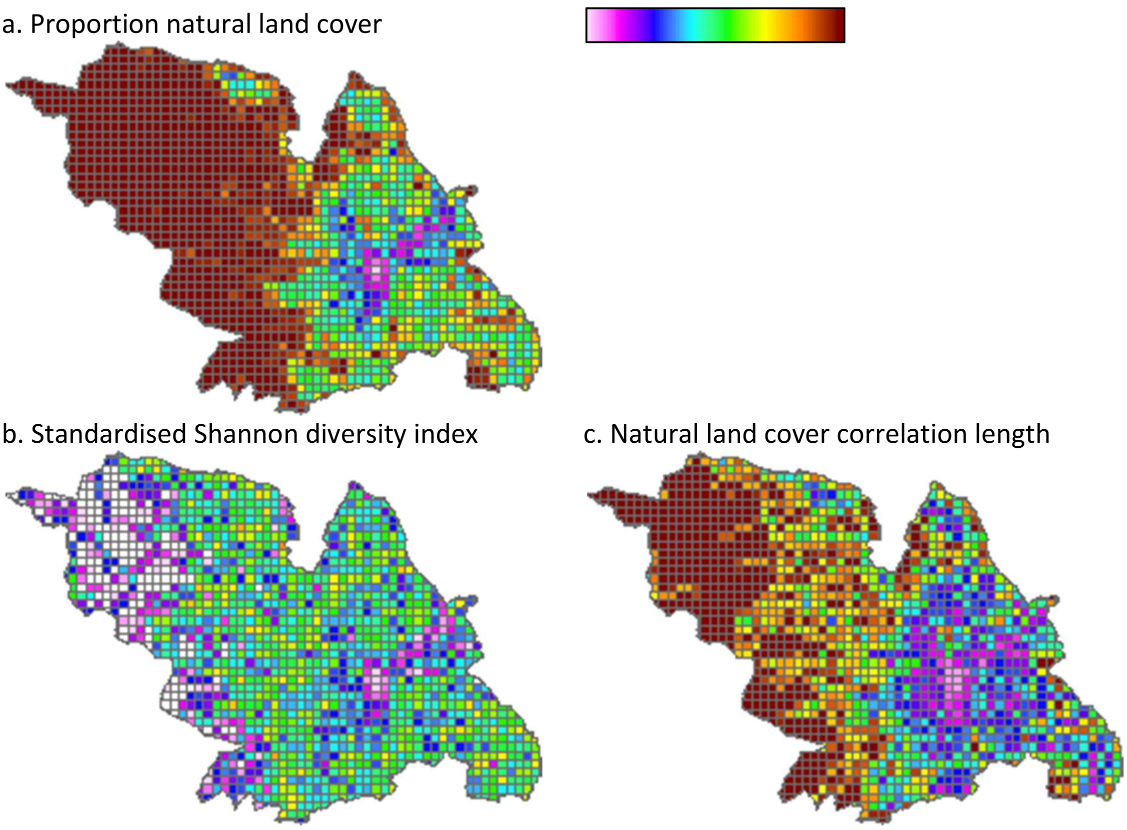


Figure 8.2. Habitat index metrics analysed over 500m grid squares. The maps each use the pseudo-continuous linear colour ramp shown in the upper right. Range is 0-1 for (a) and (c), and 0-0.82 for (b).

The natural land cover correlation lengths (Figure 8.2c) are also high in the moors and low in the city centre; but values in the agricultural belt are intermediate and more variable due to the existence of some small and some large areas of different land cover types, for example around farmsteads. There are also some regions in the urbanised centre that have high values for both this metric and the proportion of natural land cover.

The standardised Shannon diversity index shows a very different pattern (Figure 8.2b). Values are very low in the moorland, due to the relatively monotonous landscape. In the urban area, with the exception of the highly industrialised core, values are high. This shows that the urban matrix can provide a variety of land cover types, and could therefore in theory support organisms occupying many different ecological niches. However, values of the Shannon diversity index are maximal in the agricultural areas, indicating that as well as providing a variety of habitat types, the habitat distribution here is also quite even.

8.3.2. Ecosystem service maps

The index of habitat provision, i.e. the average value of the three metrics shown in Figure 8.2, is shown for 500m grid squares in Map 20 (all maps shown in Appendix D). The central agricultural area of the study region has the greatest index values, despite having only intermediate values for the Shannon diversity index and correlation length (Figure 8.2b,c). Habitat provision index values in the moorlands are on average slightly lower, despite high proportions of natural land cover and correlation lengths, because of the low values of the Shannon diversity index. Values of the habitat provision index remain moderately high well into the suburban area, only becoming especially low toward the urban core. This is the consequence of decreasing values of all three metrics.

Analysis across the other spatial units, however, reveals a different pattern. The habitat provision index is shown for Output Areas in Map 21, and for Historic Environment Character areas in Map 22. These figures suggest that the potential provision of habitat for biodiversity is slightly higher in the moorland than in the agricultural belt. This is because the Shannon diversity index is higher in the large polygons in this area, as they contain multiple habitat types. In comparison, inspection of the land cover map shows that most of the 500m grid squares include only one or a few types of land cover and thus have low Shannon diversity (Figure 8.2b). In the urbanised area, however, the pattern is the same as for the 500m grid squares, with a decrease in index values towards the urban centre.

As with the other ecosystem services, there is variation in index values between Output Areas of similar sizes (Map 21), indicating that urban design again has an influence. There is also high variability between Historic Environment Character area polygons with the same broad land use (Map 3, Map 22), which would suggest that the different land cover compositions of urban land uses determine the relative biodiversity found in, for example, parks versus playing fields.

8.4. Discussion

The index of habitat provision used here is based on several well-known and common ecological patterns, namely the species-area relationship, the positive relationship between landscape-scale diversity and species richness, and an increase in population persistence with greater habitat connectivity. As these patterns have been observed in many urban areas (see references in Section 8.2), it is likely that they also occur to some extent in Sheffield.

However, this index assumes that species occurring in all types of habitat are equally valuable, which is not necessarily the case (Hampicke 1994, Lodge and Shrader-Frechette 2003). Three types of value can be attributed to species: instrumental value, which derives from its usefulness to humans and includes its recreational and economic values; ecological value, which describes its value to the functioning of the ecosystems of which it is a part; and intrinsic value, which is the value inherent in a species' existence regardless of its human or ecological function (Sandler 2010).

Some species have high instrumental value, for example if they attract recreation such as hiking or fishing; or if they can used for food, medicine or other marketable resource such as timber (Hampicke 1994, Sandler 2010). Within the study area, species found in agricultural and plantation areas have high instrumental value for their food and wood production respectively. The Peak District, part of which lies within Sheffield's boundaries, is Britain's busiest National Park and is used for both formal and informal recreation (Peak District National Park Authority 2003). Some people visit specifically to enjoy the wildlife and landscapes, indicating that the species in this area have high instrumental value (Peak District National Park Authority 2003).

High ecological value results when a species is critical to the continued functioning, organisation and diversity of an ecosystem, as seen in "keystone" species (Mills et al. 1993). Examples of keystone species include prey species that are preferred by predators and prevent more sensitive species from being predated to (local) extinction; plants that are important to the survival of pollinator or seed disperser populations; and species such as the North American beaver that "engineer" their environment, in this case by altering hydrology and productivity by building dams (Mills et al. 1993). Examples of species with high ecological value in the Sheffield region of the Peak District National Park include oak and birch, which comprise woodlands that support high-diversity communities associated with ancient woodlands; and heather in the moorlands, which is a landscape that supports a number of rare species of birds and butterflies (Peak District National Park Authority, no date). Thus, these species that form the basis of unique habitat could be attributed with high ecological value.

Intrinsic values can be thought of as arising either dependent upon, or independently of, any valuer. In the former case, "[species] may not be valuable *in* themselves but they may certainly be valued *for* themselves. According to this … account, value is … humanly conferred, but not necessarily homocentric" (Callicott 1989, cited in Sandler 2010). In contrast, species can be thought of as having intrinsic objective value because they contain biological information that is valuable regardless of whether or not anybody is doing the valuing (Sandler 2010). In either case, the value varies between species (Hampicke 1994, Sandler 2010). Unfortunately, there have been no studies investigating intrinsic species values in Sheffield or the Peak District. It is commonly found, however, that people think of native species as being especially valuable, especially when in their native systems (Lodge and Shrader-Frechette 2003, Sandler 2010). The Peak District moorlands are an example of a long-established habitat that supports a number of the UK's native species; thus this habitat type and the species it supports might be seen as more intrinsically valuable than the more recently established and less "natural" farmland and urban greenspace habitats.

The results of this index indicate that the the moorland and agricultural areas have a similar ability to provide habitat for biodiversity. However, the discussion developed here suggests that the moorlands might support species of greater value than other land cover types, especially if intrinsic values are considered. Nevertheless, in the absence of biodiversity records and valuation exercises to determine the relative worth of species, this index is able to show where species richness is predicted to be high as a result of commonly observed ecological patterns.

9. Bringing the models together

9.1. Introduction

The previous six chapters have each presented and discussed the results of a model of a single ecosystem service, demonstrating the potential to model ecosystem service production at the neighbourhood scale from existing data sets and methods (research aims 1-3 in Section 1.4). These chapters have also discussed the nature of the production and supply of the six ecosystem services, and why they are important contributors to human welfare (see summary in Table 9.1).

Although the results of the six ecosystem service models each provide a platform for individual analysis, ecosystem services do not occur singly or independently (Rodríguez et al. 2006). A single ecosystem can produce a wide variety of ecosystem services, which may or may not show similar spatial patterns (Chan et al. 2006, Naidoo et al. 2008, Nelson et al. 2009). It is important to understand how, and the extent to which, spatial patterns differ between ecosystem services: knowing whether all ecosystem services can be produced in the same place, or whether decisions about trade-offs are necessary, is critical to the success of any management interventions (Anderson et al. 2009). A tight correlation between multiple ecosystem services indicates that the same environmental and urban factors are controlling ecosystem service generation, making planning for ecosystem services relatively easy. In contrast, weak relationships, such that ecosystem services have 'hotspots' in different places, indicate that a variety of factors are at play and therefore make management more complicated (Anderson et al. 2009).

Furthermore, ecosystem services sometimes interact. For example, reduction of air pollution by vegetation might reduce inhibition of photosynthesis due to high levels of nitrogen dioxide (Matthias et al. 2006) and thus increase carbon storage in vegetation; and also increase long-wave radiative losses of heat, thereby lessening the heat island effect (Oke 1987).

Although it was not feasible to model directly such interactive effects within this project, the models for the six individual ecosystem services that have been included

Table 9.1. Summaries of the nature of the production, supplies and benefits of ecosystem services modelled in this thesis. Elements in italics could not be modelled within the scope of this project.

Ecosystem service	Production	Supply	Benefits (examples)
Air pollution reduction	Pollutant deposition	*Reduction of pollutant concentration*	*Reduced asthma, stroke, etc., reduced fouling*
Heat island mitigation	Surface temperature reduction	Surface temperature reduction	*Reduced heat exposure morbidity and mortality, lower air conditioning costs*
Storm runoff reduction	In situ runoff reduction	*Reduced flooding*	*Improved water quality, reduced illness and costs from flooding*
Carbon storage	Carbon storage	*Climate change risk reduction*	*Many*
Cultural service opportunities	Provision of urban greenspace	Accessibility *(and suitability)* of urban greenspace	*Improved mood, improved attention, increased lifespan*
Habitat provision	Biodiversity maintenance	*N/A*	*Maintenance of ecosystem services of direct benefit to humans*

here nevertheless provide an opportunity to look at how they covary. Understanding spatial similarities and differences between the ecosystem services is also a pre-requisite to the further applications of the models in this thesis, which involve combining the model output with further datasets (Chapter 10 and Chapter 11). Developing such an understanding is the focus of this chapter.

This chapter documents the covariation between levels of production of the six ecosystem services, and discusses various aspects of the model results in relation to each other. Three approaches are used to understand the relationships between ecosystem services: correlation, spatial autocorrelation analysis and hotspot analysis. The correlation analysis begins with examination of pairwise scatterplots of ecosystem service production, then goes on to more formal statistical analysis to quantify the degree to which pairs of ecosystem services covary.

Spatial autocorrelation analysis shows the spatial structure of individual ecosystem services, thus facilitating a comparison of the spatial structures between services. Recognising the existence of spatial autocorrelation is also important when implementing statistical tests, as it can change the degrees of freedom (Section 9.2.1).

Hotspot analysis is a complementary approach to understanding spatial relationships, which investigates the extent to which areas providing high levels of one ecosystem service overlap with those providing high levels of another. Hotspot analysis is valuable in terms of management, as it facilitates identification of priority areas for conservation (Egoh et al. 2008); whereas correlations can be used to quantify the extent to which ecosystem services tend to co-occur across a wider area and can thus be conserved together (Chan et al. 2006, Naidoo et al. 2008). As mapping the supply of ecosystem services has only been possible for two of the models, the analyses here focus exclusively on ecosystem service production.

The chapter concludes with a discussion of themes arising from the model results presented in the previous six chapters as well as the analyses described in this chapter. Before these analyses are presented, however, it is necessary to discuss the approach taken to performing statistics on the datasets produced by the ecosystem service models.

9.2. Statistical approaches

The datasets produced as output from the ecosystem service models have two properties that must be taken into consideration during statistical testing, namely irregular frequency distributions, and spatial structure. Furthermore, many of the analyses performed in this chapter, Chapter 10 and Chapter 11 include multiple variables both as predictors and responses. Ideally, therefore, non-parametric spatial multivariate statistics would be used. However, suitable spatial and multivariate statistical tests could not be identified, and so series of non-parametric tests were used. This section discusses the methods used to take spatial autocorrelation and the use of series of tests into account.

9.2.1. Spatial autocorrelation

The type of investigation undertaken by this project is inherently spatial. Spatial data are frequently spatially autocorrelated: in other words, data from points close to each other are more similar than data from distant points (Rangel et al. 2006). Such patterns are clear in many of the maps displayed in Appendix D; this chapter also documents a statistical confirmation of the spatial structure. Spatial autocorrelation causes problems for statistical methods such as correlation, but the underlying structure can also be of interest in itself. Therefore, prior to attempting any statistical analysis of the model results, it is necessary to address the issue of spatial autocorrelation.

Spatial autocorrelation causes a form of statistical pseudoreplication: spatially autocorrelated data points are not completely independent and therefore do not constitute a full degree of freedom in statistical analyses (Currie 2007). Failure to account for a reduction in the effective degrees of freedom can cause inference of statistical significance where there is none, i.e. Type I errors, thereby causing problems for interpretation of statistical results (Diniz-Filho et al. 2003, Currie 2007).

Spatial autocorrelation may be caused by exogenous factors (i.e. spatially structured factors external to the object of study; for example the effects of temperature gradients on species richness) or endogenous factors (factors arising from the object of study, such as population dynamics causing nearby populations to be more similar than distant populations) (Currie 2007). It is exogenous spatial autocorrelation that is relevant here: spatial structure is obvious in the land cover, land use and soil maps that are the key spatial inputs to the models in this thesis (see Map 2, Map 3 and Map 4; all maps shown in Appendix D). Furthermore, these maps are to some extent confounded. Land categorised by the land use map as unenclosed moorland, for example, is covered by rough grassland, heath and bog land covers, and occurs over particular peaty soil types. In contrast, land used for industry is covered by very high proportions of manmade surfaces and buildings, and is present chiefly over a soil type peculiar to the land near the rivers in central Sheffield. Thus there is a degree of spatial dependence in the model results in that, to some extent, the land cover composition in one area would be less likely to occur in another area that had a different land use or soil type.

Exogenous spatial autocorrelation is (in comparison to endogenous spatial autocorrelation) often easy to deal with in statistical analyses, through implementation of algorithms to correct the number of degrees of freedom and thus produce a more reliable probability value (Rangel et al. 2006, Currie 2007). However, not all statistical tests have ‘spatial’ equivalents, and thus it is not always possible to determine statistical significance.

Rank correlations are used extensively in this and subsequent chapters. Algorithms exist to modify the degrees of freedom to take spatial autocorrelation into account when determining the statistical significance of correlation coefficients (Clifford et al. 1989, Dutilleul et al. 1993, Rangel et al. 2006). However, multivariate normality is an assumption of these algorithms (Clifford et al. 1989, Dutilleul et al. 1993). There is at present no way to correct for spatial autocorrelation in ranked data from non-normal distributions, as confirmed by Eigenbrod et al. (2010), who used rank correlations with non-normally distributed spatial data, stating that "the frequency distributions of the data limit options for controlling for spatial autocorrelation in residuals". Therefore it is necessary to use non-spatial statistics and adopt a different approach to interpretation.

Some authors have opted to ignore spatial autocorrelation and simply present the results of non-spatial statistical tests. This approach was taken by Chan et al. (2006), who made no mention of spatial autocorrelation in their pairwise correlation of ecosystem services in California. Similarly, test statistics can be presented without an attempt to infer statistical significance (e.g. Eigenbrod et al. 2010). Naidoo et al. (2008) do not mention statistical significance when presenting correlation coefficients for ecosystem services at the global scale, and Anderson et al. (2009) also do not mention spatial autocorrelation in the hotspot analysis used as a model in this chapter. In either case, the interpretation of results must be undertaken very cautiously.

An alternative approach is to reduce the degrees of freedom by a factor that seems sensible, in order to make inference more conservative. The complex pattern of spatial autocorrelation present in these datasets, and the fact that the sizes of units of analysis (i.e. Output Areas, Historic Environment Character areas) vary across the study area mean that it is difficult to determine what a sensible factor of reduction might be (Fortin and Dale 2005). Fortin & Dale (2005) cite a study in which an α value of 0.01 was used, instead of the usual 0.05, to allow for spatial autocorrelation. It was however not possible to compare the degree of spatial autocorrelation in the cited study to that present here. Therefore I have chosen to adopt an even more conservative approach by using an α value of 0.001.

Fortunately, there is no risk in the present analyses of spurious inference of significance to the wrong variables. This is because spatial patterns of land cover, land use and soils are known to be the sole drivers of spatial autocorrelation. This is in contrast to spatial studies attempting to infer correlations between empirically determined variables, in which other environmental gradients that have not been included in statistical modelling but that are correlated with predictor variables might be causing erroneous inference. This fact could be seen to reduce the importance of the consequences of spatial autocorrelation. Nevertheless, interpretation and discussion of results throughout this thesis will bear in mind that the inference of statistical significance may not be robust.

9.2.2. Bonferroni corrections

The Bonferroni correction is used to correct for the increased probability Type I errors (false positives) occurring when multiple statistical tests are performed on the same data. The Bonferroni correction should definitely be used when the significance of a clearly delineated suite of statistical tests performed on the same data is of interest (Perneger 1998, Fortin and Dale 2005). An example of this is inference of statistical significance of spatial autocorrelation correlograms, as presented in Section 9.4.1. However, the usage of Bonferroni corrections in other contexts is less clear, and several authors have commented on its correct and incorrect usage (Perneger 1998, Cabin and Mitchell 2000, Moran 2003).

The Bonferroni correction is correctly used in the testing of the universal null hypothesis, i.e. testing whether the null hypothesis can be rejected across a suite of tests, but then does not indicate whether individual tests are significant (Perneger 1998). Perneger (1998) suggests that another "situation in which Bonferroni adjustments may be acceptable is when searching for significant associations without pre-established hypotheses"; yet the correction is not universally used in such situations (Cabin and Mitchell 2000).

Some of the problems that have been pointed out about using Bonferroni corrections are as follows. Firstly, there is the logical issue that the inference of significance depends on which other tests are being performed, and there is no clear guidance on which other tests to include in the adjustment – e.g. across a table, a paper, a study (Perneger 1998, Cabin and Mitchell 2000, Moran 2003). Secondly, the correction can be seen as "punishing" a researcher for performing more work (i.e. having more tests to do) by increasing false negatives (Type II errors) at the same time as decreasing the risk of false positives (Type I errors) (Perneger 1998, Cabin and Mitchell 2000, Moran 2003). Finally, the correction ignores the number of tests in a suite that are significant. If, for example, a single test in a large table had a low probability value (for example 0.025 in a suite of ten tests), it is likely to be a false positive of the kind that the Bonferroni correction attempts to avoid; however, if many of the tests had probability values of this order of magnitude, it seems reasonable to believe that real patterns are being observed (Moran 2003).

On the other hand, ignoring increased risks of Type I errors does not seem a sensible approach. Moran (2003) therefore suggests the reporting of exact probability values, and making interpretations based on "experimental design, power analyses, differences between control and treatment groups, and basic logic". Cabin & Mitchell (2000), make different recommendations: that the decision over whether to use corrections be recognised as subjective; to use common sense in interpretation; and to state the changes that would be made in the interpretation if an alternative approach to correction was used.

In light of these discussions, Bonferroni corrections are used in this thesis in three situations. The first, as already mentioned, is in correlograms. The second situation is in post hoc multiple comparison testing in the results of analyses of variance, and the third is where several variables have been used in multiple correlations without a priori expectations about where the significant relationships may lie. These latter situations match the valid usage criterion of Perneger (1998) quoted above. In order to avoid over-reliance on a single probability value, however, the results are also considered using a non-corrected value in order to determine any differences that would be made to the interpretation; and trends in groups of results towards significance or non-significance are also noted.

9.3. Methods

9.3.1. Spatial autocorrelation – Moran's I

One of the most common statistics used to test for the presence of spatial autocorrelation within a dataset is Moran's I autocorrelation coefficient. Moran's I is used here to investigate the spatial structure of ecosystem service production.

Moran's I is an extension of Pearson's product-moment correlation coefficient, r, which measures the average degree of association between two variables (x and y) by computing their covariance as deviation from their respective means, then standardising by the product of their standard deviations (Fortin and Dale 2005):

$$r = \frac{\sum_{i=1}^{n}(x_i - \bar{x})(y_i - \bar{y})}{\sqrt{\sum_{i=1}^{n}(x_i - \bar{x})^2}\sqrt{\sum_{i=1}^{n}(y_i - \bar{y})^2} \qquad (4)$$

Moran's I instead estimates the covariance between spatially explicit pairs of data points of a single variable that are separated by a known spatial distance, or lag (Fortin and Dale 2005). Moran's I is often calculated for multiple distance classes in order to produce a correlogram, or a plot of spatial autocorrelation against distance lag. In these cases, the relevant element of the weighting matrix, w_{ij}, is assigned a value of zero or one depending on whether the pair of points falls within the distance class; although for other types of investigation w_{ij} elements can also be given other values determined by factors such as distance or connectivity (Fortin and Dale 2005, Paradis 2009).

The statistic I for a distance band d is calculated as follows (Fortin and Dale 2005):

$$I(d) = \left[\frac{1}{W(d)}\right] \frac{\sum_{i=1}^{n} \sum_{j=1}^{n} w_{ij}(d)(x_i - \bar{x})(x_j - \bar{x})}{\frac{1}{n}\sqrt{\sum_{i=1}^{n}(x_i - \bar{x})^2}}, \text{for } j \neq i \quad (5)$$

Where i and j are two sampling locations (in this case, GIS polygon centroids); x is the value of the variable of interest; $\bar{x}$ is the overall mean of x; $W(d)$ is the number of pairs of sampling locations in the distance class; n is the total number of pairs of sampling locations; and w_{ij} takes a value of one if the pair ij is within the distance band, and zero if it is not, thereby determining which location pairs are included in calculating $I(d)$.

The traditional form of Moran's I, shown in Eqn. (5), assumes that the data are normally distributed (Fortin and Dale 2005). However, this assumption does not hold for the present datasets, many of which are heavily skewed or bimodal and thus cannot be transformed. An adaptation for ranked data, suggested by Cliff and Ord (1981), is therefore used here.

The statistical significance of the observed value of I is determined by comparison with a calculated expected value and variance under the null hypothesis of zero spatial autocorrelation (Paradis 2009). In the present case, the expected observed value and variance are determined under the assumption of randomisation.

Most spatial statistics, including Moran's I, make two special assumptions: those of spatial stationarity and isotropic spatial autocorrelation (Fortin and Dale 2005, Dormann et al. 2007). Spatial stationarity refers to consistency in the way in which the dependent variable is related to the independent variables, and to itself in space; or in other words that the process under investigation is independent of absolute location over the study area (Fortin and Dale 2005, Dormann et al. 2007). A example of a non-stationary process, given by Fortin and Dale (2005), would be net primary productivity over a hypothetical continental divide: the rain shadow could cause productivity to be limited by precipitation on the side of the divide facing away from the ocean, whereas productivity on the other could be limited by temperature. Thus temperature and precipitation do not affect productivity in the same way across the whole area: location itself becomes a factor influencing how the response variable responds to the environmental variables. In the present case, however, a consistent model is applied to each of the ecosystem services across the whole study area, meaning that environmental variables cause the same effects regardless of where in the study area the ecosystem service is being modelled. Thus, there is no reason that the assumption of stationarity should be violated.

The other assumption, isotropic spatial autocorrelation, means that the processes causing spatial autocorrelation act in the same way in all directions (Fortin and Dale 2005, Dormann et al. 2007). Anisotropic spatial autocorrelation might occur, for example, in seed dispersal if there is a predominant wind direction (Fortin and Dale 2005). Again, the ecosystem service models are such that this assumption should not be violated in the present case.

A problem with computing a statistic for a single dataset over distance classes is that the same data are used in multiple classes, and the coefficients are thus not independent (Fortin and Dale 2005). Furthermore, there are no specific a priori expectations as to the distance bands that might show autocorrelation. These issues have been allowed for in interpretation of the results by adjusting the level at which probability values are considered significant according to the Bonferroni correction (Fortin and Dale 2005):

$$\alpha' = \frac{\alpha}{k} \tag{6}$$

Where $\alpha$and α' are the probability level considered before (i.e. 0.05) and after adjustment, and k is the number of distance classes. Although Bonferroni adjustments are technically concerned with table-wide significance (Perneger 1998), i.e. whether the correlogram as a whole is considered significant, this study follows Fortin and Dale (2005) in also using the corrected value in identifying individual coefficients that are significant.

A further potential issue with the Moran's I coefficient specifically is that, although the value usually ranges between 1 and -1 (indicating positive and negative spatial autocorrelation respectively), these are not mathematically stable boundaries (Rangel et al. 2006). Comparison of coefficients between datasets is not therefore reliable without standardisation by the maximum possible value of I for the given dataset (Rangel et al. 2006). However, for unknown reasons the algorithm for determining the maximum possible value of I for unranked datasets did not work correctly in tests using ranked data, and no published alternative algorithm could be found. Therefore caution must be exercised in comparing values of the I coefficient between different ecosystem services or spatial units.

Statistical calculations were performed using the package ‘spdep’ v0.4-56 (Bivand et al. 2009) for the computer program R v2.8.1 (R Development Core Team 2008). Distance class boundaries were determined using the computer program Statistical Analysis in Macroecology v3.0 (Rangel et al. 2006), which includes algorithms to determine an appropriate number of classes having similar numbers of pairs of data points in order to avoid unreliable results from having small numbers of pairs in some distance classes.

9.3.2. Correlation analysis – Spearman’s rank correlation coefficient

The ecosystem service variables show a variety of non-normal distributions. Thus, Spearman’s rank correlation coefficient, rho, is used here to quantify the degree of association between pairs of ecosystem services. The equation for Spearman’s rho is the same as that for Pearson’s r given in Eqn. (4), but where x and y give the ranks of the variables. Tied values are assigned the average rank.

It should be noted that the method used here does not take into account the area of each polygon: all polygons have equal weighting in determining the correlation coefficients, regardless of size. Not accounting for area means that each polygon is considered to represent a particular set of circumstances, one of which is size, which is not made explicit here. This means that each spatial unit, as delineated by socioeconomic conditions for Output Areas, land use/cover history for Historic Environment Character areas, and arbitrary grid lines for 500m grid squares, makes an equal contribution to the overall correlation coefficient. The analysis was also performed for 500m grid squares excluding partial squares (n = 1325) to give an overview of the area-standardised correlations at a 500m scale, but the results were very similar to those for the complete 500m grid square dataset and so are not presented.

9.3.3. Hotspot analysis

The method of hotspot analysis used here follows that of Anderson et al. (2009). The term ‘hotspot’ describes an area having a high level of a particular characteristic, although the quantitative designation is somewhat arbitrary (e.g. Anderson et al. 2009), so two thresholds are defined here: the 10% of polygons and the 25% of polygons with the highest ecosystem service values. This is done to prevent too strict a criterion from obscuring weaker patterns. In cases where there are tied levels of ecosystem service production around the threshold, all the tied polygons are included as hotspots. Polygon area is not considered in identifying hotspots, for the reasons established in Section 9.3.2; results from the 500m grid square dataset including only complete squares were again very similar to those for all 500m grid squares.

No formal statistics are attempted here, but the identified hotspots are used to develop understanding of the spatial covariance of ecosystem services in two ways. First, the hotspot overlap, i.e. the number of ecosystem services for which each polygon is a hotspot, is determined. These numbers can be used to determine the extent to which high levels of multiple ecosystem services occur in individual places (more precisely, the proportions of polygons that are hotspots for 1, 2, 3… different ecosystem services), and also to identify hotspots of multiple ecosystem services on the map. Second, the pairwise co-occurrence of ecosystem service hotspots is determined, i.e. the proportion of all polygons that are hotspots for two specific ecosystem services.

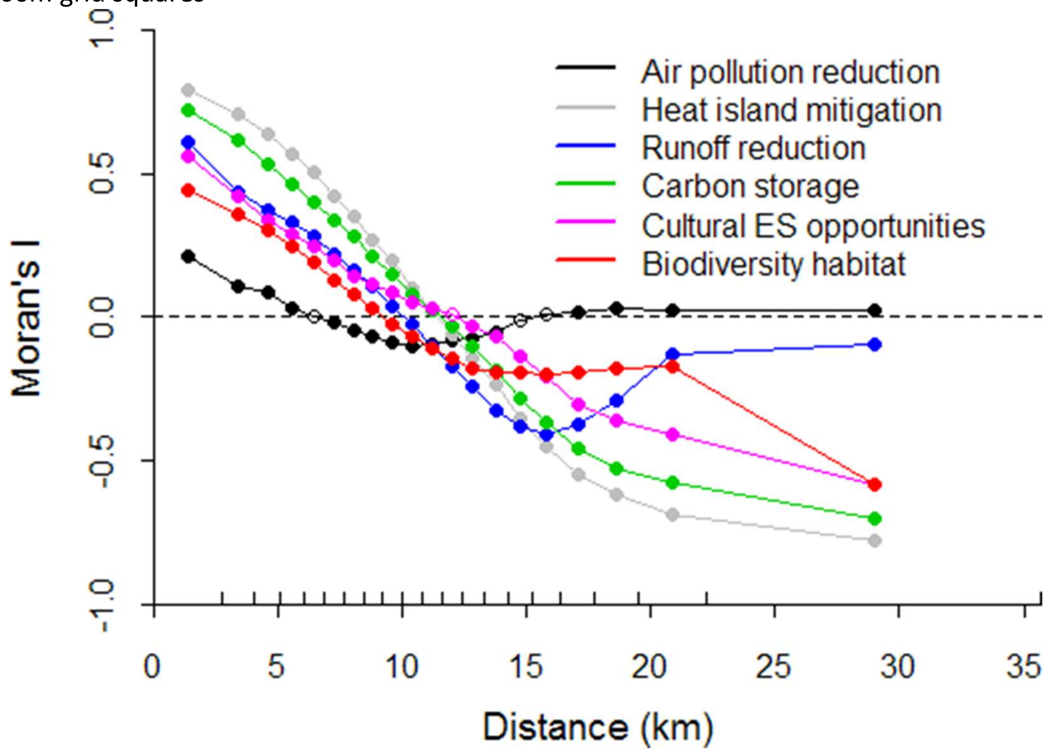
9.4. Results

9.4.1. Spatial autocorrelation

The results of the test for spatial structure, shown in the correlograms in Figure 9.1, are intriguing. There is significant autocorrelation at most distances for all ecosystem services over all spatial units, meaning that all the correlograms as a whole are statistically significant and that the model output shows spatial structure. However, although comparisons between datasets may not be reliable for the reason mentioned in Section 9.3.1 (i.e. inability to standardise the limits of the test statistic), the ecosystem services show similar patterns within spatial units, but the spatial units show different characteristic patterns.

For 500m grid squares (Figure 9.1a), with the exception of reduction of air pollution, ecosystem services are quite strongly positively autocorrelated at the shortest distances and remain positively autocorrelated up to 8-12 km, after which point they become increasingly negatively autocorrelated before levelling off – except in the case of storm water runoff reduction, which becomes less negatively autocorrelated beyond

a. 500m grid squares



b. Output Areas

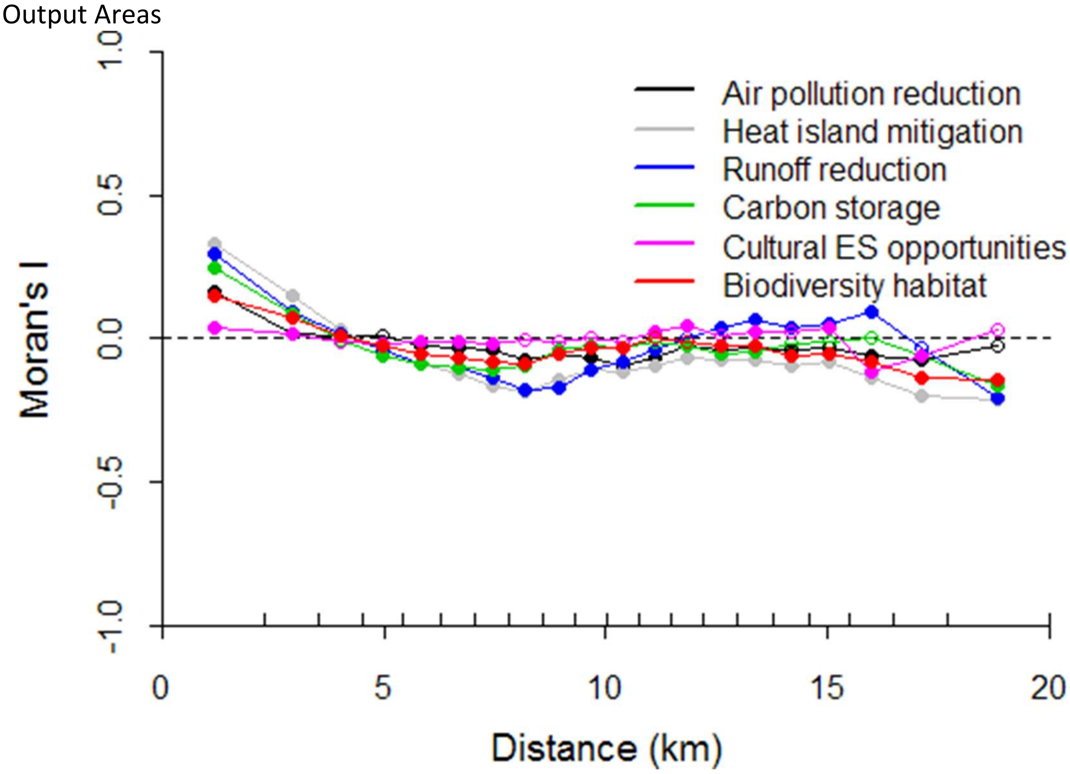


Figure 9.1. Moran's *I* correlograms of spatial autocorrelation within ecosystem services, for (a) 500m grid squares, (b) Output Areas and (c) Historic Environment Character areas. Moran's *I* is shown for each of 20-21 distance classes, using closed circles if statistically significant and open circles if not. Distance class boundaries are shown by upward ticks on the x axis. NB absolute values of *I* for different ecosystem services or spatial units may not be directly comparable.

c. Historic Environment Character areas

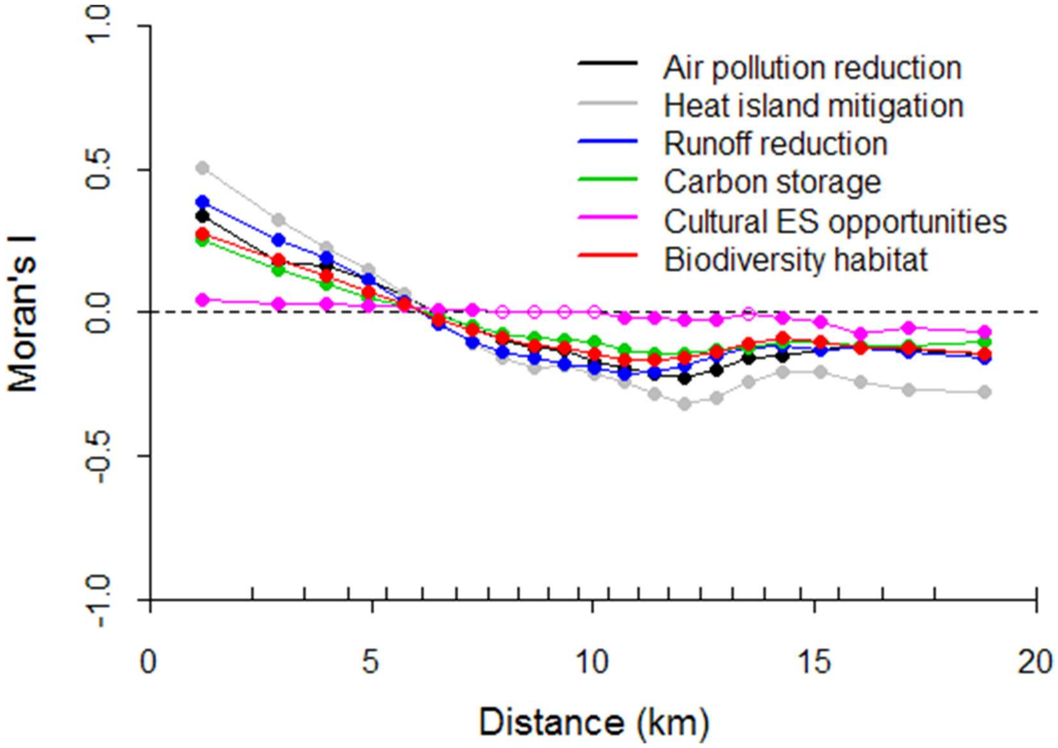


Figure 9.1 continued.

approximately 16 km. Reduction of air pollution shows a similar pattern to storm water runoff reduction, but less strongly.

In contrast to the 500m grid squares, when analysed across Output Areas all ecosystem services have a less strong pattern of changing values of I over increasing spatial lags (Figure 9.1b); although most values are statistically significant, and, to repeat, at least part of the difference in patterns may be driven by different boundaries of the I coefficient. Figure 9.1b shows that there is positive autocorrelation at the shortest distances (up to approximately 4 km), then negative autocorrelation that changes in a consistent matter for all the ecosystem services.

The pattern for Historic Environment Character areas is similar to that for Output Areas, although possibly somewhat stronger (Figure 9.1c). The evidence for spatial structure is also at a slightly larger scale, with the cross-over from positive to negative autocorrelation occurring at around 6 km. Again, most cases are statistically significant.

9.4.2. Correlation analysis

Table 9.2 shows the Spearman's rho for each pair of ecosystem services for all spatial units. All except one of the coefficients are statistically significant at a probability threshold of 0.0002, which is the Bonferroni corrected threshold equivalent

Table 9.2. Spearman's *rho* rank correlation coefficient matrix for ecosystem services assessed over three spatial units of analysis. Darker greens indicate strong correlations while lighter yellows indicate weak correlations. White indicates result taken to be statistically non-significant, regardless of whether the Bonferroni corrected threshold is used (none are significant only at one threshold).

		Air pollution reduction	Heat island mitigation	Runoff reduction	Carbon storage	Cultural opportunities	Habitat provision
500m grid squares	Air pollution reduction	-					
	Heat island mitigation	0.18	-				
	Runoff reduction	0.20	0.69	-			
	Carbon storage	0.20	0.92	0.67	-		
	Cultural opportunities	0.03	0.72	0.48	0.73	-	
	Habitat provision	0.52	0.65	0.62	0.60	0.33	-
Output Areas	Air pollution reduction	-					
	Heat island mitigation	0.83	-				
	Runoff reduction	0.47	0.55	-			
	Carbon storage	0.61	0.72	0.60	-		
	Cultural opportunities	0.44	0.34	0.28	0.40	-	
	Habitat provision	0.59	0.66	0.67	0.82	0.42	-
Historic Environment Character Area s	Air pollution reduction	-					
	Heat island mitigation	0.89	-				
	Runoff reduction	0.64	0.71	-			
	Carbon storage	0.65	0.70	0.86	-		
	Cultural opportunities	0.28	0.28	0.40	0.44	-	
	Habitat provision	0.63	0.70	0.85	0.85	0.40	-

of 0.001 (the conservative α level used to account for the fact that spatial statistics are not being used; see Section 9.2). The exception is the correlation between reduction of air pollution and opportunities for cultural ecosystem services for 500m grid squares ($p = 0.18$). Using the approach to interpreting statistical results described in Section 9.2, this strong trend towards statistical significance across the whole table of results indicates that the correlations are unlikely in general to have occurred by chance.

For 500m grid squares, there are strong correlations ($rho \geq 0.6$) between heat island mitigation, reduction of stormwater runoff, carbon storage and habitat provision. Correlations between these ecosystem services and opportunities for cultural ecosystem services are weaker ($rho \geq 0.33$); and with air pollution reduction are weaker still ($rho \leq 0.2$ except with habitat provision).

A slightly different pattern of correlation strengths is apparent in Output Areas and Historic Environment Character area polygons, although these two spatial units show similar patterns to each other. Correlations are at least moderately strong ($rho \geq 0.44$) between all pairs of ecosystem services except opportunities for cultural ecosystem services, for which the strongest correlation is $rho = 0.44$. For these spatial units, correlations with air pollution reduction are only very slightly weaker than between pairs of the other ecosystem services.

9.4.3. Hotspot analysis

Table 9.3 shows the degree of hotspot overlap for all the spatial analysis units. Using the 10% threshold (the 10% of polygons with the highest ecosystem service production levels), 28-41% of polygons are hotspots for at least one ecosystem service, depending on the spatial analysis unit (Table 9.3). Of the polygons that are hotspots for at least one ecosystem service, around 40-60% (i.e. 13-19% of the total) are hotspots for one ecosystem service only, with decreasing proportions for increasing numbers of services. Very few polygons (≤ 5%) are a hotspot for more than three ecosystem services, regardless of the spatial units used.

For the 25% threshold (the 25% of polygons with the highest ecosystem service production levels), 57-60% of polygons are hotspots for at least one ecosystem service, of which 30-43% are hotspots for one service only (equivalent to 17-25% of the total polygons). In contrast to the 10% threshold, there are a few locations that are hotspots for all six ecosystem services for both Output Areas and Historic Environment Character areas. For both thresholds, the results for 500m grid squares are not very different to those for Output Areas and Historic Environment Character areas, although these latter two spatial units are again more similar to each other.

Map 23 to Map 28 show the hotspot overlaps mapped onto the study area. For all spatial units, hotspots of more than three ecosystem services are mainly found in the west of the study area while most of the urbanised area is a hotspot for no or few ecosystem services – although there are some exceptions, especially when looking at Historic Environment Character areas. Similarly, there are some areas in the moorlands that are hotspots for fewer ecosystem services, at least when looking at 500m grid squares (Map 23, Map 24).

Table 9.3. Proportion of spatial units that are hotspots for multiple ecosystem services. Table entries indicate the proportion of spatial unit polygons that are a hotspot for the number of ecosystem services given in the column heading, where hotspots are defined as the (a) top 10%, and (b) top 25% of polygons for a given ecosystem service. Darker greens indicate high proportions while lighter yellows indicate low proportions.

a. Top 10%

Number of ecosystem services	0	1	2	3	4	5	6
500m grid squares	0.59	0.19	0.11	0.07	0.04	0.00	0.00
Output Areas	0.72	0.13	0.06	0.03	0.04	0.01	0.00
Historic Env. Character areas	0.65	0.20	0.06	0.04	0.03	0.01	0.00

b. Top 25%

Number of ecosystem services	0	1	2	3	4	5	6
500m grid squares	0.41	0.18	0.12	0.11	0.14	0.03	0.00
Output Areas	0.43	0.22	0.11	0.07	0.05	0.05	0.06
Historic Env. Character areas	0.40	0.25	0.10	0.08	0.08	0.06	0.01

To further aid understanding of which ecosystem services tend to occur together, Table 9.4 and Table 9.5 show the pairwise overlaps of ecosystem service hotspots, i.e. the proportion of polygons that are hotspots for both of a pair of ecosystem services. For the 10% hotspot threshold for 500m grid squares (Table 9.4), the pairwise overlaps are low (≥ 0.08), with the exception of pairs involving opportunities for cultural ecosystem services; but this is because a large number of ties caused 25% of polygons to be included as hotspots. The only other pair that both occur as hotspots in > 4% of polygons is heat island mitigation and carbon storage. When the threshold is raised to 25% (Table 9.5), there are stronger overlaps between all three pairs within heat island mitigation, stormwater runoff reduction and carbon storage, with at least 17% of grid squares being hotspots for at least two of these ecosystem services. There is again also good overlap between these ecosystem services and opportunities for cultural services. At both thresholds, overlaps involving either habitat provision or air pollution reduction are relatively infrequent.

The results for Output Areas and Historic Environment Character areas are once again similar to each other and quite different from 500m grid squares, with the exception of reduction of stormwater runoff. For Historic Environment Character areas, levels of co-occurrence including stormwater runoff reduction are approximately one

Table 9.4. Pairwise overlap in ecosystem service hotspots, with hotspots defined as the top 10% of polygons for a given ecosystem service. Table entries show the proportion of spatial unit polygons that are a hotspot for both the ecosystem services listed in the column and row headers. Darker greens indicate high proportions while lighter yellows indicate low proportions.

		Air pollution reduction	Heat island mitigation	Runoff reduction	Carbon storage	Cultural opportunities	Habitat provision
500m grid squares	Air pollution reduction	-					
	Heat island mitigation	0.00	-				
	Runoff reduction	0.02	0.03	-			
	Carbon storage	0.00	0.08	0.04	-		
	Cultural opportunities	0.03	0.11	0.08	0.09	-	
	Habitat provision	0.02	0.00	0.01	0.00	0.01	-
Output Areas	Air pollution reduction	-					
	Heat island mitigation	0.05	-				
	Runoff reduction	0.02	0.04	-			
	Carbon storage	0.06	0.06	0.02	-		
	Cultural opportunities	0.03	0.03	0.01	0.05	-	
	Habitat provision	0.04	0.06	0.04	0.06	0.03	-
Historic Env. Character areas	Air pollution reduction	-					
	Heat island mitigation	0.05	-				
	Runoff reduction	0.01	0.01	-			
	Carbon storage	0.05	0.04	0.01	-		
	Cultural opportunities	0.03	0.03	0.01	0.04	-	
	Habitat provision	0.04	0.04	0.01	0.04	0.03	-

third of the values for other ecosystem service pairs; whereas for Output Areas, runoff reduction co-occurs similarly frequently to other ecosystem services.

With the exclusion of stormwater runoff reduction for Historic Environment Character areas, ecosystem service hotspots co-occur in 1-6% of polygons for the 10% threshold, meaning that 10-60% of polygons that are a hotspot for one ecosystem service are also a hotspot for another; and in 9-19% of polygons for the 25% threshold, i.e. 36-76% of hotspot polygons are also a hotspot for another service.

Table 9.5. Pairwise overlap in ecosystem service hotspots, with hotspots defined as the top 25% of polygons for a given ecosystem service. Table entries show the proportion of spatial unit polygons that are a hotspot for both the ecosystem services listed in the column and row headers. Darker greens indicate high proportions while lighter yellows indicate low proportions.

		Air pollution reduction	Heat island mitigation	Runoff reduction	Carbon storage	Cultural opportunities	Habitat provision
500m grid squares	Air pollution reduction	-					
	Heat island mitigation	0.03	-				
	Runoff reduction	0.06	0.17	-			
	Carbon storage	0.03	0.21	0.17	-		
	Cultural opportunities	0.03	0.21	0.16	0.21	-	
	Habitat provision	0.12	0.06	0.08	0.07	0.06	-
Output Areas	Air pollution reduction	-					
	Heat island mitigation	0.18	-				
	Runoff reduction	0.12	0.13	-			
	Carbon storage	0.15	0.16	0.11	-		
	Cultural opportunities	0.12	0.11	0.09	0.15	-	
	Habitat provision	0.14	0.16	0.13	0.19	0.13	-
Historic Env. Character areas	Air pollution reduction	-					
	Heat island mitigation	0.19	-				
	Runoff reduction	0.05	0.05	-			
	Carbon storage	0.15	0.16	0.05	-		
	Cultural opportunities	0.09	0.09	0.03	0.12	-	
	Habitat provision	0.15	0.16	0.06	0.19	0.10	-

9.5. Discussion

The previous six chapters have presented the results of six ecosystem service models, and this chapter has brought them together and compared them. This study appears to be the first that has produced and analysed maps of these six particular ecosystem services. It also appears to be the first to have investigated the spatial covariance of ecosystem services in an urban area. Other studies have undertaken ecosystem service correlation or hotspot analyses, but using different – and, in most cases, fewer – ecosystem services (Chan et al. 2006, Naidoo et al. 2008, Anderson et al.

2009); while others have looked at multiple urban ecosystem services, but not performed these kinds of analyses (Whitford et al. 2001, Tratalos et al. 2007).

The previous chapters provided evidence that ecosystem services production is spatially highly variable across Sheffield, with different services showing different patterns, although there are some spatial similarities and common themes that will be discussed here. This chapter has presented evidence for a reasonably high level of co-occurrence of many ecosystem services; these results are also discussed below.

9.5.1. Spatial themes and patterns

The results presented in the previous six chapters have shown that it is possible to model a variety of ecosystem services, using relatively readily accessible data sources, which give an overview of the spatial variability of the production of those services. A frequently occurring pattern in the model output is that the production of ecosystem services in general increases along a gradient from the urban centre of Sheffield outward, which broadly coincides with decreasing levels of urbanisation. This pattern is commonly seen along urbanisation gradients (Oke 1987, Gill 2006, Tratalos et al. 2007). There are, however, also exceptions: levels of the reduction of air pollution are not much greater in the moorlands than in the city centre, but are considerably higher in the interposing region (Section 3.3.2); and were it not for the existence of the Peak District National Park, areas with high levels of opportunities for cultural ecosystem services would appear to be randomly distributed (Section 7.3.2). These exceptions show that increasing urbanisation does not necessarily decrease the production of ecosystem services; but rather that, although this is a common pattern, the specific characteristics of individual ecosystem services will determine the exact response. In the case of air pollution reduction the observed pattern occurs because moorland is a comparatively poor producer of the ecosystem service, whereas the urbanised area is a complex matrix of land covers that are good and bad producers. Opportunities for cultural ecosystem services, in contrast, are provided by specific land uses; and since the benefits of this ecosystem service are already implicitly valued, space is reserved specifically for those uses in such a way as to provide reasonable levels of access across the urban area.

Regardless of the general pattern, it can also be seen for all ecosystem services that there is considerable variation in levels of production within the highly urbanised region, which is especially visible when using Output Area or Historic Environment

Character area boundaries. In some cases, these levels can be quite high: there are polygons within the city centre that act as hotspots for multiple, and even in some cases all, ecosystem services (Section 9.4.3). For Historic Environment Character areas these polygons represent land uses such as parks, which contain a matrix of land covers that produce high levels of ecosystem services. Output Areas, in contrast, do not delineate land use and so instead represent differences in local urban design that result in variation in ecosystem service production.

Furthermore, there is variation in ecosystem service production between Output Areas of similar size, i.e. having similar population density; and also variation between Historic Environment Character areas representing the same land use. These patterns are true throughout the study area, although they are of more interest in the urbanised region. The first pattern is indicative that there are types of urban morphology that result in higher levels of ecosystem service production even when the same numbers of people are housed there. The variation in production levels between areas of the same land use suggests that, within any single use, ecosystem services can be increased by particular patterns of land cover.

It must be acknowledged that each model is associated with a number of limitations, as discussed in the individual model chapters and Appendix A, which are the result of simplifications of the complexity of nature necessary for tractable models. The two indices (opportunities for cultural ecosystem services, and provision of habitat for biodiversity) are a different kind of simplification to the other models, being based respectively on empirically derived patterns between officially designated land use and individual choices about whether and how to access that land, and land cover composition and ability to support biodiversity. It can be argued that these indices implicitly incorporate some of the complexity specified in more mechanistic models. Building more complex models (either in place of the indices or by increasing the complexity of the existing mechanistic models) would involve significantly greater time, financial and personnel resources. Thus these models represent a trade-off between what can reasonably be achieved with the given resources, and accuracy. Furthermore, simpler and more tractable models can be more readily implemented, perhaps making them a more practical approach to investigating problems.

The fact that the results are output from comparatively simple models is more likely to impact on the absolute values of the results than on the relative values. This is because the model formulae in general are linear; and, presuming that the phenomena they represent are indeed linear in nature, the relationships between the results for a given location remain the same regardless of the absolute input values. Similarly, many of the land cover specific input data are known reasonably well in relationship to each other (for example, average carbon storage levels in different types of vegetation) even if the absolute values have a large error in the present case. When interpreting analyses involving the output of these models, it is therefore preferable to think about ecosystem service production being higher or lower, rather than high or low in absolute terms.

9.5.2. Spatial autocorrelation

When analysed across 500m grid squares, there is strong positive spatial autocorrelation at smaller spatial scales (sites close to each other are more similar than expected by chance) and increasingly negative spatial autocorrelation at larger scales (sites far apart are more different from each other than expected by chance), for most of the ecosystem services (Figure 9.1a). This form of autocorrelation is indicative of a gradient across the study area (Fortin and Dale 2005), as is evident from the ecosystem service maps (Map 8, Map 14, Map 17, Map 20). Positive autocorrelation at short distances reflects the fact that the factors controlling ecosystem service values – i.e. mainly land cover, land use and soil type – show evidence of clustering and also of co-clustering, and suggests that these clustering patterns dominate at spatial scales of up to 8-12 km. In contrast, at larger scales, patterns of land cover, land use and soil type are distinctly different across the west to east gradient of moorland – agricultural land – urban centre. The decrease in negative autocorrelation at the largest scales for storm water runoff reduction is driven by an area of bog moorland in the far west of Sheffield, which produces relatively low levels of the ecosystem service (Map 11).

Reduction of air pollution probably shows less evidence of spatial autocorrelation because of the fact that woodlands produce by far the highest levels of this ecosystem service, and this land cover type tends to be found only in small patches scattered across the study area (Map 2, Figure 3.4). The majority of the study area provides far lower levels of the ecosystem service (Map 5). Thus the production of this ecosystem service is highly localised with no regular pattern or clustering, meaning that there is far less spatial autocorrelation beyond the scale of individual woodland patches. Nevertheless, the general pattern of increasing and then decreasing air pollution levels, with similar values at the east and west (Map 5), is picked up.

In contrast, Output Areas and Historic Environment Character areas show weaker spatial structure, with less pattern at large spatial scales (Figure 9.1b,c). This may be evidence of a large patch structure (Fortin and Dale 2005). If this is the case, it is likely that this patch is the urban centre, which is surrounded by a less developed band (resulting in the slight hump at a lag of around 12-15 km); and the moorland to the west of the study area, which has no equivalent at the other study area borders, may be responsible for the final increase in negative autocorrelation.

Thus aside from the initially high and decreasing positive autocorrelation at short distances, which is expected in any spatial dataset (Fortin and Dale 2005), the patterns are surprisingly different between the spatial units. A possible explanation for the apparently less strong pattern for Historic Environment Character areas arises from the fact that the borders of these polygons are drawn specifically to describe land use.

It has already been shown that land cover compositions differ considerably between land uses (Table 2.11). Thus the Historic Environment Character areas delineate regions of relatively homogeneous land use and cover. Adjacent Historic Environment Character areas show evidence of land use clustering at two levels: polygons of a land use are often adjacent to polygons of the same or a broadly similar land use; and polygons of several different land use types are often found in close association. For example, as shown in Map 3, commercial and industrial land uses are clustered together; and residential land use polygons are clustered but also interspersed with institutional and ornamental, parkland and recreational land use polygons, which are less frequently seen away from residential polygons.

This hierarchical clustering, variation in typical ecosystem service values between land uses, and also the differences in typical polygon sizes between land use types (obvious in Map 3) interact in such a way as to disguise some of the spatial structure evident in the 500m grid squares. The polygons in the city centre are typically small, and mostly comprise the following land use types: commercial; industrial; institutional; ornamental, parkland and recreational; and residential (Map 3). Of these, commercial and industrial are largely composed buildings and manmade surfaces, whereas residential and institutional areas typically have high proportions of natural as well as impervious land covers. Ornamental, parkland and recreational land uses typically have very high proportions of natural cover. As a result, the highly urbanised region contains an irregularly structured matrix of areas with low, medium and high ecosystem service production values. Furthermore, polygons of these small sizes are not seen further out from the city centre. Consequently, less spatial autocorrelation is seen at smaller distances, compared to the arbitrary grid squares that will often average over multiple land uses of different ecosystem service values and thus show less evidence of variation at small scales i.e. higher autocorrelation.

The complex spatial pattern of ecosystem service production in the city centre continues to play a role at larger spatial scales. For example, adjacent city centre polygons of, say, parkland and commercial uses will be a similar distance from an agricultural or moorland polygon, and thus will produce a less consistent spatial effect. There is also an additional hierarchical clustering that comes into play once the medium-sized polygons in the central band of the study area are included. Polygons of enclosed land, which mostly consists of grassland and arable land, are interspersed with water bodies, woodland and a proportion of unenclosed land (chiefly moorland and grassland) that increases further west (Map 3). Woodland land cover produces very high levels of all ecosystem services, while these other natural land covers produce variable amounts of ecosystem services that differ depending on which ecosystem service is being studied. Thus comparisons of patterns of ecosystem service production are inconsistent at all spatial scales, and spatial autocorrelation appears to be reduced. In contrast, the 500m grid squares again do not take account of different patterns of land use and therefore show less variation overall and stronger autocorrelation.

The same idea cannot, however, be applied to Output Areas, which also appear to show less autocorrelation than either of the other spatial units. Nevertheless, Output Areas of similar size consistently show quite different levels of ecosystem service production. This result must be caused by differences in land use and land cover composition (i.e. urban design), and in some cases soil properties, between Output Areas. Therefore although these polygons are not designed to delineate land use or land cover, they obviously carry distinct urban design signatures that drive a similar pattern, i.e. there is no consistent difference in ecosystem service production values between polygons within any distance class.

Much of this discussion is speculative due to uncertainty over how far it is possible to compare statistical results from the different datasets. It nevertheless points towards fundamental differences in what each set of polygons represents and thus their suitability for use in mapping exercises and analyses (Section 9.5.4).

The evidence for spatial autocorrelation makes it difficult to interpret the reliability of estimates of probability values (Section 9.2.1). It is worthwhile noting, however, that while the positive autocorrelation at some, especially short, distance lags (Figure 9.1) would decrease the effective degrees of freedom and thus cause Type I errors, the existence of sometimes strong negative autocorrelation at other lags would increase the effective degrees of freedom (Fortin and Dale 2005). Fortin and Dale (2005) suggest that it can sometimes even be the case that "the test statistic may require inflation rather than deflation to achieve significant results at the correct nominal rates", and that some attempts to correct for spatial autocorrelation by using a lower threshold for statistical significance "may have greatly overcorrected based on the short-range positive autocorrelation but leaving out of consideration the longer rather negative autocorrelation". Although it is not possible to determine the relative effects of the positive and negative autocorrelation here, the presence of both negative and positive autocorrelation means that the approach to statistical inference taken in this study (Section 9.2.1) is even more likely to be adequately conservative.

9.5.3. Ecosystem service correlations and hotspots

Section 9.4.2 and Section 9.4.3 presented the results of two analyses investigating the extent to which levels of ecosystem service production covary. The first of these was a rank correlation analysis, which found quite high correlations between all ecosystem services except with air pollution reduction for 500m grid squares, and opportunities for cultural ecosystem services for Output Areas and Historic Environment Character areas (Table 9.2).

That the correlations between heat island mitigation, storm water runoff reduction and carbon storage are strong is unsurprising, given that production of these ecosystem services is high in areas of natural, vegetated land cover (in the cases of storm water runoff reduction and carbon storage, especially scrubland and woodland) and low in other areas. The correlations between these services and provision of habitat for biodiversity are also quite high, but presumably slightly lower because land cover configuration is also an important factor determining habitat provision.

Correlations between reduction of air pollution and the other ecosystem services are on average lower, and for the 500m grid squares often very low. This is because of the fact that, whereas the other ecosystem services show an increasing gradient outwards from the urban centre, reduction of air pollution is low in both the urban centre and the western moorlands (Map 5). Opportunities for cultural ecosystem services also have a lower correlation with the other ecosystem services, due to dependence on different input data, i.e. the land use map. Furthermore, except for the moorland there is again no gradient from the urban centre outwards (Map 17).

This finding of generally high ecosystem service correlations is in contrast to those from previous studies, which have found quite low correlations (Chan et al. 2006, Naidoo et al. 2008). One reason for this is that the present study used similar input data sources for many of the models, whereas Chan et al. and Naidoo et al. used a wider variety of data sources for their different ecosystem services. In addition, the present study uses a far smaller spatial scale and extent than the other studies, meaning it is possible that the polygons and whole study area used here contain considerably less environmental heterogeneity, and consequently ecosystem service heterogeneity. The other studies also looked at a selection of different ecosystem services, and used different methods for some of the shared ecosystem services, meaning that the results are not directly comparable. Another possible contributing factor is that the present study is looking specifically at an urban area.

There are notable differences in correlation coefficients between the three spatial units. The most notable differences are that correlations between air pollution removal and other ecosystem services are higher for Output Areas and Historic Environment Character areas than for 500m grid squares; and that correlations with opportunities for cultural ecosystem services show the opposite pattern. This is probably due to the fact that the small size of Output Area and Historic Environment Character areas in the urban centre enables more of an air pollution reduction gradient to be observed than is the case for 500m grid squares (Map 5 to Map 7). For cultural service opportunities, the statistical pattern appears to be driven by the fact that there are many 500m grid squares in the moorlands, which have high values of this and other ecosystem services (Map 17), thus increasing the overall influence of this part of the study area on statistical results. In contrast, there are few Output Area and Historic Environment Character areas polygons in this region (Map 18, Map 19), meaning that the statistical pattern is dominated by the random-like spatial pattern of parks and other publicly accessible greenspaces.

An important finding is that for no spatial unit of analysis is any ecosystem service model producing very similar correlation coefficients as any other. This indicates that the output from each model is sufficiently different to warrant analysis of each ecosystem service individually in further investigations, rather than attempting to create an index to simplify statistical testing.

The second investigation identified the degree to which hotspots of different ecosystem services occur in the same places (Section 9.4.3). This analysis found that hotspots of many ecosystem services are rare, although it is not uncommon for a location to provide high levels of 2-3 services (Table 9.3). This general pattern is unsurprising: the correlation analysis has already shown that high levels of ecosystem services are not produced in exactly the same places for all services, but that there is some degree of correlation. This analysis does, however, provide further evidence of the existence of areas that can subjectively be called ‘good’ for ecosystem service production, and that it is therefore worth investigating the characteristic features of these areas.

Map 23 to Map 28 show the hotspot overlaps mapped onto the study area. It is obvious that there is a considerable difference in the amount of dark coloured areas (i.e. high numbers of ecosystem service hotspots) for the Output Areas and Historic Environment Character areas (Map 25, Map 26, Map 27, Map 28), than for the 500m grid squares (Map 23, Map 24). This is a consequence of the largest polygons of the former two spatial units being located in areas that are ‘good’ for ecosystem service production, i.e. the moorland in the west of the study area. This can also explain why a great proportion of polygons are hotspots for at least one ecosystem service for 500m grid squares (Table 9.3): the finer spatial resolution of 500m grid squares in the moorlands, in combination with heterogeneity at relatively small scales that drives differences in ecosystem service production, means that more individual polygons are covered by high levels of production but fewer are hotspots for many services.

The maps also show the entire urbanised area to be particularly poor in terms of ecosystem service hotspots. This is in contrast to what might be expected from the individual ecosystem service maps, most of which suggest that the particularly environmentally degrading effects of the urban core extent over a smaller footprint than is apparent from this result. There are nevertheless some hotspots within the city centre. In Map 27 and Map 28, many of these are due to polygons representing parks and other public greenspace (the other spatial units do not have polygons specifically representing these areas, and subsequently their effects on ecosystem services are less visible), but there are also other instances. These polygons might be a particularly good focus to identify characteristics of ‘good’ ecosystem service producing urban areas.

The pairwise overlaps of ecosystem service hotspots (Table 9.4, Table 9.5), like the correlation analysis, suggest that patterns of co-occurrence and covariation differ between the spatial units: Output Areas and Historic Environment Character areas show similar patterns to each other, but different to those for 500m grid squares. Again, this is likely to be because of the uniform size of 500m grid squares across the study area, whereas the other spatial units have very large polygons in the west and some very small polygons in the city centre. As with the correlation analysis, pairwise overlap is higher than has been observed in a previous study (Anderson et al. 2009). This is again likely to be because of the use of different methods, scales and ecosystem services, and the fact that Sheffield is an urban area, whereas Anderson et al. looked at the entirety of the UK.

Unsurprisingly, the pairwise overlaps show similar patterns of high and low values to the pairwise correlations (Table 9.2). The correlations showed the tendency of levels of ecosystem service production to be high, moderate or low at the same place for pairs of ecosystem services. The additional information yielded by the pairwise overlap of hotspots is specifically that high ecosystem service production levels occur in the same places. Thus, cells in Table 9.4 and Table 9.5 with high values indicate where it is possible to identify locations that are 'good' for production of that pair of ecosystem services, rather than just locations that have similar levels of ecosystem service production.

More importantly, the high degree of ecosystem service covariance indicated by the results from these analyses suggest that there is a potential to design cities such that they have islands of high ecosystem service production; although lower overlaps with provision of habitat and reduction of air pollution suggest that not all ecosystem services can be provided simultaneously. This conclusion provides a good reason to proceed to analyse other, management-related factors that might vary with ecosystem service production and thus be harnessed to improve urban ecosystem service provision.

9.5.4. Choice of spatial units of analysis

Although the effect of choice of spatial scale of analysis on the results of ecosystem service investigations has been analysed in a previous study (Anderson et al. 2009), to the author's knowledge this is the first time that results using different spatial units have been compared. This chapter has shown that consistently different results are obtained by using different spatial units for analysis. This is apparent in the types of detail observable in the ecosystem service maps (see Map 5 to Map 22), and further confirmed in differences in patterns of spatial autocorrelation (Section 9.4.1) and ecosystem service correlations (Section 9.4.2). The fact that these differences are not only visible to the eye, but also influence the results of more formal analysis, indicates that the choice of delineation of spatial units for analysis is important.

In general, results for Output Areas and Historic Environment Character areas are fairly similar to each other, but different to 500m grid squares. As has been mentioned previously, the differences result at least partially from variation in polygon area: 500m grid squares have a consistent size across the whole study area, but both Output Areas and Historic Environment Character areas tend to decrease in size towards the city centre. Therefore more detail can be observed in results for Output Areas and Historic Environment Character areas for the city centre, where population densities are high and land parcels small; whereas in contrast the 500m grid squares provide finer resolution further to the west of the study area.

Moreover, the three different spatial units of analysis represent fundamentally different things, and the fact that the choice of units can profoundly affect the conclusions drawn (Section 9.5.2, Section 9.5.3) means that it is critical to choose an appropriate way to delineate spatial areas according to the question that is being asked. For example, Historic Environment Character areas of the same land use frequently have different levels of ecosystem service production. This must be driven by variation in one or more features of the land cover or soil between those areas. Soil is difficult to manage in terms of altering the processes that determine its characteristics, and excepting soil preservation practices it is therefore not a good target for land management decisions. Land cover, on the other hand, is more easily determined by decision makers. An investigation of differences in land cover composition and layout within a land use thus has the potential to uncover differences in land cover design that increase the production of ecosystem service production. Historic Environment Character area boundaries are ideal for such an investigation because they delineate areas of relatively homogeneous land use, and by extension urban morphology; and are therefore used as the units of analysis in the investigation of urban morphology – ecosystem service relationships in Chapter 10.

Similarly, Output Areas of similar size, and therefore indicating areas of a similar population density, also often show differences in levels of ecosystem service production; and there is obviously huge variation in levels across Output Areas of different sizes. These differences must be driven by some combination of variation in land cover, land use and soil type. Output Areas are less suited to an analysis related to urban design, because the boundaries do not represent areas with a homogeneous development history. They do, however, represent certain socioeconomic aspects of the resident population. Thus these boundaries facilitate the investigation of questions about relationships between ecosystem service production and socioeconomic conditions, and are therefore used in Chapter 11.

Ecologists most commonly use arbitrarily assigned grid squares (or other regular shapes) as the spatial unit of analysis. However, the field of ecosystem services is interdisciplinary, involving questions that go beyond ecology to include disciplines from social sciences, engineering etc. It is therefore important that ecologists carefully consider the spatial unit of analysis when addressing ecosystem service (and other interdisciplinary) questions.

9.5.5. Modelling production versus supply

It is important to take note of what the model output analysed in this chapter actually represents, i.e. levels of the production of ecosystem services at any given site, in contrast to levels of the ecosystem services that are supplying benefits to human welfare. As indicated in Table 9.1, modelling production was possible for all six ecosystem services. Production models have a high potential utility for planning and management decisions, because it is ecosystem service production that planning and management directly alter. These models are ideal for addressing issues such as relationships between urban morphology and ecosystem services (the fifth research aim); and also for identifying priority areas for ecosystem service conservation, which is a common aim of ecosystem service research (Chan et al. 2006, Naidoo et al. 2008).

However, it is obvious from Table 9.1 that modelling the supply of, and benefits from ecosystem services presented challenges that in many cases could not be overcome within this thesis. This was due to the facts that many ecosystem services have complex spatiotemporal properties (Section 1.2.1, Table 2.1), and that there is little quantitative data on ecosystem service benefits available at present. Although the production of all six ecosystem services in Table 9.1 could be modelled, quantifying the supply was only possible for two; for the others, the methods required to model the spatiotemporal ‘flow’ of the ecosystem service from the site of production to the site of supply, i.e. where humans benefit from it, could not feasibly be implemented within this project. Quantifying the benefits was not possible for any ecosystem service, due to a lack of quantitative data about the relationships between ecosystem service production/supply and human welfare benefits.

9.6. Summary

This chapter has been concerned with the output of the models described in the previous six chapters. Some strong patterns of spatial variation can be observed in the output from individual models, and there are also varying degrees of covariation between the ecosystem services. The findings and discussions in this chapter satisfy the fourth research aim listed in Section 1.4.

There is a strong suggestion in these patterns that variation in patterns of land cover, even within single land uses, have a strong effect on the production of ecosystem services. Another pertinent finding from this chapter is strong evidence of spatial autocorrelation in levels of ecosystem service production. This has consequences for inferences that may be made from statistical testing, and thus a cautious approach must be adopted. A final suggestion is that results are strongly dependent on the spatial units being analysed, and it is therefore important to use spatial units that are appropriate to the kind of analysis being undertaken.

10. Urban morphology

10.1. Introduction

Urban morphology, which was briefly introduced in Chapter 1, is the spatial structure and composition of human settlements. Cities are complex entities, and urban morphology can be thought about from a variety of different perspectives, such as density, housing and building types and characteristics, land use patterns, layout and transport infrastructure/accessibility (Jones et al. 2010a). There are theoretical expectations that urban morphology influences ecosystem service production (Alberti 2005); to date, there is very little empirical evidence, but that which does exist strongly supports the existence of such relationships.

These relationships are expected to occur because urban morphology essentially describes the biophysical structure of the urban environment, which as shown in Figure 1.1 influences the environmental processes and functions responsible for the production of ecosystem services (Alberti 2005, Haines-Young and Potschin 2008). However, because the concept of urban morphology is so complex and multi-faceted, in order to perform analyses in a quantitative manner it is necessary to reduce it to simpler metrics.

These metrics include measures such as population or address density (Tratalos et al. 2007, Dempsey et al. 2010, Fuller et al. 2010), the proportional cover of greenspace (Alberti 2005, Bramley et al. 2010), density of road intersections (Stone Jr 2008), the proportion of houses that are detached or semi-detached (Tratalos et al. 2007), and formal indices of land use diversity and the level of aggregation of urban land covers (Alberti 2005, Stone Jr 2008). A longer list of examples of metrics relevant at the neighbourhood scale is shown in Table 10.1. The metrics used in any particular study appear to be selected according to prior expectations and hypotheses about the processes under investigation, as authors rarely cite specific reasons for metric choice. Other authors take a data exploration approach, selecting a larger number of metrics and searching across all possible relationships.

These metrics are used to represent patterns in urban morphology such as land use intensity, heterogeneity and connectivity (Alberti 2005). Table 10.2 summarises the existing empirical evidence that could be found for ecosystem service – urban morphology relationships for the services featured in this project (except for

Table 10.1. Examples of urban morphology metrics. (NB not all referenced studies are investigating relationships with ecosystem services.)

Urban morphology component	Metric description(s)	Reference(s)
Aggregation of the built environment	The extent to which developed land is aggregated (versus dispersed throughout an area)	Alberti (2005)
Amount of green-space/vegetation	Proportional area that is not under developed land uses	Giridharan et al. (2007), Tang & Wang (2007)
Building design	Metrics such as average building size, average height:floor area ratio, sky view factor (proportion of a ground-level fish eye lens image that comprises sky), number of bedrooms (residential only), number of stories	Davies et al. (2008), Giridharan et al. (2007), Stone & Bullen (2006)
Connectivity	Density of the street network e.g. number of street intersections per unit area, total length of roads	Cutts et al. (2009), Davies et al. (2008), Frank et al. (2006), Stone Jr (2008), Wood et al. (2010)
Land use mix	Metrics such as ratio of jobs to population, number and/or evenness of different land uses, proportion of land in different land uses e.g. buildings	Alberti (2005), Davies et al. (2008), Frank et al. (2006), Kaczynski et al. (2009), Stone Jr (2008), Tang & Wang (2007), Wood et al. (2010)
Lot design	Metrics such as area of paving, distance from road to building, whether houses are detached/semi-detached/terraced etc.	Stone & Bullen (2006), Tratalos et al. (2007)
Population (or housing, or address) density	Number of resident people per unit area, or amount of land per resident	Davies et al. (2008), Stone Jr (2008), Tang & Wang (2007), Tratalos et al. (2007), Wood et al. (2010)

biodiversity/habitat provision, which was reviewed in Section 1.2.6). Only six studies could be found (one of which looked at multiple services), of which three investigated heat island mitigation, three investigated stormwater runoff reduction, and two investigated cultural ecosystem services. The studies mostly use different urban morphology metrics, making it difficult to compare results between studies; the exception is that low land use diversity is correlated with greater physical activity levels in both studies of cultural ecosystem services (Kaczynski et al. 2009, Wood et al. 2010). While more studies have looked at urban morphology – biodiversity relationships (Section 1.2.6), there are few components of urban morphology that have been studied extensively enough to make generalisations, and there are often differences

Table 10.2. Summary of previous studies investigating relationships between urban morphology and the ecosystem services modelled in this study (excluding biodiversity/habitat provision).

Ecosystem service(s)	Methodological details	Results
Heat island mitigation (Giridharan et al. 2007)	Urban morphology parameters and heat island effect measured at 17 high rise developments in Hong Kong.	Different urban morphology metrics are found to influence heat island intensity under different weather conditions and at different times of day; e.g. amount of vegetation, height:floor area ratio of buildings (representing thermal mass), sky view factor.
Heat island mitigation (Johansson and Emmanuel 2006)	Urban morphology and temperature parameters measured in five street canyons in Colombo, Sri Lanka.	Temperatures experienced by people are lower in deeper street canyons (which provide shade) or streets with trees, awnings etc. to provide shade; irregularity in building positioning and height encourages airflow and cooling.
Heat island mitigation, stormwater runoff reduction (Tratalos et al. 2007)	Urban morphology parameters measured, and ecosystem services estimated, at inner, middle and outer sites in five UK cities.	High urban density is associated with low ecosystem service levels, although high variability indicates scope for improvement at a given density. Housing type is also important for some ecosystem services.
Stormwater runoff reduction (Stone and Bullen 2006)	Design of 38,000 single family residential lots in Madison (Wisconsin, USA) analysed for relationships with estimated stormwater runoff	Narrowing the distance between the street and the house reduces stormwater runoff due to less street paving and increase likelihood of multi-story construction & less floor area. For large lots, decreasing lot size is also associated with decreased runoff due to lower impervious cover.
Cultural ecosystem services (Kaczynski et al. 2009)	Land use diversity in a 500m buffer surrounding parks, and use of parks for physical activity, measured for 32 parks in a Canadian city.	Lower land use diversity surrounding a park is associated with higher use of the park for physical activity, especially in parks with more facilities.
Cultural ecosystem services (Wood et al. 2010)	Interviews with 609 residents of the Atlanta (Georgia, USA) region, and measurement of urban morphology parameters around their residences.	Sense of community and leisurely walking are associated with lower levels of land use mix and a lower proportional cover of buildings in commercial districts.

between taxa; an exception is that very high proportional cover by impervious surfaces reduces biodiversity across most taxa.

Despite the sparse evidence, each of the studies listed in Table 10.2 found significant correlations between urban morphology and ecosystem services; therefore it is expected that patterns will also occur in Sheffield for the ecosystem services studied here. The purpose of this chapter is to undertake an investigation of relationships between ecosystem service production and a selection of metrics that were chosen to represent qualitatively diverse facets of urban morphology, in order to produce recommendations for urban planners and decision makers who wish to increase levels of ecosystem services within neighbourhoods.

10.2. Methods

10.2.1. Urban morphology metrics

Seven qualitatively diverse metrics are used in this study, and a data exploration approach is taken, i.e. relationships are tested for between all metrics and all ecosystem services, without specific hypotheses for individual metrics or ecosystem services. The metrics are defined fully in Appendix B.2; Appendix B also presents the results of an analysis showing the metrics to have some degree of independence from one another. This section describes the metrics more briefly to convey a functional understanding of what each represents. These metrics are measured at a neighbourhood scale; specifically, the Historic Environment Character area polygons are used, as they are reasoned to contain relatively homogeneous areas in terms of urban morphology (as each polygon has a unique history, including date and type of development), compared to other available GIS datasets. Maps of all metrics can be found in Appendix D (Map 29 to Map 35).

The *proportion of impervious cover* (i.e. buildings and manmade surfaces) is a simple representation of the amount of development, and also conversely the amount of ecosystem service-providing greenspace, in an area. It is expected, both from theory and from the way the ecosystem service models work, that high proportions of impervious cover will be associated with low ecosystem service provision.

The *building density* and *mean building size* give information about the 'amount' of building in an area and the average areal footprint respectively; together, these metrics also show the proportion of building cover (building density multiplied by mean building size).

The *impervious surface normalised landscape shape index (nLSI)* is used to describe the layout of the built environment. It quantifies the extent to which the layout of the impervious surfaces in an area diverges from the most compact shape, and thus is a measure of land cover fragmentation and landscape complexity (Patton 1975, McGarigal et al. 2002). This facet of urban morphology is important with regards to ecosystem services because it is related to the matrix of habitat patches available for biodiversity and ecosystem services; for example, a single large patch of greenspace might support populations of more sensitive species, while a number of smaller patches may mean an ecosystem service is more spread out and accessible (Handley et al. 2003, Garden et al. 2006, McKinney 2008).

Population density is frequently included in studies of urban morphology as it describes patterns of residence and is often related to patterns in greenspace occurrence (Burton 2000, Tratalos et al. 2007); it is therefore also included here. The *proportion of households in detached and semi-detached houses (proportion detached houses)* is to some extent related to population density, but it is possible for different numbers of people to occupy the same type of housing. Furthermore, housing type strongly influences neighbourhood design, and detached and semi-detached houses in particular tend to have a larger footprint than other housing types (Environment Agency 2004).

Finally, the *weighted road length density index (road LDI)* is used to describe the transportation network; specifically, how well connected an area is to surrounding regions. A system of weighting is used, as it was found to improve representation of connectivity. Transport network connectivity is important because of its relationship with neighbourhood accessibility and environmental disturbances caused by traffic (Volchenkov and Blanchard 2007, Bouchard et al. 2009).

10.2.2. Methodological overview

The general hypothesis of this chapter is that patterns of urban morphology are associated with levels of ecosystem service production; a particular result expected from theory and previous studies is that patterns of urban morphology that are typical of less intensively developed land use types will correlate with higher levels of production, because lower levels of development typically mean that there is less environmental disturbance and greater greenspace cover (Alberti 2005, Tratalos et al. 2007). However, more intensively developed land use types are by definition more constrained in the types of morphology that are possible: for example, a higher building density and proportion of impervious cover are necessary to accommodate housing, factories etc. compared to agriculture; and such land uses will result in higher population densities.

This is a relevant problem for the present study: values of the urban morphology metrics are often strongly dependent on the land use type, as shown in Appendix B.4.3. It is therefore necessary to account for the confounding of urban morphology with land use in the analysis of urban morphology – ecosystem service relationships. Ideally, this issue would be tackled using a multiple regression approach. However, given the need for nonparametric statistical methods with this dataset, and the fact that land use is a categorical variable with many categories (making inclusion in regression modelling challenging), alternatives had to be found.

Therefore several different analyses were used to address different parts of the question of how urban morphology and land use affect ecosystem service production singly, in combination, and interactively. The relationships between these analyses are shown in Figure 10.1a.

The first analysis examined relationships between urban morphology and ecosystem services (labelled 1 in Figure 10.1a). Rank correlation tests quantified the strength and direction of relationships between each combination of ecosystem service and urban morphology metric. This made clear where the strong relationships are.

The second analysis, labelled 2 in Figure 10.1a, used a data mining tool to compute the relative importance of land use type and the urban morphology metrics for predicting levels of ecosystem service production. This tool was a recursive partitioning based method, namely a variant of random forest analysis.

Finally, for ecosystem services for which land use is an important factor (as identified by the random forest analysis), rank analysis of variance and post hoc pairwise comparison tests were used to establish how the ecosystem service varies with the confounding factor of land use. This third analysis is labelled 3 in Figure 10.1a.

In synthesis, these analyses are able to give an overview of the influence of urban morphology on ecosystem services across a wide range of land use types, and how the relationship is driven by land use. A complementary approach to accounting for land use is to investigate relationships within individual land uses. Therefore a subset of the analyses applied to the whole dataset was repeated for two data subsets including only one broad land use each: residential and industrial (illustrated in Figure 10.1b). The results from these subsets show how ecosystem service production is influenced by urban morphology, within the constraints on morphology that are imposed by the particular land use type.

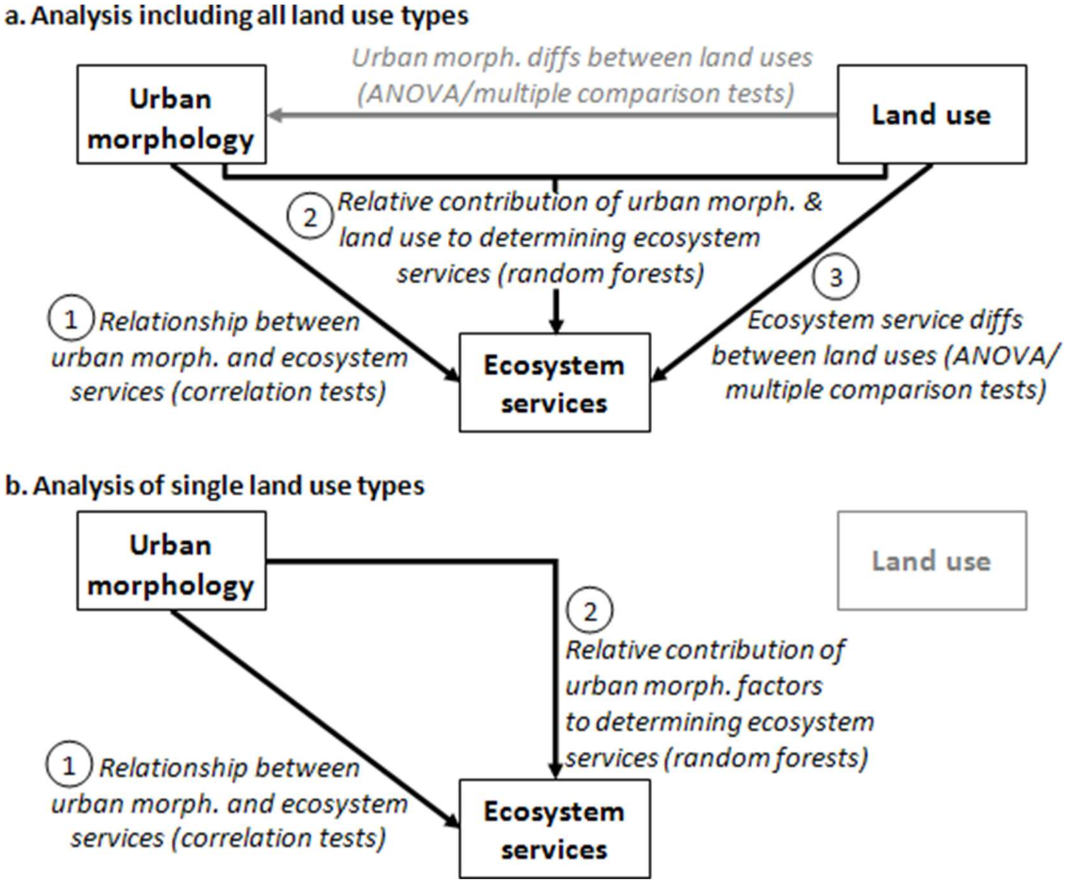


Figure 10.1. Use of analyses to investigate the effects of various aspects of urban morphology and land use type on ecosystem service production, and their importance relative to each other (numbers correspond to descriptions in text). Arrows show relationships analysed by tests described in corresponding italic text. Arrow shown in grey indicates rank analysis of variance and post hoc multiple comparison tests to identify the confounding of land use with urban morphology (Appendix B.4.3).

Residential and industrial land uses were selected because they cover the largest areas of the more intensively developed land use types (22.2% and 4.4% respectively; Table 2.12) and are also represented by sufficient polygons in the dataset to facilitate analysis (866 and 262 respectively). Moreover, analysis of the full dataset found that, of the intensively developed land use types, residential and industrial land uses are towards opposite ends of the scales of ecosystem service production levels and most urban morphology metrics.

There remains some variation in land use within these broad land use types (see Table 10.3 for a breakdown of residential and industrial land use subtypes in Sheffield). However, it can reasonably be expected that there is less variation within than between broad land use categories because, for example, residential areas typically have gardens

Table 10.3. Subcategories of (a) residential and (b) industrial present in Sheffield, as defined by the Historic Environment Character areas. Asterisk indicates categories subsumed into "other" due to small sample size.

a. Residential

Category	**Count**
Back to back/courtyard houses*	2
Burgage plots*	2
Elite residence	25
Farm complex	28
High rise flats*	16
Low rise flats	47
Planned estate (social housing)	197
Prefabs*	6
Private housing estate	112
Semi-detached housing	167
Terraced housing	151
Romany/traveller community site*	2
Vernacular cottages	20
Villas/detached housing	91
Total	*866*

b. Industrial

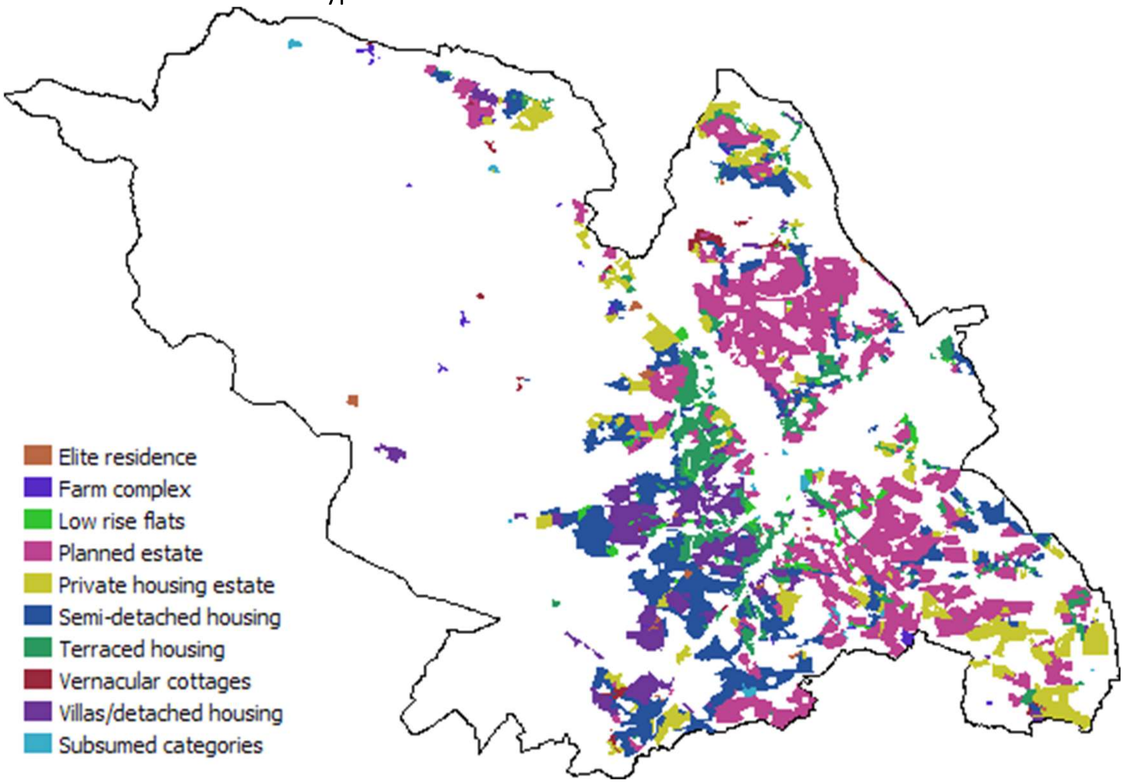
Category	**Count**
Chemical*	3
Craft industry*	1
Metal trades (heavy)	65
Metal trades (light)	64
Metal trades (support)*	8
Other industry	90
Textile trade*	1
Utilities*	12
Water powered site*	18
Total	*262*

and are usually arranged along streets; whereas industrial areas normally require warehousing, access roads and large paved areas for logistical purposes.

Figure 10.2 shows the spatial distribution of residential and industrial land use subtypes across Sheffield. For both residential and industrial land uses, a number of subtypes are subsumed into a single subtype "other" for the purposes of statistical analysis, due to small sample size (shown with an asterisk in Table 10.3).

The rank correlations of ecosystem service – urban morphology metric relationships and random forest analyses are repeated for the single land use types (Figure 10.1b, labels 1 and 2 respectively). Whereas broad land use type is included in random forest analysis of the full dataset, land use *subtype* is included here to check whether and how much finer scale land use differences affect ecosystem service production, relative to urban morphology. However, as land use subtype is found to be relatively unimportant compared to urban morphology for all ecosystem services, rank analysis of variance and pairwise comparison testing is not repeated here; hence land use is shown in grey in Figure 10.1b. Opportunities for cultural ecosystem services were not analysed for these individual land uses, as neither residential nor industrial sites are considered to provide

a. Residential land use subtypes



b. Industrial land use subtypes

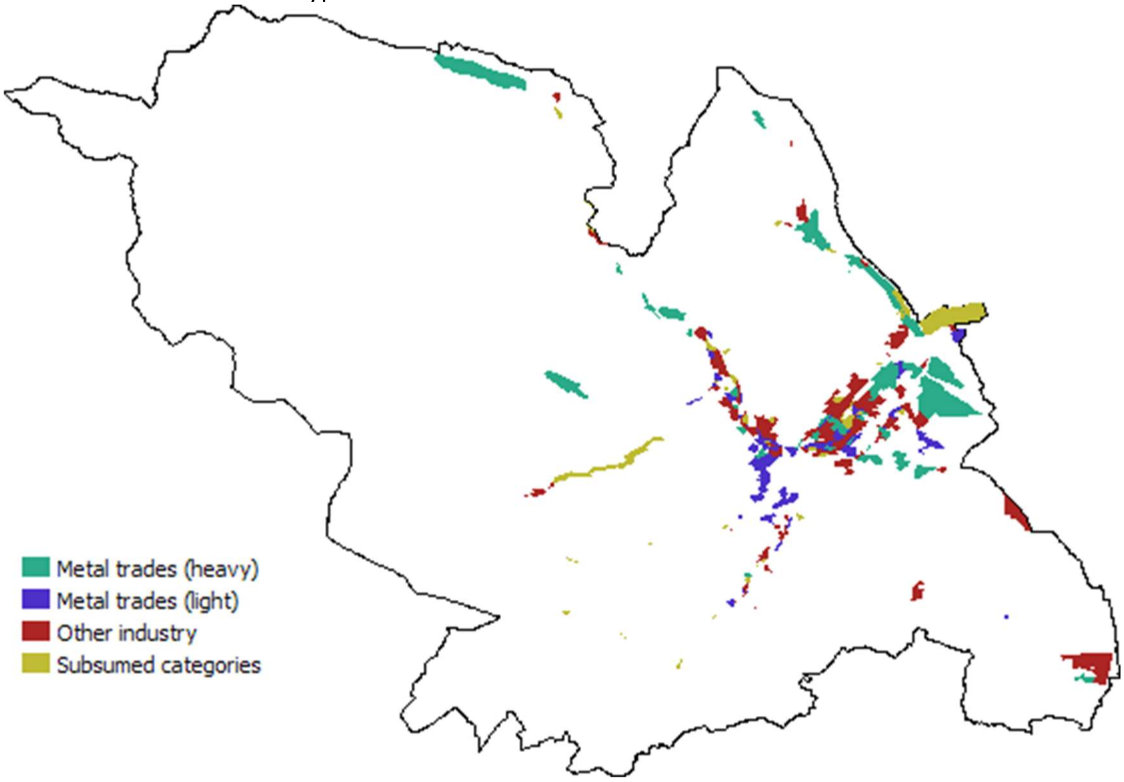


Figure 10.2. Location of (a) residential and (b) industrial land use subtypes. Subsumed categories represent the subtypes marked with an asterisk in Table 10.3.

the necessary infrastructure and thus the production of this ecosystem service is always zero.

10.2.3. Analysis one: rank correlations

Spearman's rank correlation coefficient, rho, is used to quantify the strength and direction of the relationship between each urban morphology metric – ecosystem service pair. The procedure followed here is the same as described in Section 9.3.2, and the approach to statistical inference is as described in Section 9.2. The Bonferroni corrected significance threshold is $\frac{0.001}{42} = 2.38 \times 10^{-5}$ (the uncorrected threshold is 0.001, rather than 0.05, to account for spatial autocorrelation; see Section 9.2.1). This method is applied both to the whole dataset and the residential and industrial subsets.

10.2.4. Analysis two: random forests

10.2.4.1. Background to random forest methods

Random forest analyses are used in this study to estimate the relative importance of the seven urban morphology metrics and land use type (or subtype) to each of the ecosystem services. Random forest analysis was developed as a data mining method, and is popular in bioinformatics and other statistical fields of biology for building predictive models using large numbers of predictor variables, and analysis of the relative importance of those variables to the value of the response variable (Cutler et al. 2007, Strobl et al. 2007, Strobl et al. 2008).

Random forests are an ensemble technique (i.e. uses a set of alternative models) built using recursive binary partitioning trees, such as classification and regression trees, which are an alternative to statistical modelling. The general approach in recursive binary partitioning tree methods is to search across all the predictor variables to find the dichotomous split that maximises some measure of homogeneity in the two resultant groups (De'ath and Fabricius 2000). This process is repeated recursively for each of the resultant groups until some criterion to stop "growing" the tree is met, thereby producing a dichotomous decision tree in which the predictor variables are used to partition the data into groups in which values of the response variable are as similar as possible (De'ath and Fabricius 2000).

In the present study, trees are built using a conditional inference framework (Hothorn et al. 2006, Strobl et al. 2007). This framework overcomes a problem associated with other tree-based methods, namely bias in variable selection towards variables with a greater number of potential splitting points (De'ath and Fabricius 2000, Hothorn et al. 2006). Conditional inference statistical tests are used to select the splits, with greater degrees of freedom penalising predictor variables with more potential splitting points: thus the variable with the lowest probability of independence with the response variable is selected (Hothorn et al. 2006, Strobl et al. 2007).

As well as unbiased variable selection, the strengths of this method are that a mixture of categorical and numerical explanatory variables can be used; missing data do not compromise tree-building; non-linear and interactive relationships can be investigated; and the frequency distribution of variables is unimportant (De'ath and Fabricius 2000, Strobl 2009). On the other hand, individual trees are unstable, such that small changes to input data can result in very different trees (Strobl et al. 2007).

This problem can be overcome by using an ensemble of trees, i.e. a random forest. Random forests are built by repeatedly extracting a bootstrap sample from the original data, and fitting a classification and regression tree to that sample (Strobl et al. 2007). Bootstrapping without replacement is used in order to minimise the artificially induced association between variables that results from performing statistical inference on bootstrapped samples (Strobl et al. 2007). At each split in the tree, a random subsample of the predictor variables is searched for the optimal split, a process that can reveal otherwise hidden interactive effects (Strobl et al. 2008). Thus the ensemble of trees in a random forest is very diverse.

A predictive model is built from this forest of trees by inputting the predictor variable values from new data into every tree. The response value is then predicted across the forest from the average or majority vote (for regression and classification tasks respectively) of the predictions from each single tree (Peters et al. 2007, Strobl et al. 2007). The criterion to stop growing each tree in a random forest is when each data point is contained in a separate "leaf" of the tree (Breiman 2001). This causes over-fitting of single trees; but for random forests the law of large numbers, and the empirically confirmed fact that the unstable individual trees are nevertheless on average correct, mean that the predictive results converge once there are adequate trees in the forest (Breiman 2001, Peters et al. 2007, Strobl et al. 2008). Convergence is tested using the generalisation error for the forest, as calculated from the error of out-of-bag observations (i.e. those not in the bootstrap) from each tree, across the whole forest

(Peters et al. 2007). When adding further trees to the forest does not decrease the error, convergence has been reached.

There are several methods that can be used to estimate the relative importance of the predictor variables to the value of the response variable, of which the most advanced is the permutation accuracy importance (Strobl et al. 2007). The permutation accuracy is calculated from the increase in generalisation error that results from random permutation of the values of a single predictor variable in the out-of-bag observations (Breiman 2001, Peters et al. 2007). The original association between the response variable and that predictor variable is thus broken, and the resultant decrease in the model's predictive accuracy is proportional to the predictor variable's importance (Strobl et al. 2007). Predictor variables that are more important will cause a greater decrease in the model's predictive accuracy compared to those that are less important.

Bias towards correlated predictor variables is minimised by using a permutation scheme that considers the effect of permutation of a variable conditional on the values or other variables (Strobl et al. 2008). This prevents inflation of the importance of variables with a large influence when the effect of permutation is looked at across the whole variable space, but with little or no influence when the values of other predictor variables are controlled for (Strobl et al. 2008).

10.2.4.2. Implementation

These procedures were implemented using the package 'party' v0.9-994 (Hothorn et al. 2006, Strobl et al. 2007, Strobl et al. 2008) for the computer program R v2.8.1 (R Development Core Team 2008). The settings recommended to minimise sources of bias were used; the only other control parameter is the size of the subset of variables tested at each split. Strobl et al. (2008) suggest that the square root of the total number of predictor variables is often optimal in empirical studies; therefore the closest whole number to the square root is used here (i.e. eight predictor variables = three tested at each split).

The conditional permutation procedure proved to be computationally unfeasible for datasets with a sample size of greater than approximately n=500, so repeated random subsampling was used for analysis of the full dataset and residential land uses, using a different random seed number each time. For the full dataset, fifteen forests of size n=500 (21% of the dataset) were grown; and for residential land uses, eight forests of size n=400 (46% of the data points) were grown. For industrial land uses, all data points were used to grow three forests.

The results presented here are the mean, maximum and minimum importance of each variable, after scaling the importances from each random forest to sum to one. On occasion, the variable importance algorithm computes importances of less than zero. In these cases, the negative values were set to zero prior to scaling.

Convergence can be tested by repeating the analysis using a different random seed number and checking that the rank order of variable importance does not change. Convergence was tested in the present study by building forests on full (not subsampled) datasets using three different random seed numbers. Convergence was considered to have occurred if the rank order did not vary except for variables with <5% difference in scaled importance between runs. A forest size of 500 trees was found to be adequate in all cases.

10.2.5. Analysis three: analyses of variance and pairwise comparison testing

This method was used in investigation of the full dataset only, and only for the ecosystem services for which the random forest analysis found land use to be an important factor. To investigate how ecosystem service levels vary with land use type, Kruskal-Wallis rank analysis of variance was performed for each urban ecosystem service. The results of all tests were significant at the Bonferroni-corrected level, confirming the existence of differences between land use types. Post hoc pairwise comparison tests, namely Mann-Whitney U tests, were therefore undertaken between all pairs of land use types to identify the land use types between which significant differences lay. Because the numbers of polygons vary across two orders of magnitude between land use types (Table 10.4), a correction for unbalanced designs was used. A total of 66 pairwise comparisons was made for each urban morphology metric and ecosystem service; therefore the Bonferroni corrected significance threshold is $\alpha = \frac{0.001}{66} = 1.51 \times 10^{-5}$.

Table 10.4. Numbers of polygons described as each land use type in the Historic Environment Character GIS dataset.

Land use type	Number of polygons
Residential	866
Institutional	344
Industrial	262
Ornamental, parkland and recreational	245
Commercial	206
Woodland	129
Enclosed land	125
Unenclosed land	95
Communications	39
Extractive	18
Water bodies	16
Horticulture	2

10.3. Results

10.3.1. Analysis across all land uses

10.3.1.1. Rank correlations: urban morphology – ecosystem service relationships

Table 10.5 shows Spearman's rank correlations between urban morphology metrics and ecosystem services across all land use types. All relationships are statistically significant regardless of whether the Bonferroni correction is used or not, with the exception of impervious surface nLSI, which is only significantly correlated with two of the ecosystem services.

There are some general patterns in the strength of relationships. Correlations with opportunities for cultural ecosystem services are generally lower than is the case for the other services, while the others all show similar correlation strengths. The proportion of impervious cover is the urban morphology metric with the strongest relationship with all ecosystem services (a lower proportion of impervious cover results in higher ecosystem service production). The mean building size and proportion of detached houses also show strong correlations with all ecosystem services except opportunities

Table 10.5. Spearman's *rho* rank correlation coefficients for ecosystem services against urban morphology metrics. Yellows indicate weak correlations; darker greens and reds indicate stronger positive and negative correlations respectively. White background indicates results taken to be statistically non-significant, regardless of whether the Bonferroni correction is used.

	Air pollution reduction	Heat island mitigation	Runoff reduction	Carbon storage	Cultural opportunities	Habitat provision
Proportion impervious cover	-0.64	-0.71	-0.90	-0.90	-0.49	-0.90
Building density	-0.32	-0.30	-0.39	-0.37	-0.48	-0.46
Mean building size	-0.42	-0.46	-0.56	-0.61	-0.40	-0.51
Impervious surface nLSI	0.05	0.07	0.03	0.09	0.12	-0.04
Population density	-0.30	-0.31	-0.22	-0.19	-0.20	-0.30
Proportion detached houses	0.47	0.59	0.55	0.52	0.12	0.51
Road LDI	-0.35	-0.39	-0.45	-0.46	-0.31	-0.54

for cultural ecosystem services ($0.42 \leq rho \leq 0.61$). The only metric showing weak correlations is the impervious surface nLSI.

There are also strong patterns in correlation directions, with the direction for all ecosystem services being the same for each urban morphology metric except the impervious surface nLSI. However, the rank order of the strength of correlations within an urban metric morphology and between ecosystem services differs between metrics.

10.3.1.2. Random forest analysis: importance of metrics and land use

The relative importance of each of the urban morphology metrics and land use type, as computed by permutation accuracy of variables in random forests, is shown in Figure 10.3. Some of the confidence intervals are quite large, and the rank order of variable importance often differed between random forest runs (results not shown). This is an artefact of using random data subsamples, arising from some variables being important for the prediction of ecosystem service production only for some data subsamples, and suggests that the "true" rank order of variable importance may not be correct in

Figure 10.3 where the confidence intervals overlap. There is also some variation introduced by the use of different random seed numbers, but this variation is small by comparison.

Land use category is the most important variable in only one case: opportunities for cultural ecosystem services. In comparison to land use category, which accounts on

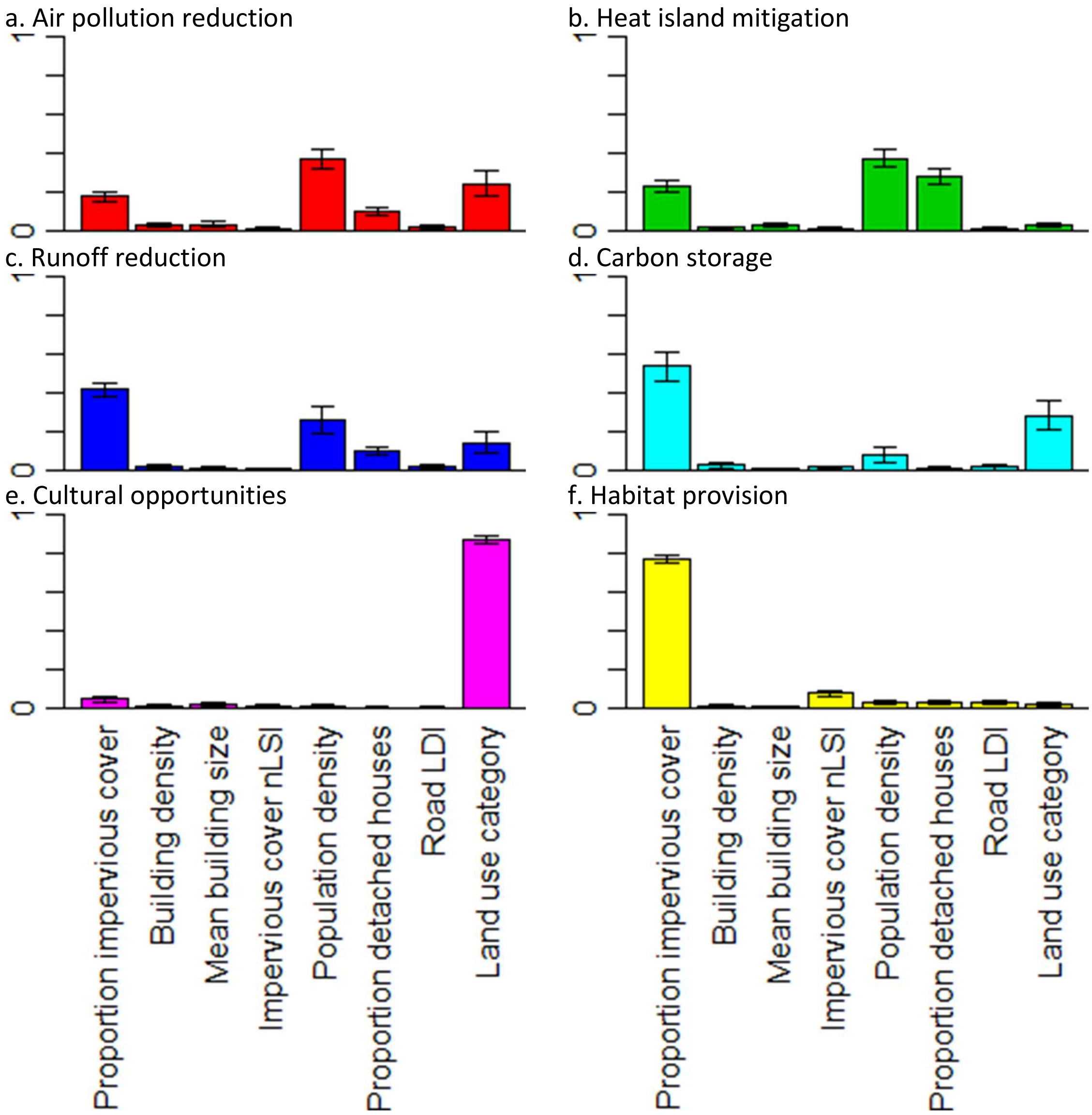


Figure 10.3. Relative importance of urban morphology metrics, land use category and date of establishment of present land use type, to the production of ecosystem services, as determined through random forest analysis. Variable importances are means from ten random subsamples, with the results from each subsample scaled to sum to one for each ecosystem service. Error bars show the 95% confidence interval across subsamples.

average for 87% of the random forest model's predictive ability, no urban morphology metric is of substantial value in predicting the provision of this ecosystem service. Land use category is also substantially important to two other ecosystem services, namely the reduction of air pollution and carbon storage (25% and 29% of total predictive ability, on average, respectively).

For all the ecosystem services except opportunities for cultural ecosystem services, the urban morphology metrics represent an average of at least 71% of the total predictive value of the variables included in the random forest models. However, three of the metrics do not account for more than an average of 4% of the total variable importance for any of the ecosystem services. These metrics are building density, mean building size, and the road LDI.

The proportion of impervious cover is the most important variable for predicting levels of three ecosystem services, namely reduction of stormwater runoff (accounting for 42% of total predictive ability), carbon storage (54%) and habitat provision (77%). In addition, stormwater runoff is also sensitive to population density (26% of population density), proportion detached houses (10%) and land use type (15%). Carbon storage is also sensitive to land use category (29%) and a little to population density (8%). Aside from the proportion of impervious cover, habitat provision is only slightly sensitive to the impervious surface nLSI (8%).

A variety of urban morphology metrics are important to the two other ecosystem services: reduction of air pollution and heat island mitigation. Population density is the most important predictor of both of these services, accounting for 37% and 38% of the total predictive ability respectively. The proportion of impervious cover and land use category are also important to air pollution reduction, and also to some extent the proportion of detached houses (18%, 25% and 10% respectively of total predictive ability respectively). The proportion of impervious cover and the proportion of detached houses also have some importance to heat island mitigation (23% and 28% respectively).

10.3.1.3. Multiple comparison tests: land use – ecosystem service relationships

Figure 10.4 shows how the average ranks of ecosystem service production levels vary between land use types for reduction of air pollution, carbon storage and opportunities for cultural ecosystem services (land use was not found to play an important predictive role, i.e. >15% of total predictive ability, for the other ecosystem services). As was found to be the case between urban morphology metrics and land use (Appendix B.4.3), there are considerable differences in mean ranks between land use types, which often translate into significant pairwise differences (Table 10.6).

Average ranks for different land use types tend to be similar between air pollution reduction and carbon storage, with the exception of water bodies (Figure 10.4a,b, Table 10.6a,b). This is because carbon in soil under water bodies is not included in the carbon storage model (the source data does not include estimates for carbon storage in soil under water bodies). The ranks for opportunities for cultural ecosystem services (Figure 10.4c, Table 10.6c), however, show a very different pattern. Land uses involving intensive building (residential, institutional, communications, industrial and commercial) form a group providing lower levels of air pollution reduction and carbon

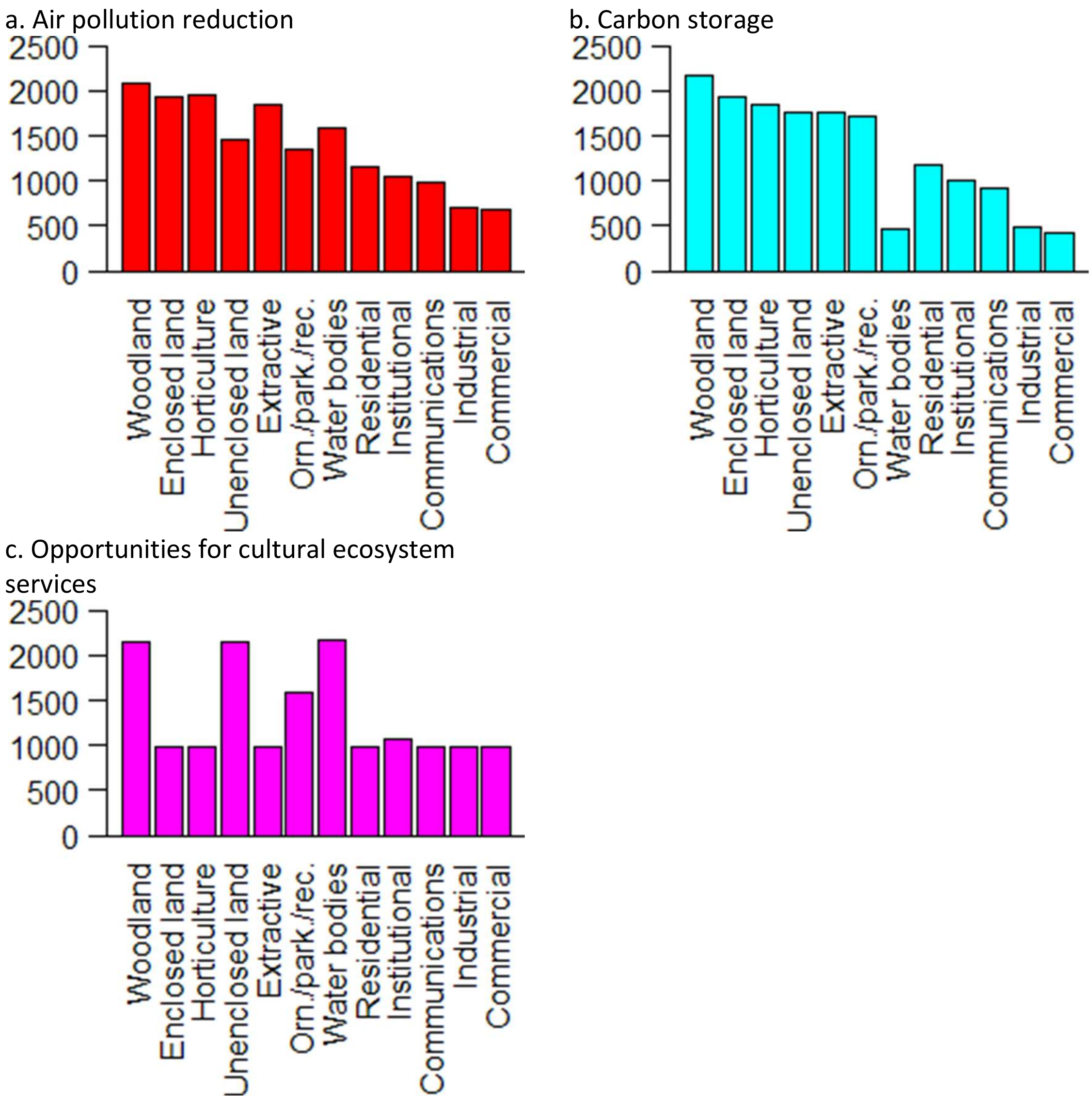


Figure 10.4. Average rank of ecosystem service production levels across different land use types. High ranks indicate high production values. Land use types ordered by decreasing mean across all ecosystem services (including those not shown here).

Table 10.6. Mean ranks and pairwise multiple comparisons of ecosystem service production between different land use types. Numbers in second column show the average rank of ecosystem service production for the land use type in the first column (where high ranks are high production). In the subsequent columns in the cells under the dashes, black cells indicate where the land use type in that column produces significantly greater ecosystem service levels than the land use type in that row at the Bonferroni-corrected significance threshold; a grey fill indicates a difference significant at the non-corrected threshold; and white indicates no significant difference. NAs in (c) arise due to tied ranks. (Multiple comparison test: Mann-Whitney U-test.)

a. Air pollution reduction

		Woodland	Horticulture	Enclosed land	Extractive	Water bodies	Unenclosed land	Orn./park./rec.	Residential	Institutional	Communications	Industrial	Commercial
Woodland	*2084*	-											
Horticulture	*1954*		-										
Enclosed land	*1931*	■		-									
Extractive	*1853*	▒			-								
Water bodies	*1589*					-							
Unenclosed land	*1455*	■		■			-						
Orn./park./rec.	*1353*	■		■	▒			-					
Residential	*1148*	■		■	■		■	■	-				
Institutional	*1047*	■		■	■		■	■		-			
Communications	*980*	■		■	▒		▒				-		
Industrial	*693*	■		■	■	■	■	■	■	■		-	
Commercial	*671*	■		■	■	■	■	■	■	■			-

storage than less intensive land uses (Table 10.6a,b); with the exception of water bodies for carbon storage, again because carbon in soil under water bodies is not included.

Woodlands provide the highest levels of both air pollution reduction and carbon storage (Table 10.6a,b), although there is not a great difference between this and several other land uses (especially enclosed land, horticulture and extractive, although not all these differences are statistically significant). Unenclosed land and ornamental, parkland and recreational land uses also provide quite high levels of these ecosystem services. Residential and institutional land also has higher levels of both carbon storage and air pollution reduction than other intensive land uses, with pairwise comparisons being significant in many cases.

Table 10.6 continued.

b. Carbon storage

		Woodland	Enclosed land	Horticulture	Unenclosed land	Extractive	Orn./park./rec.	Residential	Institutional	Communications	Industrial	Water bodies	Commercial
Woodland	*2166*	-											
Enclosed land	*1924*		-										
Horticulture	*1855*			-									
Unenclosed land	*1769*				-								
Extractive	*1761*					-							
Orn./park./rec.	*1724*						-						
Residential	*1172*							-					
Institutional	*994*								-				
Communications	*909*									-			
Industrial	*478*										-		
Water bodies	*469*											-	
Commercial	*411*												-

In contrast, land uses fall into one of two groups with regards to opportunities for cultural ecosystem services (Figure 10.4c, Table 10.6c). Water bodies, unenclosed land and woodland have high values, while institutional, industrial, residential, enclosed land, horticulture, extractive, communications and commercial land uses have low levels. Only ornamental, parkland and recreational land uses have a value between these two groups. However, this result conveys relatively little insight since the index was designed such that certain land uses provide this ecosystem service, but others do not.

10.3.2. *Residential and industrial land uses*

10.3.2.1. *Rank correlations: urban morphology – ecosystem service relationships*

Table 10.7 shows the correlations between urban morphology metrics and levels of ecosystem service production for residential and industrial land use areas. It is obvious from Table 10.7 that there is one urban morphology metric in each land use type that

Table 10.6 continued.

c. Opportunities for cultural ecosystem services

		Water bodies	Unenclosed land	Woodland	Orn./park./rec.	Institutional	Industrial	Residential	Enclosed land	Horticulture	Extractive	Communications	Commercial
Water bodies	*2162*	-											
Unenclosed land	*2152*		-										
Woodland	*2147*			-									
Orn./park./rec.	*1586*				-								
Institutional	*1065*					-							
Industrial	*980*						-						
Residential	*978*							-					
Enclosed land	*976*								-				
Horticulture	*976*								NA	-			
Extractive	*976*								NA	NA	-		
Communications	*976*								NA	NA	NA	-	
Commercial	*976*								NA	NA	NA	NA	-

does not correlate with any ecosystem system. For residential areas, this is building density; whereas for industrial areas it is population density. These are also in contrast to correlations across all land use types, for which the impervious surface nLSI has the weakest correlations (Table 10.5). As was the case for analysis across all land uses, if the metrics that do not correlate with ecosystem service production are excluded, the direction of correlations is the same for all ecosystem services for a given metric.

In both residential and industrial areas, the proportion of impervious cover is the metric with by far the strongest relationship for reduction of storm water runoff, carbon storage, opportunities for cultural ecosystem services, and habitat provision. However, in residential areas the population density is a slightly stronger correlate for air pollution reduction and heat island mitigation than the proportion of impervious cover. In industrial areas, the proportion of detached houses has a relationship with air pollution reduction and heat island mitigation that is of similar strength to the proportion of impervious cover.

10.3.2.2. *Random forest analysis: importance of metrics and land use*

Figure 10.5 shows the relative importance of the urban morphology metrics and land use subtype to predicting the production levels of ecosystem services. Land use subtype accounts for no more than an average of 9% of the total predictive ability for any ecosystem service within either residential or industrial land uses, and as such is considered to be of little importance to levels of ecosystem service production.

As with the full dataset, building density, mean building size and the road LDI do not contribute significantly to the prediction of any ecosystem service (the maximum contribution of any of these metrics is an average of 8% across runs). The impervious surface nLSI also does not contribute more than 6% of the total predictive ability for any ecosystem service.

Table 10.7. Spearman's *rho* rank correlation coefficients for ecosystem services against urban morphology metrics for residential and industrial land uses. Yellows indicate weak correlations; darker greens and reds indicate stronger positive and negative correlations respectively. White background indicates results taken to be statistically non-significant at the Bonferroni-corrected threshold; grey background indicates results only significant at the non-corrected threshold.

		Air pollution reduction	Heat island mitigation	Runoff reduction	Carbon storage	Habitat provision
Residential	Proportion impervious cover	-0.42	-0.49	-0.77	-0.80	-0.66
	Building density	-0.08	-0.01	-0.01	0.07	-0.03
	Mean building size	-0.16	-0.26	-0.38	-0.49	-0.32
	Impervious surface nLSI	0.00	0.08	0.17	0.32	0.01
	Population density	-0.46	-0.52	-0.34	-0.36	-0.48
	Proportion detached houses	0.31	0.43	0.46	0.54	0.36
	Road LDI	-0.15	-0.18	-0.32	-0.42	-0.37
Industrial	Proportion impervious cover	-0.40	-0.46	-0.96	-0.92	-0.91
	Building density	-0.16	-0.17	-0.31	-0.34	-0.35
	Mean building size	-0.25	-0.28	-0.45	-0.42	-0.39
	Impervious surface nLSI	-0.23	-0.29	-0.62	-0.61	-0.65
	Population density	0.07	0.05	0.02	0.01	-0.01
	Proportion detached houses	0.41	0.46	0.33	0.28	0.32
	Road LDI	-0.16	-0.20	-0.12	-0.08	-0.22

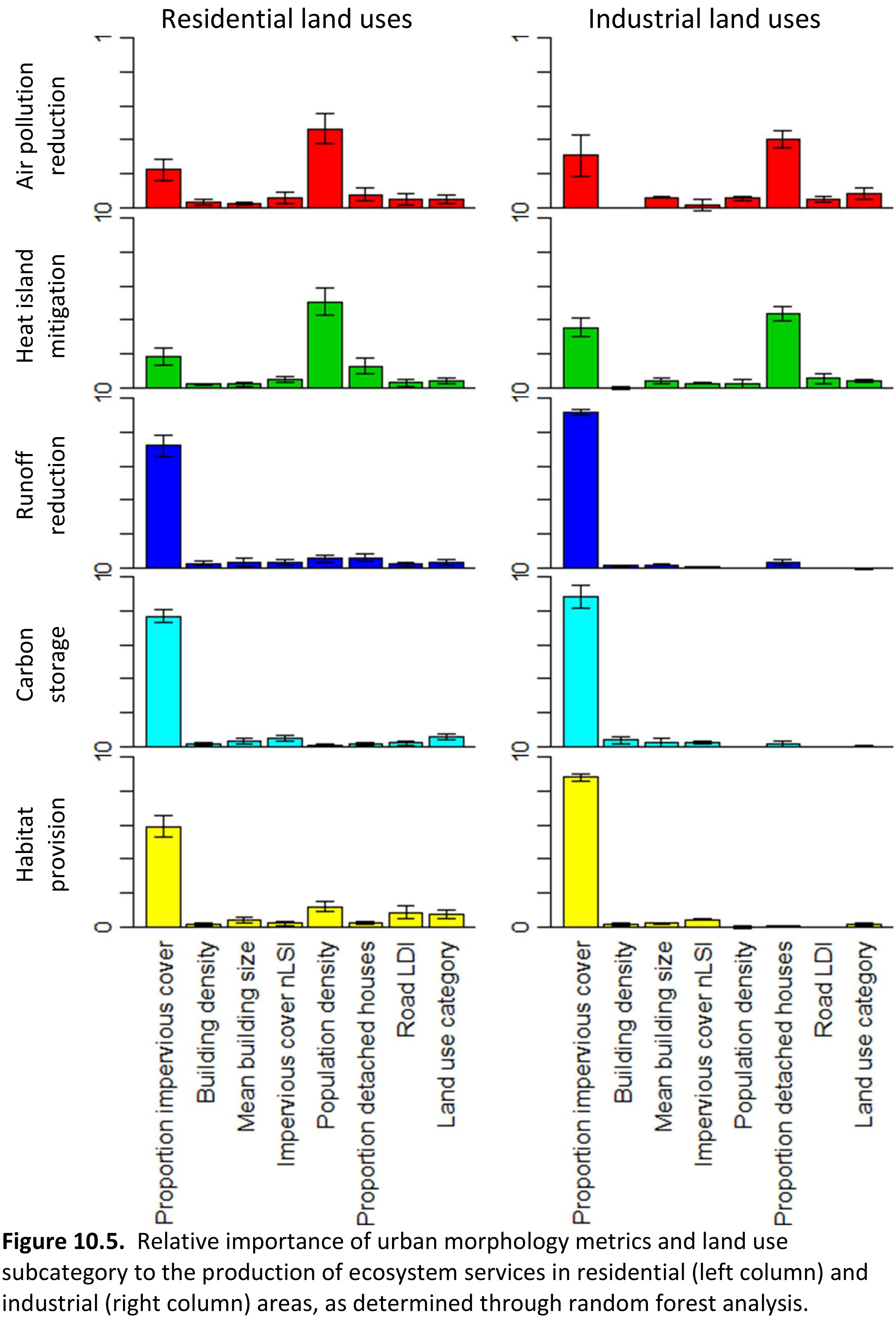


Figure 10.5. Relative importance of urban morphology metrics and land use subcategory to the production of ecosystem services in residential (left column) and industrial (right column) areas, as determined through random forest analysis. Variable importances are means from multiple runs, also using different data subsamples for residential land uses, with the results from each subsample scaled to sum to one for each ecosystem service. Error bars show the 95% confidence intervals across subsamples/runs.

For both residential and industrial land uses, as well as the whole dataset, the proportion of impervious surfaces is the most important metric for prediction of three ecosystem services: reduction of storm water runoff, carbon storage, and habitat provision. In the case of reduction of storm water runoff, contributions to predictive ability are similar and very high across all three datasets (averages of 72% for residential, 91% for industrial and 41% for full dataset). In contrast to the full dataset, no other metric contributes more than 6% for these ecosystem services.

For carbon storage, the proportion of impervious surfaces contributes averages to the total predictive ability of 77% for residential, 88% for industrial and 53% for the full dataset. In contrast to the full dataset, however, no other variables are particularly important, contributing no more than 6% for residential areas or 3% for industrial areas.

Habitat provision in industrial areas is predicted almost entirely by the proportion of impervious cover (88% of total predictive ability). In residential areas population density also has some predictive use (12%, with the proportion of impervious surfaces contributing 59%).

Reduction of air pollution and heat island mitigation show similar patterns of variable importance to each other within each analysis, but the patterns are different for residential areas, industrial areas and the full dataset (Figure 10.5,

Figure 10.3). In residential areas, population density is the most important predictor (47% for reduction of air pollution and 51% for heat island mitigation), followed by the proportion of impervious cover (23% and 18%) and, to a lesser degree, the proportion of detached houses (8% and 13%). In contrast, the proportion of detached houses is the most important predictor in industrial areas (41% and 44%) followed by the proportion of impervious cover (31% and 35%).

10.4. Discussion

10.4.1. Evaluation of methods and metrics

The analyses described in this chapter have produced a number of useful and informative findings. Before discussing these findings it is worth considering the limitations of the methodology in order to understand how the conclusions described in the following sections are drawn.

Many of the general patterns in the strength of relationships reflect the model input data and methods. For example, opportunities for cultural ecosystem services presumably shows lower correlations with metrics than the other ecosystem services (Table 10.5, Table 10.7) because this ecosystem service index is derived from land use, whereas the other ecosystem service models all use the land cover map as a data input, and the urban morphology metrics are also calculated from the land cover map. Similarly, in the random forest analysis land use is the most important predictor of opportunities for cultural ecosystem services (

Figure 10.3) because it is data associated with the land use map that is used to determine whether a location represents publicly accessible greenspace (Section 7.2). Although this makes this particular random forest result trivial, being obvious from the method of modelling, it does indicate that the variable importance algorithm is able to identify predictively valuable variables despite correlations between land use and urban morphology as well as subsampling of the dataset.

According to the rank correlations, the impervious surface nLSI shows the weakest correlations with ecosystem service production, and in many cases the relationships are not statistically significant (Table 10.5, Table 10.7). The other metrics all show considerably stronger relationships. In contrast, the results from the random forest analysis find that three other metrics are also unimportant: building density, mean building size and the road LDI (

Figure 10.3, Figure 10.5). None of these variables, or impervious surface nLSI, contributes more than 10% of the total predictive ability. This is in spite of some quite strong rank correlations between ecosystem service production and these metrics (Table 10.5, Table 10.7). These strong correlations perhaps arise from correlations between metrics (Appendix B.4.1), whereas random forests are able to account for correlated predictor variables (Strobl et al. 2007).

For these reasons the random forest analysis seems to be a better identifier of urban morphology metrics that are associated with ecosystem services than the correlation analysis, and therefore is given more weight in the following discussions. However, random forest analysis does not indicate the direction of relationships, so the results of the correlation analysis are used for this.

It has already been mentioned that four of the metrics do not appear to be reliable and important predictors of ecosystem services production (impervious surface nLSI, building density, mean building size, road LDI). In contrast, the proportion of impervious cover is the most valuable predictor variable for all ecosystem services except opportunities for cultural ecosystem services across all land use types, and for reduction of stormwater runoff, carbon storage and habitation provision in residential and industrial land use types; there are also particularly strong rank correlations here (Table 10.5, Table 10.7,

Figure 10.3, Figure 10.5).

This suggests that providing more greenspace within an urban area, regardless of the composition of that greenspace or the characteristics of the built environment, is of prime importance for the production of these ecosystem services. This is an unsurprising result, as the proportion of impervious cover is essentially the inverse of the proportion of the area representing greenspaces where ecosystem services are produced.

It is also unsurprising from a methodological point of view, given that the ecosystem service models (except opportunities for cultural ecosystem services) use proportional land cover data as a key input, with natural land cover type providing higher levels of the ecosystem service. Nevertheless, the random forest analysis has identified other urban morphology metrics that are associated with ecosystem service production levels. This shows that the analytical results are not simply a mirror of the modelling methods, and that there are associations – and absences of associations – between urban morphology and ecosystem service production that could not be easily guessed from the methods. For example, from the random forest analysis across all land uses, it appears that none of the ecosystem services are sensitive to variation in building density or size independently of the proportion of impervious cover. This is interesting as two of the models (reduction of air pollution and heat island mitigation) treat buildings differently from manmade surfaces, so it would be expected that building density and size – which together indicate the area of buildings – would be useful as predictors of ecosystem service production as well as the proportion of impervious surface. These less predictable relationships with other metrics that are worthy of discussion, as well as the stronger, more expected associations.

A final point to note here is that, when performing studies of this type, it is important to check for correlations between urban morphology. This is because, even though none of the pairs of metrics used here correlate strongly with each other in the study area (see Appendix B, Table , Table), the weak and moderate correlations that do exist were found to have a confounding effect on correlations between single metrics and levels of ecosystem services. This was reflected in the fact that even strong correlations between metrics and ecosystem services were not necessarily important in terms of permutation accuracy in random forest analysis (Table 10.5,

Figure 10.3; Table 10.7, Figure 10.5). This was found to be true across all land uses, and within two single land uses (residential and industrial). This should be a consideration for urban morphology studies using suites of metrics.

10.4.2. Air pollution reduction & heat island mitigation

Air pollution reduction and heat island mitigation show broadly similar patterns to each other in terms of which other variables are associated strongly with their levels, although these patterns are slightly different between all, residential and industrial land use types. Therefore the results from these ecosystem services are discussed together.

Across all land uses and within residential land uses, population density is the most important predictor of air pollution reduction and heat island mitigation (

Figure 10.3, Figure 10.5), with lower population density associated with higher ecosystem service production (Table 10.5, Table 10.7). The proportion of impervious cover, the proportion of detached houses and (for air pollution reduction only) land use category are also important.

The maps indicate that levels of air pollution reduction (Map 7) are lowest in the city centre, especially around the rivers, and also in the moorlands; and that the areas around the rivers and the moorlands both have low population density (Map 33). However, the population density is very high in some areas with quite low levels of air pollution reduction, and the population density is also low in the areas with high levels of reduction, suggesting an interaction with the other variables important to predicting levels of air pollution reduction. Levels of reduction are very low where the proportion of impervious cover is very high (Map 29). Where the proportion of households in detached or semi-detached houses is moderate to low (Map 34), there is in residential areas a negative relationship between population density (and the proportion of impervious cover) and the proportion of households in detached or semi-detached houses. In contrast, in areas with a high proportion of households in detached or semi-detached houses the relationship is negative (because few people live in the moorlands, compared to farmland areas).

The importance of land use to air pollution reduction may be at least partially because areas of coniferous woodland land cover, which have the highest rates of pollutant removal (Figure 3.4), obviously tend to coincide with areas designated as having the woodland land use (Table 2.12, Map 3). Thus, this land use class has some value in predicting where high levels of air pollution reduction occur. The large variation in predictive ability (see the confidence intervals in

Figure 10.3) can be accounted for by variation in the frequency with which these relatively uncommon land uses (Table 10.4) occur in different data subsamples. The nature of the land use associations are also discussed in Section 10.4.7.

Interaction effects can also explain the importance of population density to heat island mitigation, levels of which are lowest in areas of low population density in the city centre but highest in areas of low population density outside the urbanised area (Map 10, Map 33). Again, this is because the relationship between population density and heat island mitigation is negative where the proportion of households in detached or semi-detached households is low and vice versa. There is also a generally positive relationship between the proportion of impervious cover and heat island mitigation.

The patterns of variable importance for residential land uses are broadly similar between all land uses and residential land uses, except that in residential areas land use subtype is not an important predictor variable for air pollution reduction, and population density becomes even more important. This is perhaps because, in residential areas, population density correlates not only with the proportion of impervious cover but also the types of pervious surfaces present. For example, an area with low population density, in addition to having less impervious cover, may also have a greater proportion of woodland within the pervious areas than different sites with higher population density.

Industrial land uses, however, show a different pattern of variable importance. Population density is not important, while the importance of the proportion of impervious cover and the proportion of detached houses increases. The proportion of impervious cover is presumably important because of the increased space available for ecosystem services to occur. It is not however clear why the proportion of detached houses should be important.

It is interesting that reduction of air pollution and heat island mitigation show similar patterns of variable importance to each other within single land use analyses, but that the patterns are different between residential and industrial areas. This perhaps suggests that constraints on urban morphology imposed by particular land uses change the particular facets of morphology that have the greatest impact on the production of these ecosystem services.

Relationships between air pollution reduction and urban morphology do not appear to have been studied previously, so it is not possible to put these results in the context of existing knowledge beyond the basic hypothesis that land use and morphology patterns drive variation in levels of ecosystem service production (Alberti 2005). There are however three previous studies investigating heat island – urban morphology relationships. One of these was concerned with temperatures in street canyons in a tropical city, finding that narrower, deeper street canyons with shade provided by trees minimised the heat island effect as measured by the "physiologically equivalent temperature", a function of air temperature, mean radiant temperature, humidity and air movement (Johansson and Emmanuel 2006). Although this result is not directly comparable with those presented in this chapter, it seems to suggest that more compact urban morphologies (i.e. buildings closer together and high-rise) reduce the heat island effect in a tropical climate, which is in contrast to Sheffield: here, areas with low population density and a high proportion of detached houses have the lowest heat island effect. Johansson and Emmanuel suggest that shade and air circulation are important to the physiologically equivalent temperature; these are not factors that could be included in the model used here, and in addition a different measure of the heat island effect is used, which may explain the different results obtained here. Alternatively it may be due to the climate differences: it is possible that climate determines the relative importance of the different heat fluxes.

The second previous study of heat island – urban morphology relationships is an investigation of environmental and morphological features influencing the air temperature at various residential developments in sub-tropical Hong Kong (Giridharan et al. 2007). The urban morphology variables found to be important to minimise the heat island effect in this case were low sky view factor (low proportion of the total view from ground level comprising sky), high proportional cover of vegetation taller than 1m, and small thermal mass of buildings (as represented by the average height:floor area ratio). Thus again the pattern is for more compact designs with tall buildings and a lot of shade tend to have lower temperatures, perhaps due to the importance of shade and air circulation.

The third study was methodologically the most similar to the present investigation, using the same method to model surface temperatures and correlating against some of the same urban morphology measures, and using UK cities as study sites (Tratalos et al. 2007). High address density was found in Tratalos et al's study to be associated with greater heat island intensity, as was a low proportion of detached houses; building density and household density were not, however, significantly correlated. These results match those from the present study, which found population density (likely a correlate of address density) and the proportion of detached houses to be important for heat island mitigation, whereas building density was not (Section 10.3.1.2).

It is difficult to determine the extent to which these previous studies corroborate or contradict the findings in this chapter because of differences in methods and urban morphology metrics. Nevertheless the modelling methods used here are founded in well known theory (Tso et al. 1991), making it likely that the results show true patterns, even if they do not include all the factors necessary to give the full picture.

10.4.3. Stormwater runoff reduction

Across all land covers, reduction of stormwater runoff shows broadly similar patterns of variable importance to heat island mitigation, except that the proportion of impervious cover is more important than population density (

Figure 10.3). Thus the urban morphology metrics that are associated with high levels of runoff reduction are, in decreasing order of importance, low proportion of impervious cover, low population density and high proportion of detached houses. It is possible that there are interactive effects also occurring here, such that areas in the city centre with a very high proportion of impervious cover, low population densities/high proportions of detached houses are associated with low runoff reduction; while in areas with lower proportions of impervious cover, low population density/high proportion of detached houses are associated with high ecosystem service production. Certain types of land use types also appear to be weakly associated with runoff reduction levels, although the contribution of land use to prediction was not considered sufficient to warrant investigation with multiple comparison tests.

Within industrial and residential land uses, the proportion of impervious cover is the only important predictor variable (Figure 10.5). This suggests that within these individual land use types, storm water runoff reduction is increased by increasing the proportion of natural surfaces in an area regardless of the characteristics of the built environment and regardless of particular patterns of urban morphology associated with particular land use types. The contrast between this result and that from the analysis across all land uses may be indicative of consistent land use specific differences, which the analysis in Appendix B.4.3 suggests do exist, or may reflect patterns that arise from variation in urban morphology within other land use types not specifically analysed here.

Two previous studies have looked at urban morphology relationships with stormwater runoff. One of these examined how the design of individual lots of residential land could be changed to reduce stormwater runoff, as modelled by the same methods used in this thesis (Stone and Bullen 2006). Due to complete differences in the scale of land parcels studied and in urban morphology metrics used, the results cannot be compared; Stone and Bullen do however find that there is considerable scope for reduction of stormwater runoff with only modest changes to urban morphology that affect the proportion of impervious cover, and with no change in the size or density of residential lots. This corroborates the finding in this study that the proportion of impervious cover is the most important facet of urban morphology affecting runoff, especially within single land uses (residential and industrial).

The second study is again that of Tratalos et al. (2007), using the same methods and some of the same metrics as this thesis, and investigating UK cities. Tratalos et al. found that high address density (but not building or household density) and low proportion of detached houses are associated with levels of high stormwater runoff. Both population density and the proportion of detached houses are found to be important in this study, although less so than the proportion of impervious cover, which was not included by Tratalos et al. (2007).

10.4.4. Carbon storage

The proportion of impervious cover is strongly associated with carbon storage in both single land use analyses and across all land uses. Land use type also contributes a substantial portion of the total predictive ability of the random forest model across all land uses. Land use type is probably important for carbon storage prediction because the highest levels of carbon storage are found under peaty soils to the west of the study area (Figure 6.1, Map 4), which are also mostly classified as unenclosed land by the land use map (Map 3). This means that, as with the woodland land use for air pollution reduction, this specific land use category indicates relatively reliably where high carbon storage occurs.

A third variable that is only slightly important to carbon storage across all land use types is population density. This pattern might arise from the tendency of areas with lower population density to have larger gardens with more space available to plant trees, although this idea cannot be tested using the present datasets and could also simply be due to the pattern of lower population densities in areas with higher carbon storage in soils (Figure 6.1b, Map 16).

Carbon storage does not appear to have been studied previously in the context of urban morphology. It cannot therefore be said whether the results found here reflect situations found elsewhere.

10.4.5. Opportunities for cultural ecosystem services

In contrast to the other ecosystem services, land use type is the only variable important to the prediction of opportunities for cultural ecosystem services. As has already been discussed in Section 10.4.1, this is due to the methods used to identify areas of public greenspace: water bodies, unenclosed land and woodland provide the highest levels of the ecosystem services, as all land use subtype within these categories represent publicly accessible greenspace; while ornamental, parkland and recreational land uses and institutional land provide slightly lower levels as only some land use subtypes provide public greenspace. The fact that none of the urban morphology metrics contribute to the predictive ability of the random forest model indicates that none of them vary in a consistent manner with the land uses that provide public greenspace, compared to those that do not.

Two previous studies have investigated actual (as opposed to potential, i.e. opportunities for) cultural ecosystem services. One looked specifically at the relationship between physical activity in parks and land use diversity surrounding the parks, finding that low land use diversity was associated with higher levels of park use for physical activity (Kaczynski et al. 2009). The other investigated relationships between sense of community, walking, and several urban morphology measures (Wood et al. 2010). This second study found that low land use diversity was associated with higher sense of community, especially where commercial land uses are absent, and that sense of community was also correlated with leisurely walking activity. Thus it appears that low land use diversity around residences promotes at least some cultural ecosystem services. Unfortunately, none of the urban morphology metrics used in this study are comparable to land use diversity, meaning it is not possible to say whether this pattern is also observed for opportunities for cultural services in Sheffield.

10.4.6. Habitat provision

The proportion of natural cover in an area, i.e. the proportion not covered by impervious surfaces, is the main variable that is valuable to the prediction of the habitat provision index, both across all land uses and for residential and industrial land uses only (

Figure 10.3, Figure 10.5). Across all land uses, the impervious surface nLSI also plays a small role. Thus although the proportion of greenspace is by far the most important contributing factor to habitat provision, there is also some relationship between the shape of that greenspace (as a corollary of the shape of the built environment) and its ability to provide habitat for biodiversity. This is in spite of the non-significant rank correlation between habitat provision and the impervious surface nLSI (Table 10.5), and is likely due mathematically to relationships between this metric (measuring aggregation of the built environment) and the natural surface correlation length metric that is part of the habitat provision index (measuring extent of natural environment patches; see Appendix A.5.1.3). The third component of the habitat provision index is the standardised Shannon diversity index. The value of the standardised Shannon diversity index correlates highly significantly with both the proportion of natural cover (Spearman's $rho = 0.51$) and the natural surface correlation length (Spearman's $rho = 0.19$). Thus the importance of a diversity of natural land cover types is probably to some extent also captured in the proportion of impervious cover and impervious surface nLSI.

This result is corroborated by extensive empirical evidence that, in general, species richness decreases along a gradient from low to high levels of impervious cover (i.e. high to low levels of vegetative cover); and also with higher levels of habitat fragmentation, which is represented in the urban morphology metrics by the impervious surface nLSI, i.e. the distance that can be travelled before encountering an impervious surface (Alberti 2005, Garden et al. 2006, McKinney 2008, Schlesinger et al. 2008, Evans et al. 2009). These similarities indicate that the index of habitat provision is reasonably reliable in representing the aspects of the environment that influence biodiversity.

In contrast, in residential areas the impervious surface nLSI is not important, but the population density and road LDI do have a low level of importance. This result suggests that lower population density and lower road LDI both improve levels of habitat provision independently of the proportion of impervious cover. In the case of the road LDI this relationship probably arises from the fragmentation effect that transport infrastructure has on the natural environment. For population density, the effect may be due to larger houses (where population densities are lower) tending also to have larger gardens.

Road densities and numbers of intersections have also been found in empirical studies to disrupt biodiversity (Alberti 2005, Garden et al. 2006); and population density is likely to correlate with levels of disturbances (e.g. driving, dog walking and fly tipping) that affect more sensitive species (Ehrenfeld 2004, McKinney 2006, Schlesinger et al. 2008). Again, the similarity between this study and empirical results indicates that the index of habitat diversity is an appropriate way to represent levels of biodiversity.

10.4.7. Land use versus urban morphology

The previous sections discussed the fact that land use is the only important predictor of opportunities for cultural ecosystem services, as a result of the way this index is calculated. Land use type is also associated with air pollution reduction and carbon storage, contributing 25-26% of the total predictive ability of the random forest model (

Figure 10.3). The multiple comparison tests in Section 10.3.1.3 give further insight to the nature of these relationships.

Figure 10.4 shows that the average rank of ecosystem service production levels tend to be similar between air pollution reduction and carbon storage; these patterns are further confirmed by the similarities between Table 10.6a and Table 10.6b. The exception is water bodies, the average rank of which is far lower for carbon storage than for air pollution reduction. This is because carbon in soil under water bodies is not included in the carbon storage model.

The grouping of intensive building land uses (residential, institutional, communications, industrial and commercial), which provide relatively low levels of air pollution reduction and carbon storage (Table 10.6a,b), is also found with several urban morphology metrics (Appendix B.4.3). The other, less intensively built-up land uses (enclosed land, horticulture, extractive, and ornamental, parkland and recreational land uses), provide higher levels of ecosystem services and a different urban morphology profile. This suggests that it may be the particular combination of metric values that makes land use a good predictor variable here: values of the urban morphology metrics covary, often strongly, with land use type (Appendix B, Figure , Table), which is perhaps because the use to which a parcel of land is designated to some extent constrains the patterns of urban morphology that are suitable. Some evidence supporting this idea comes from differences in patterns of correlations between metrics in different land use types. Differences in patterns of correlations between metrics and ecosystem services, and in the relative importance of the metrics to ecosystem service production, between different land use types might also be driven by such constraints. These constraints may apply for facets of urban morphology not described by the metrics used here, as well as those that were; i.e. the importance of land use might also reflect the inadequacy of the metrics used here for explaining variation in ecosystem service production.

An alternative or complementary explanation is that the land cover profile found in these land uses is particularly good at providing these ecosystem services. For example, woodlands are by far the best land covers for removing air pollutants (Figure 3.4), and the woodland land covers are present over an average of 86% of the woodland land use (Table 2.11). Woodlands also provide the highest levels of carbon storage (Table 10.6b), despite a seemingly lower proportion of woodlands being found on peaty soils compared to unenclosed land (Map 3, Map 4). Enclosed and unenclosed land, horticulture, extractive and ornamental/parkland/recreational land uses also provide quite high levels of this ecosystem service, suggesting that these land uses are often found on soils with high carbon contents, and have combinations of land cover that store high levels of carbon.

Residential and institutional land also has higher levels of both carbon storage and air pollution reduction than other intensive land uses, with pairwise comparisons being significant in many cases (Table 10.6a,b). This is probably because of the inclusion of more greenspaces such as gardens and roadside verges in areas of these land uses.

The rank average of opportunities for cultural ecosystem services (Table 10.6c), however, shows a very different pattern due to its dependence on land accessibility rather than just the physical infrastructure. The patterns here directly reflect the land uses that represent publicly accessible greenspace.

When the contribution of land use subtype to prediction of ecosystem service production in residential or industrial land uses is analysed, however, land use is insignificant is comparison to urban morphology (Section 10.3.2.2). The small contribution that land use subtype does make is probably due to the tendency to find certain land use subtypes in particular parts of the study area; for example, light metal trades and other industry tend to be found closer to the city centre than heavy metal trades, and terraced housing is common closer to the city centre whereas detached and semi-detached housing become more common further away (Figure 10.2). It is possible that land use subtype would become more important if more categories were included separately (i.e. not aggregated into "other"), but the data could not be analysed this way due to sample sizes.

10.4.8. Synthesis: planning recommendations

The analyses in this chapter have found strong evidence for relationships between urban morphology, land use and ecosystem services, meaning that there is scope for improving ecosystem service production by consideration of the way in which urban developments are designed. The relationships generally corroborate evidence from previous studies, where such studies exist; but the methodological differences make it difficult to make reliable comparisons (Section 10.4.2). Where the models match empirical evidence, they may provide insight into the mechanisms driving patterns seen in observational studies; the similarities also suggest that the models provide reasonable representations of real world processes.

The relationships are summarised in Table 10.8, which in the upper part of the table shows the relative importance of the urban morphology metrics and land use type/subtype (for analyses across all and single land uses respectively) by averaging across all six ecosystem services. The lower part of Table 10.8 summarises the importance of the metrics relative to each other, ignoring the relative importance of these metrics to land use type/subtype, and also shows the directions of relationships between the metrics and ecosystem service production.

It is immediately clear that urban morphology is far more important than land use type as a predictor of ecosystem service levels. This is even more the case if opportunities for cultural ecosystem services is excluded from all land use types: land use type then becomes responsible for only 12% of the average total predictive ability.

Importantly, the directions of correlations for each urban morphology metric are almost exclusively the same for all ecosystem services (Table 10.5, Table 10.7); the two exceptions are both for weak correlations that are not considered statistically significant. In most cases the directions are also the same regardless of whether single or all land use types are considered (Table 10.8). The exceptions are population density, which is negatively correlated with all ecosystem services for all land uses and residential areas, but the non-significant relationships for industrial areas are mostly positive (although

Table 10.8. The average relative variable importance (from random forest permutation accuracy analysis) of urban morphology metrics and land use type (across all land use types) or subtype (for residential and industrial land uses) across all ecosystem services. Lower part of table shows average relative importance of metrics scaled to sum to one regardless of importance of land use type/subtype. -, + and 0 indicate the direction of relationships, as determined by rank correlation analysis (Table 10.5, Table 10.7), i.e. increasing the value of the metric decreases, increases and has no simple/linear effect on levels of ecosystem service production respectively. Pale yellows indicate low importance and dark greens indicate high importance.

	All land use types	Residential	Industrial
Urban morphology metrics	0.73	0.95	0.97
Land use type/subtype	0.27	0.05	0.03
Sum	*1.00*	*1.00*	*1.00*
Proportions within urban morphology metrics:			
Proportion impervious cover	0.50 (-)	0.53 (-)	0.69 (-)
Building density	0.03 (-)	0.03 (0)	0.02 (-)
Mean building size	0.03 (-)	0.04 (-)	0.04 (-)
Impervious cover nLSI	0.04 (0/+)	0.05 (0/+)	0.03 (-)
Population density	0.26 (-)	0.25 (-)	0.02 (0)
Proportion detached houses	0.12 (+)	0.07 (+)	0.19 (+)
Road LDI	0.03 (-)	0.05 (-)	0.02 (-)
Sum	*1.00*	*1.00*	*1.00*

small); and the impervious surface nLSI, which is moderately strongly negatively correlated with ecosystem services in industrial areas, and usually positively (although mostly not significantly) correlated in residential areas and across all land use types.

The fact that the correlations are in the same direction is important because it means that ecosystem service trade offs are unlikely, i.e. promoting the production of one service is unlikely to reduce the production of another, especially if the urban morphology changes are focused on particular land use types. It is therefore possible to make urban morphology recommendations that are likely to improve the production of all the ecosystem services modelled in this thesis. Specifically, these recommendations are: to build impervious cover over a smaller proportion of land; to build fewer, smaller buildings; to reduce the population density and increase the proportion of the population living in detached and semi-detached houses; to build fewer roads; and to lay out the areas of impervious cover in as compact a way as possible.

However, the rank order of the strength of correlations within an urban metric morphology and between ecosystem services differs between metrics. This means that implementing a change in the urban morphology factor with the strongest correlation with a given ecosystem service may have the highest chance of actually improving that ecosystem service, but may be less likely to influence the levels of other ecosystem services than changing other morphology factors. In other words, the security of obtaining ecosystem service improvements on the ground if an urban morphology metric is changed varies between ecosystem services.

Nevertheless, Table 10.8 can be used to make recommendations about urban design priorities to improve levels of ecosystem service production in general. Clearly, the single best improvement to ecosystem service production can be made by reducing the proportion of impervious cover in an area. This appears to hold true regardless of the land use type(s) analysed. Reducing the population density, especially by increasing the proportion of housing that is low density (i.e. detached and semi-detached), also makes a good contribution to improving levels of ecosystem services.

The other metrics do not appear to be priorities for improving ecosystem services (Table 10.8). Two of the metrics, namely the building density and mean building size, never contribute more than 6% of the total predictive ability for any ecosystem service (

Figure 10.3, Figure 10.5). The impervious surface nLSI and the road LDI are similar, although the former contributes 8% to the prediction of habitat provision levels across all land use types, and the latter 9% to habitat provision levels in residential areas.

However strong the general patterns, it remains important to be explicit about the effects of urban morphology on individual ecosystem services, because in some cases the recommendations would be quite different. Table 10.9 shows recommendations for improvement of the production of each ecosystem service. As well as highlighting differences in recommendations between ecosystem services, Table 10.9 shows how recommendations can vary between different land uses. Differences in recommendations may partially reflect the constraints imposed on morphology by particular land uses. Thus they are particularly useful because they show how to optimise the chances of increasing levels of ecosystem service production, within the range of urban morphologies that are practical for a particular land use type.

10.5. Summary

This chapter has used the output of the ecosystem services models that were the focus of Chapters 3 to 9, in combination with the metrics of urban morphology

Table 10.9. Recommendations to improve levels of production of individual ecosystem services. 1 indicates highest priority; where increasing the cover of particular land uses is recommended, the first listed is the priority land use. ? indicates that the effectiveness of implementing this recommendation is likely to be small.

	All land uses	Residential	Industrial
Air pollution reduction	1. Reduce population density 2. Reduce proportion of impervious cover 3. Increase cover of: woodland, horticulture, enclosed land, extractive, water bodies	1. Reduce population density 2. Reduce proportion of impervious cover 3. Increase proportion of households in detached/semi-detached houses	1. Increase the proportion of households in semi/detached houses 2. Reduce population density
Heat island mitigation	1. Reduce population density 2. Increase proportion of households in detached/semi-detached houses 3. Reduce proportion of impervious cover	1. Reduce population density 2. Reduce proportion of impervious cover 3. Increase proportion of households in detached/semi-detached houses	1. Increase the proportion of households in detached/semi-detached houses 2. Reduce population density
Runoff reduction	1. Reduce proportion of impervious cover	1. Reduce proportion of impervious cover	1. Reduce proportion of impervious cover
Carbon storage	1. Reduce proportion of impervious cover 2. Increase cover of: woodland, enclosed land, horticulture, unenclosed land, extractive, ornamental/parkland/ recreational 3. Reduce population density	1. Reduce proportion of impervious cover	1. Reduce proportion of impervious cover
Cultural opps.	1. Increase cover of: water bodies, unenclosed land, woodland, ornamental/ parkland/recreational	-	-
Habitat provision	1. Reduce proportion of impervious cover 2. Decrease nLSI?	1. Reduce proportion of impervious cover 2. Reduce population density 3. Reduce weighted road density index?	1. Reduce proportion of impervious cover

introduced in this chapter, to develop an understanding of the relationship between urban morphology and ecosystem services in Sheffield and the way in which these factors interact with land use. The results of the analysis performed in this chapter have been used in Section 10.4.8 to develop two sets of recommendations for improving levels of the production of ecosystem services by designing areas with particular morphologies.

The first set of recommendations is for improvement of ecosystem service production in general. The recommendations are to reduce the proportion of impervious cover and to reduce the population density by building more low density housing.

The second set of recommendations is targeted toward individual ecosystem services. The recommendations vary between services and land use types, and are summarised in Table 10.9; they variously include reducing population density, the proportion of impervious cover, and the proportion of the population living in low density housing types; and increasing the cover of particular land use types.

These recommendations show that it is possible to use the models implemented in this thesis to inform urban planning for ecosystem service optimisation, thus satisfying the fifth research aim listed in Section 1.4.

11. Socioeconomic conditions

11.1. Introduction

> "Currently,…it is rare to find a very poor urban community that does not face serious environmental health hazards."

This quotation, taken from the Urban Systems chapter of the Millennium Ecosystem Assessment Global Assessment Report, is a reference to the fact that the residents of lower income cities and neighbourhoods tend to suffer more severe threats to their welfare due to environmental degradation (Millennium Ecosystem Assessment 2005c). In other words, human well-being suffers as a consequence of over-burdening of the infrastructure that provides ecosystem services. The Global Assessment Report goes on to say (Millennium Ecosystem Assessment 2005c):

> "Urban development can easily threaten the quality of the air, the quality and availability of water, the waste processing and recycling systems, and many other qualities of the ambient environment that contribute to human well-being. Certain groups (such as low income residents) are particularly vulnerable."

Most of the evidence cited to support these assertions, however, comes from whole-city scale analyses. There seem to have been comparatively few within-city scale studies. It is nevertheless apparent from the existing studies (summarised in Table 11.1) that there is, in general, a positive relationship between socioeconomic status and ecosystem service production. Although some studies find no relationship, it is almost never the case that more deprived groups live in areas with greater production of ecosystem services. The exception to this is access to parks or public greenspace in some cities (similar to the opportunities for cultural ecosystem services measured here); although in at least some cities, neighbourhoods with little public greenspace tend to have more private greenspace i.e. gardens (Barbosa et al. 2007).

This relationship is important, because of the very well known relationship between many measures of socioeconomic status and health: indicators of health improve with socioeconomic status, not only below the poverty line but at all socioeconomic strata

Table 11.1. Summary of previous studies investigating relationships between socioeconomics and the ecosystem services modelled in this study.

Ecosystem service(s) & location	Methodological details	Results
Air pollution, Vancouver (Marshall et al. 2009)	Nitrogen oxide and ozone concentrations modelled and compared to income.	Higher income areas have lower nitrogen oxide and ozone concentrations.
Air pollution, Los Angeles (Ponce et al. 2005)	Air pollution from traffic modelled. Socioeconomic status determined by unemployment/uptake of social assistance.	High neighbourhood socioeconomic status is associated with lower exposure to air pollution from traffic.
Air pollution, Helsinki (Rotko et al. 2000)	Fine particulate matter exposure measured by personal samplers. Socioeconomic status determined by occupation type.	Lower socioeconomic groups are exposed to double the concentrations of fine particular matter as higher groups (although this is not due to variation in outdoor concentrations).
Air pollution, Helsinki (Rotko et al. 2001)	Nitrogen dioxide exposure measured by personal samplers. Socioeconomic status determined by occupation type.	No relationship observed between nitrogen dioxide exposure and socioeconomic group.
Air pollution, Hamilton, Ontario (Buzzelli and Jerrett 2004)	Particulate matter directly measured and compared to demographic composition.	Areas with many Latin Americans tend to have high air pollution; the opposite is true for Asian Canadians.
Heat island intensity, Phoenix (Buyantuyev and Wu 2010)	Surface temperature as determined by remote sensing compared to census block data including income.	Less affluent neighbourhoods experience higher temperatures, with a greater temperature difference during the hottest months.
Heat island intensity, Phoenix (Harlan et al. 2006)	Heat stress modelled from climate variables in eight neighbourhoods and compared to income, ethnic and age composition, education.	Residents of less affluent neighbourhoods experience greater heat stress, especially during heatwaves.
Heat island intensity, Philadelphia (Johnson and Wilson 2009)	Analysis of socioeconomic data of people who died from heat-related causes during an extreme heat event.	Poverty is the strongest predictor of risk of death during extreme heat events.
Access to public greenspace, Sheffield (Barbosa et al. 2007)	Distance from each address to nearest public greenspace and municipal park calculated from road network. Socioeconomic status from Mosaic geodemographic database.	Neighbourhoods with many elderly and/or less affluent people tend to have better access to public greenspace and municipal parks.

Table 11.1 continued.

Ecosystem service(s) & location	Methodological details	Results
Access to parks, Phoenix (Cutts et al. 2009)	Census block data on ethnicity, poverty and under 18s compared to parks within 0.25 miles.	Areas with many Latin Americans and African Americans have good park access. Areas with many young people have poor park access.
Access to parks, New York City (Maroko et al. 2009)	Census block data on ethnicity, poverty and education levels compared to GIS layer of park acreage density within one mile.	Parks are not equally distributed across the city, but distribution does not correlate with ethnicity, poverty or education levels.
Heat island, stormwater runoff & habitat provision, UK (Tratalos et al. 2007)	Modelled surface temperature and runoff and eight metrics representing habitat provision compared to proportion of population in higher socioeconomic groups in inner, middle and outer neighbourhoods in each of five cities.	One habitat metric (proportion of tree cover over gardens/greenspace) is higher in neighbourhoods with more affluent residents. No other significant relationships.
Biodiversity, Phoenix (Hope et al. 2003)	Plant diversity measured by field survey and compared to socioeconomic variables including median family income at 204 sites.	Plant diversity is twice as high in neighbourhoods with above average median income as in those below the average.
Biodiversity, Phoenix (Kinzig et al. 2005)	Diversity of birds (point counts) and plants (surveys) measured in sixteen neighbourhoods with a park within 0.5 miles and compared to median family income.	Neighbourhood median income correlates strongly with neighbourhood plant diversity (but weakly with park plant diversity), and moderately with avian diversity.
Biodiversity, Chicago (Loss et al. 2009)	Point surveys of bird diversity at 42 neighbourhoods compared to per capita income.	Total avian diversity was not related to income, although diversity of native species was higher in less affluent areas.
Biodiversity, Leipzig (Strohbach et al. 2009)	Breeding bird survey data compared to census data including unemployment and income.	Neighbourhoods with more high income households tend to have greater avian diversity.

(Kaplan et al. 1987, Smith 1998, Adler and Ostrove 1999). The range of health problems that correlate with socioeconomic status is wide, including (in the US) heart disease, arthritis, epilepsy, anaemia, neural tube defects, tuberculosis, injury and low birth rate; hospital stays and days of restricted activity due to ill health are also more frequent amongst people of lower socioeconomic status (Kaplan et al. 1987). It is believed that the most obvious reasons for this relationship, i.e. inadequate medical care and nutrition, hazardous living circumstances etc., are insufficient to explain the whole pattern, as the relationship still holds when comparing amongst more affluent groups, and because some of the health issues are unlikely to be caused by these types of influences (Kaplan et al. 1987, Adler and Ostrove 1999). It is therefore possible that low levels of ecosystem services are a contributor to poor health amongst people of lower socioeconomic status. Even if this is not the case, it nevertheless remains as a welfare issue in addition to the health issues.

The purpose of this chapter is to investigate spatial relationships between socioeconomic variables and ecosystem service production in Sheffield. The ecosystem service models described in earlier chapters of this thesis provide an opportunity to undertake analyses of a different scope to those shown in Table 11.1, including one ecosystem service that appears not to have been investigated previously in relation to local socioeconomics (carbon storage), and also including more ecosystem services than have previously been investigated in a single study (or by separate studies in a single city). The analyses undertaken in this chapter aim to investigate whether the generalisation that the socioeconomic status of a neighbourhood correlates with levels of ecosystem services also holds true for Sheffield; and to identify which sectors of Sheffield's society are the most deprived in terms of ecosystem services. This knowledge is valuable if issues of environmental justice are to be addressed, and is a pre-requisite to attempts to improve geodemographic inequities.

11.2. Methods

11.2.1. Socioeconomic datasets

The UK's Office for National Statistics provides a central point for the collation and dissemination of statistics and analysis for the facilitation of decision making and accountability (Office for National Statistics, no date). A range of social and economic statistics are produced (e.g. population and demographics, government and economic activity, statistics about neighbourhoods and families), which are publicly disseminated at a variety of spatial scales, from national to Output Area level (Office for National Statistics, no date).

The Office for National Statistics' Neighbourhood Statistics website (http://www.neighbourhood.statistics.gov.uk/dissemination/; accessed 17/03/2010) was used to explore the datasets available at suitably small spatial scales in order to identify those likely to be most useful for the present purpose. Three datasets were identified on the basis that they each synthesise several aspects of socioeconomic conditions. One of these datasets, the Index of Multiple Deprivation, did not produce any significant findings upon analysis, and is therefore not presented here. The other two datasets are introduced below, with further relevant details given in Appendix C. The analyses in this chapter were undertaken at Output Area-based scales, as Output Areas are constructed to be maximally socially homogeneous (Section 2.3.3) and thus provide the ideal unit for socioeconomic investigations.

The following sections introduce the datasets and discuss the spatial patterns that they show. Different methods are required to analyse the relationships between each of these datasets and ecosystem services; the methods used for analysis are also described. Finally, the datasets are cross-tabulated to facilitate interpretation of the two analyses.

11.2.2. Approximated social grade

The social grading system used here is that of the Market Research Society, which is correlated with both media and consumer behaviour and affluence (National Readership Survey, no date). There are six social grades, named A to E (with two C grades), which are described in Table 11.2. Grades A to C1 represent decreasing seniority in non-manual occupations; grades C2 and D represent manual work; and grade E represents dependence on state welfare with no regular job income.

The social grade of a household is based on the occupation of the chief income earner, and is traditionally established by means of a detailed interview (National Readership Survey, no date). Although social grade is not actually based on income, households of “higher” grades do tend to have higher income (National Readership Survey, no date).

An algorithm has been developed to approximate social grade from the demographic data collecting in the census, and has been used to estimate the social grade of every household in the UK using the 2001 Census with reasonable accuracy (Market Research Society, no date). It is this approximated grade, aggregated to Output Area level, which is used here. The census approximation algorithm aggregates grades A and B, leaving a total of five categories.

Table 11.2. Social grades as defined by the Market Research Society, and the percentage of the population in each grades in 2008. Sources: Market Research Society (2006), National Readership Survey (no date).

Grade	Occupation	% of population (2008)
A	Higher managerial, administrative and professional (e.g. senior managers in business/commerce; top level civil servants). Also retired people, previously grade A, and their widows.	4
B	Intermediate managerial, administrative and professional (e.g. qualified middle management executives in large organisations; principle officers in local government and civil service; top management/owners of small business, educational and service establishments). Also retired people, previously grade B, and their widows.	23
C1	Supervisor, clerical and junior managerial, administrative and professional (e.g. junior management; owners of small establishments; all others in non-manual positions). Varied responsibilities and educational requirements. Also retired people, previously grade C1, and their widows.	29
C2	Skilled manual workers and manual workers with responsibility for other people. Also retired people, previously grade C2, with pensions from job; and widows if receiving their job's pension.	21
D	Semi-skilled and unskilled manual workers, apprentices and trainees to skilled workers. Also retired people, previously grade D, with pensions from job; and widows if receiving their job's pension.	15
E	Casual and lowest grade workers without a regular income, unemployed (for at least six months), or dependent long-term on state benefits only (e.g. due to sickness or old age).	8

11.2.2.1. Spatial patterns

Maps of the proportion of the population belonging to each approximated social grade are shown in Appendix D:

- Approximated social grade AB – Map 36
- Approximated social grade C1 – Map 37
- Approximated social grade C2 – Map 38
- Approximated social grade D – Map 39
- Approximated social grade E – Map 40

There is a clear tendency for people of different social grades to live in different areas of Sheffield, with quite a strong east-west divide between the most and least affluent communities. Non-manual workers, especially more senior non-manual workers (grade AB), make up a far smaller proportion of the total population in the east of the urbanised area than they do elsewhere (Map 36, Map 37). In particular, there is a cluster of Output Areas in the southwest suburban area with a high proportion of the population in grade AB, while people of grade C1 are particularly likely to live in more rural regions as well as in this cluster.

The distribution of Output Areas with high proportions of manual workers (grades C2 and D) shows a contrasting pattern. Skilled and responsible manual workers (grade C2) are quite widely distributed amongst areas across the whole of Sheffield except for the cluster with high frequencies of grade AB, with the highest proportions occurring in suburbs to the north and east (Map 38); while unskilled manual workers (grade D) less frequently live in rural Output Areas and are particularly likely to live in the eastern half of the urbanised area (Map 39).

People dependent on state welfare (grade E) are also widely distributed, showing large variation in proportion of Output Area populations and with little spatial pattern; although high proportions are more commonly found in the urbanised region than elsewhere (Map 40). There are a few Output Areas where more than 60% (and up to 84.7%) of the population is of grade E, and these are scattered throughout Sheffield. However, they typically comprise less than 20% of the population in the western half of the study area.

11.2.2.2. Ecosystem service – approximated social grade rank correlations

Spearman's rho rank correlation coefficient was used to test the strength of associations between ecosystem service production and the proportion of Output Area populations in each social grade. The method used was the same as described in Section 9.3.2. The approach to statistical inference described in Section 9.2 was also used in this and the other analyses presented in this chapter, i.e. the use of a significance threshold of $\alpha = 0.001$ to account for spatial autocorrelation, and presentation of both Bonferroni corrected and uncorrected significance levels. In this case the Bonferroni corrected significance threshold was $p < \frac{0.001}{30} = 3.33 \times 10^{-5}$.

11.2.3. Area classification of Output Areas

The second variable used to indicate socioeconomic conditions is the area classification of Output Areas based on 2001 Census data. This scheme classifies each

Output Area into a single hierarchical categorical variable according to many demographic characteristics of the people who live there (Vickers and Rees 2007). The classification was devised using a clustering analysis approach with the "minimum possible number of variables that satisfactorily represent the main dimensions of the 2001 census", using only data from that census (Vickers and Rees 2007).
The result of this process is 21 categories each distinguished by a set of traits that diverge from the national average in distinctive ways.

Table 11.3 shows a summary of each of the groups, and their distinguishing variables. Further information and full profiles for each of the categories are provided by the Office for National Statistics (2005). All of the categories are well represented within Sheffield, with 22-222 Output Areas each, except the three Countryside categories, which are each represented by less than ten Output Areas.

11.2.3.1. Spatial patterns

The spatial distribution of area classification categories in Sheffield is shown in Map 41. There is a variable degree of scatter in the distribution of different groups and supergroups, although there is a broad tendency towards clustering both for supergroups and groups within supergroups.

The most obvious pattern is that countryside Output Areas (supergroup 3) occur mainly in the western half of the study area, outside of both the city centre and other "satellite" populated areas. Two supergroups are found almost exclusively in the city centre: supergroup 2 (city living) and supergroup 7 (multicultural). Supergroup 2 occurs mainly in a single cluster in the western half of the urban core, in areas of particularly high population density. Within this supergroup, group 2a (transient communities) and group 2b (settled in the city) are spatially segregated. The distribution of multicultural Output Areas is slightly wider, occupying most of the remainder of the urban core and also stretching out to the east of the study area. Again, within this supergroup, Asian communities (group 7a) and Afro-Caribbean communities (group 7b) tend to occur in clusters. Output Areas belonging to supergroup 1 (blue collar communities) occur mainly in the northeast half of the study area, in areas of quite high population density, although scattered both close to the urban core and in more suburban regions. Older blue collar workers (group 1c) tend to be found further out from the city centre, although terraced and younger blue collar Output Areas (group 1a and 1b) are found throughout. The prospering suburb Output Areas (supergroup 4)

Table 11.3. Area classification of Output Area categories. First column shows supergroup names. No. shows the number of Output Areas in the category in the study area. Distinguishing traits columns show variables, defined in Table , with range-standardised values that are at least 0.15 above or below the UK mean. Category names are determined from the whole cluster profiles, which are not shown here. (Source: Office for National Statistics 2005, Vickers and Rees 2007)

	Group	No.	Distinguishing traits	
			Far below national average	**Far above national average**
1 Blue collar communities	**1a** Terraced blue collar	73	No central heating Rent (private) Detached housing HE qualification All Flats Born outside the UK	Terraced housing Rent (public)
	1b Younger blue collar	222	Detached housing HE qualification All flats	Lone parent household No central heating Terraced housing Rent (public)
	1c Older blue collar	106	All flats	Rent (public)
2 City living	**2a** Transient communities	22	Detached housing Households with non-dependent children Terraced housing Age 5-14 2+ car household Working part-time Economically inactive looking after family Rooms per household Mining/quarrying/construction employment Age 0-4 Lone parent household	Financial intermediation employment No central heating HE qualification Public transport to work Single person household (not pensioner) Born outside the UK Rent (private) All flats
	2b Settled in the city	88	Detached housing Households with non-dependent children	HE qualification Born outside the UK Rent (private) All flats
3 Countryside	**3a** Village life	9	Public transport to work Population density All flats	Agriculture/fishing employment Detached housing
	3b Agricultural	2	Population density Terraced housing All flats Public transport to work Rent (public)	2+ car household Work from home Detached housing Agriculture/fishing employment
	3c Accessible countryside	8	Rent (public) Population density Public transport to work	2+ car household Agriculture/fishing employment Detached housing

Table 11.3 continued.

	Group	No.	Distinguishing traits	
			Far below national average	Far above national average
4 Prospering suburbs	**4a** Prospering younger families	37	All flats Rent (public) No central heating Single pensioner household Age 65+ Terraced housing	2+ car household Detached housing
	4b Prospering older families	73	Terraced housing All flats Rent (public) No central heating Rent (private) Single person household (not pensioner) Lone parent household	2+ car household Detached housing
	4c Prospering semis	142	All flats Rent (public) Terraced housing Rent (private)	(none)
	4d Thriving suburbs	95	Terraced housing Rent (public) No central heating	2+ car household Detached housing
5 Constrained by circumstances	**5a** Senior communities	72	2+ car household Detached housing Age 5-14 Age 0-4 Rooms per household Economically inactive looking after family	Age 65+ Single pensioner household Rent (public) All flats
	5b Older workers	183	Detached housing	All flats Rent (public)
	5c Public housing	66	Detached housing 2+ car household HE qualification Rent (private)	Public transport to work Unemployed Lone parent household All flats Rent (public)

are, unsurprisingly, found away from the city centre but not in rural areas. They are most common in the south and west suburban regions, and typically have relatively low population densities. Group 4b (prospering older families) tend to be found in a cluster in the southwest, while the other groups do not seem to show any particular clustering. Output Areas described as constrained by circumstances (supergroup 5) show a similar distribution to those in supergroup 1 (blue collar workers), but with additional scatter throughout all except rural regions. Output Areas with typical traits (supergroup 6) are, unsurprisingly, also found throughout the study region, but not in rural areas.

Table 11.3 continued.

	Group	No.	Distinguishing traits	
			Far below national average	Far above national average
6 Typical traits	**6a** Settled households	112	All flats Rent (public)	Terraced housing
	6b Least divergent	76	(none)	(none)
	6c Young families in terraced homes	103	Detached housing Rent (public)	Rent (private) No central heating Terraced housing
	6d Aspiring households	55	Rent (public)	(none)
7 Multicultural	**7a** Asian communities	146	Detached housing	No central heating Public transport to work Rent (private) Terraced housing Born outside the UK Black African, Black Caribbean or Other Black Indian, Pakistani or Bangladeshi
	7b Afro-Caribbean communities	54	Detached housing 2+ car household	Rent (private) Unemployed Indian, Pakistani or Bangladeshi Public transport to work Born outside the UK Rent (public) All flats Black African, Black Caribbean or Other Black

11.2.3.2. Ecosystem service – area classficiation analysis of variance and pairwise comparison tests

Kruskal-Wallis rank analyses of variance and post hoc pairwise comparison testing of all pairs, using Mann-Whitney U tests, were used to identify area classification groups between which levels of the production of each ecosystem service differed. The methods used were the same as those described in Section 10.2.5. Groups within supergroup 3 (i.e. countryside) were aggregated in order to have an adequate sample size. The results of the Kruskal-Wallis tests were highly significant for all ecosystem services. Consequently pairwise comparison testing was performed for all ecosystem services.

11.2.4. Dataset cross-tabulation

This section details how the socioeconomic variables intersect spatially, and is included in order to aid interpretation of the results from the analysis of the different

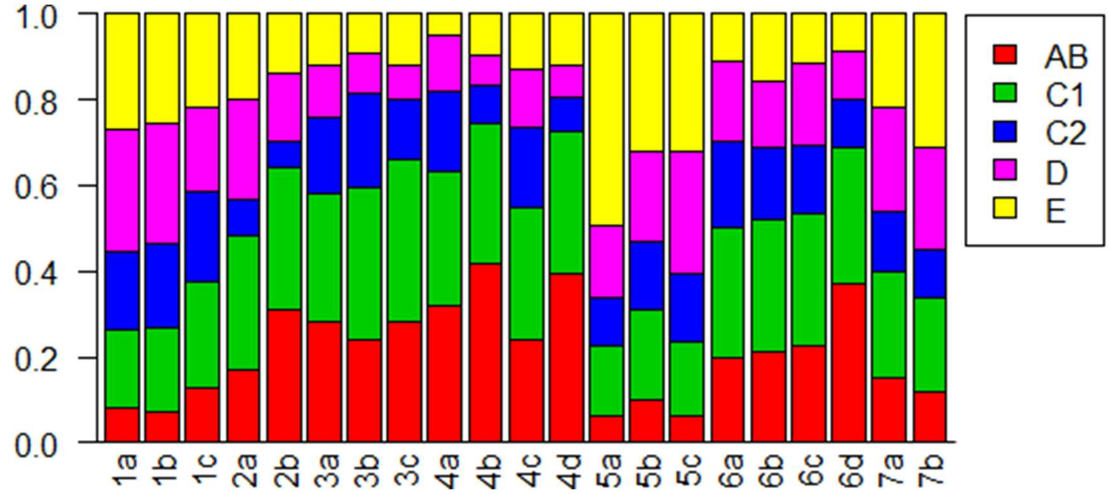


Figure 11.1. Average proportion (y axis) of Output Area residents in approximated social grades (x axis).

datasets. Figure 11.1 shows the average population composition of Output Areas in each area classification group in terms of approximated social grade. Output Areas belonging to groups in supergroup 1 (blue collar workers) unsurprisingly have relatively high proportions of the population in manual labour, and also dependent on state welfare. Also as expected, supergroup 5 (constrained by circumstances) includes the groups with the highest average proportions of the population dependent on state welfare, i.e. grade E.

The groups with relatively high proportions of non-manual workers (grades AB and C1) tend to be in supergroup 3 (countryside) or supergroup 4 (prospering suburbs). Although supergroup 3 is typified by agricultural employment (Table 11.3), the group profiles are not incompatible with this type of household, with high frequencies of working from home, multiple car ownership, and bought detached housing (Office for National Statistics 2005).

The high proportion of lower-level non-manual workers (grade C2) in supergroup 2 (city living) provides further evidence that the types of people living in these areas are young professionals. Similarly, the high proportions of grades D and E in supergroup 7 (multicultural) indicates that people in these Output Areas are generally not well-off.

11.3. Results

11.3.1. Approximated social grade

Significant associations between production of ecosystem services and the proportion of residents in particular social grades are generally not found except for social grades C2 and D, i.e. manual workers, which are correlated with all ecosystem services except opportunities for cultural ecosystem services. These results are shown in Table 11.4. There is one ecosystem service (habitat provision) that is significantly positively correlated with the proportion in social grade AB, but only at the more conservative significance threshold; and there are also two (habitat provision and reduction of air pollution) significantly correlated with social grade E, one of which remains significant at the more conservative threshold.

Interestingly, the proportion of Output Area residents in social grade C2 is positively correlated with ecosystem service production, while the proportion in grade D shows negative correlations (excluding opportunities for cultural ecosystem services; Table 11.4). The correlations for grade E are also negative, although most are not statistically significant.

The negative correlations between approximated social grade D (and to a lesser extent grade E) and all ecosystem services except opportunities for cultural ecosystem services (Table 11.4) are the result of Output Areas with high proportion of people in these grades tending to be located in the very centre of Sheffield, especially near the commercial/industrial areas (Map 39, Map 40, Map 2), where ecosystem service

Table 11.4. Spearman's *rho* rank correlation coefficients for association between levels of ecosystem service production and the proportion of the population belonging to approximated social grades. Darker greens and reds indicate stronger positive and negative correlations respectively. White background indicates results taken to be statistically non-significant at the Bonferroni-corrected threshold; grey background indicates results only significant at the non-corrected threshold.

	AB	C1	C2	D	E
Air pollution reduction	0.08	0.01	0.09	-0.14	-0.10
Heat island mitigation	0.00	-0.03	0.16	-0.12	-0.03
Runoff reduction	0.06	0.01	0.18	-0.16	-0.07
Carbon storage	0.05	-0.02	0.11	-0.12	-0.07
Cultural opportunities	-0.04	-0.06	0.06	0.04	0.02
Habitat provision	0.09	0.04	0.12	-0.15	-0.11

production levels are low (Map 6, Map 9, Map 12, Map 15, Map 18, Map 21). However, these are the same areas that have low proportions of residents in social grade AB, and yet the correlations for this grade are not as strong. This may be because Output Areas with high proportions of residents in social grade AB tend to occur on the outskirts of the urbanised area, where ecosystem service production is intermediate; with lower proportions in the rural area where ecosystem service production is high; and the lowest proportions in the city centre (Map 36). This would also explain why the correlations for social grade C1, which essentially shows a weaker version of the pattern seen for grade AB (Map 37), are so close to zero.

The positive correlations for social grade C2 are also surprising, as Output Areas with high and low proportions of residents in this grade seem to be fairly randomly distributed across the study area (Map 38). It is possible that this is because social grade C2 represents people who are both able to, and most consistently likely to, chose a residence according to features of the natural environment that correlate with higher levels of ecosystem service production.

As analyses in the previous chapters have found, spatial patterns are quite different for opportunities for cultural ecosystem services compared to the other ecosystem services (Table 11.4). This again results from the use of very different model inputs. The fact that none of the social grades are correlated with opportunities for ecosystem services suggests that there are no social inequalities in the production of this ecosystem service, at least as measured by variation in resident occupation. This is in contrast to the other services, which are in general more accessible to members of some social grades than others.

11.3.2. Area classification

The average ranks of ecosystem service production levels are shown for each area classification group in Figure 11.2, and the results of pairwise comparison tests in Table 11.5. One result that is immediate obvious from Figure 11.2, and which is confirmed in Table 11.5, is that Output Areas belonging to the Countryside groups (group 3) are in areas of far greater ecosystem service production than the other groups. This is not surprising, given that these Output Areas by definition are in areas of low population density with high levels of employment in agriculture, and thus are more likely than other groups to be situated in areas with more ecosystem service-producing greenspace.

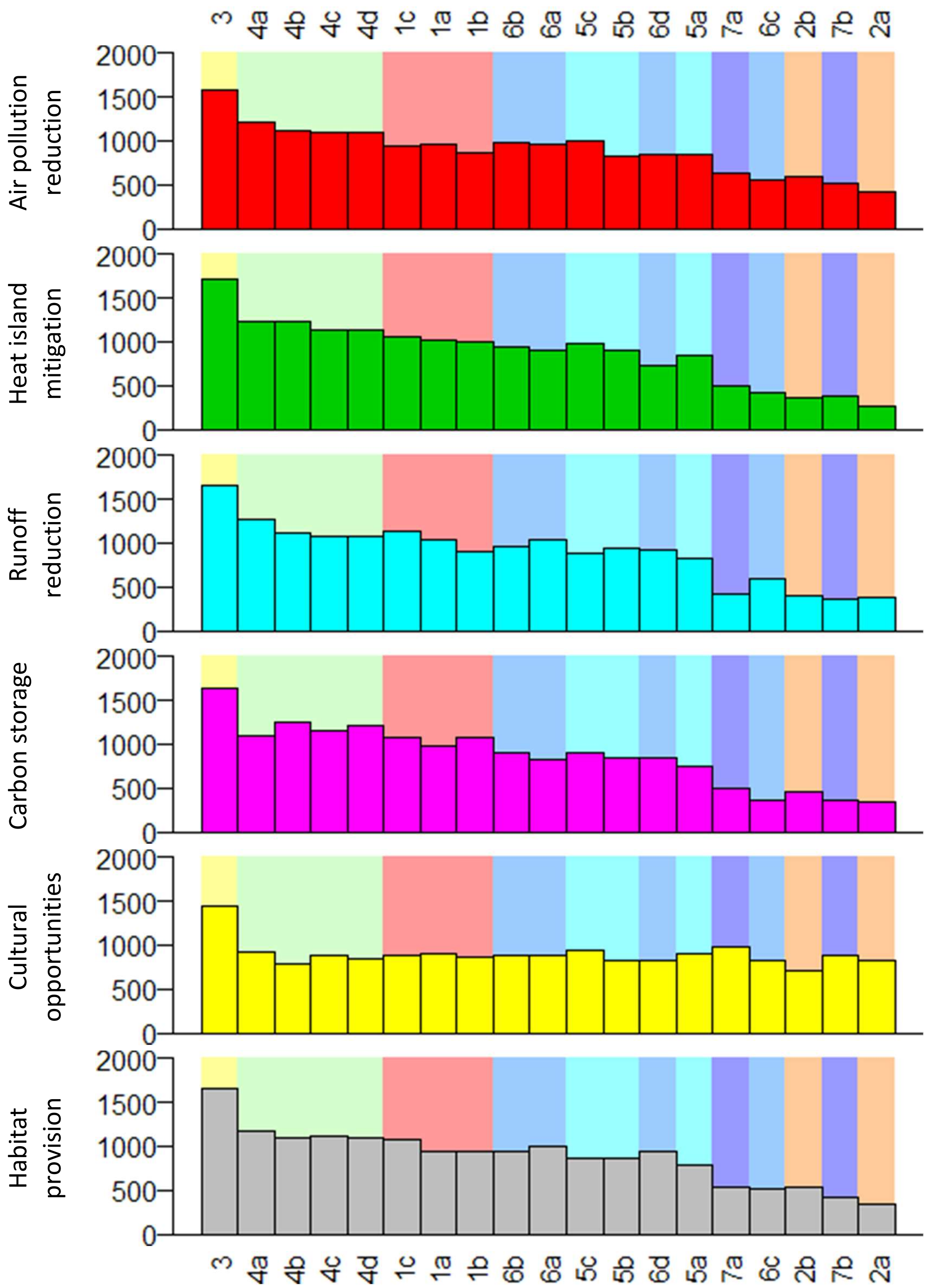


Figure 11.2. Average rank of ecosystem service production levels across different area classification groups. High ranks indicate high production. Groups ordered by decreasing rank averaged across ecosystem services. Background shading differentiates supergroups; colours as per Map 41.

Table 11.5. Mean ranks and pairwise multiple comparisons of ecosystem service production levels between different area classification groups. Numbers in second column show the average rank of ecosystem service production for the area classification group in the first column (where high ranks are high production). In the subsequent columns in the cells under the dashes, black cells indicate where the area classification group in that column has significantly greater ecosystem service levels than the group in that row at the Bonferroni-corrected significance threshold, while a grey fill indicates a difference that is significant according to the non-corrected threshold. White fills indicate no significant difference. (Multiple comparison test: Mann-Whitney U-test.)

a. Air pollution reduction

		3	4a	4b	4c	4d	5c	6b	6a	1a	1c	1b	6d	5a	5b	7a	2b	6c	7b	2a
3	*1584*	-																		
4a	*1119*		-																	
4b	*1201*			-																
4c	*1087*				-															
4d	*1086*					-														
5c	*942*						-													
6b	*870*							-												
6a	*994*								-											
1a	*956*									-										
1c	*972*										-									
1b	*962*											-								
6d	*843*												-							
5a	*824*													-						
5b	*845*														-					
7a	*637*															-				
2b	*595*																-			
6c	*509*																	-		
7b	*562*																		-	
2a	*422*																			-
		3	4a	4b	4c	4d	5c	6b	6a	1a	1c	1b	6d	5a	5b	7a	2b	6c	7b	2a

A second pattern is that while levels of five of the ecosystem services differ substantially between groups, levels of opportunities for cultural ecosystem services again show almost no significant variation except for the countryside group (Table 11.5e). This provides further evidence for the suggestion that access to this ecosystem service is fairly socially equal, at least for people living in non-rural situations.

For the other groups and ecosystem services, although there is some variation in the rank order of average ranks (Figure 11.2), there is also quite a lot of consistency in the groups that have significantly higher and lower levels of ecosystem service production (Table 11.5). The groups that consistently have the lowest levels of ecosystem service

Table 11.6 continued.

b. Heat island mitigation

		3	4b	4a	4d	4c	1c	1a	1b	5c	6b	6a	5b	5a	6d	7a	6c	7b	2b	2a
3	*1707*	-																		
4b	*1022*		-																	
4a	*996*			-																
4d	*1059*				-															
4c	*266*					-														
1c	*368*						-													
1a	*1220*							-												
1b	*1232*								-											
5c	*1133*									-										
6b	*1133*										-									
6a	*851*											-								
5b	*893*												-							
5a	*976*													-						
6d	*908*														-					
7a	*948*															-				
6c	*417*																-			
7b	*732*																	-		
2b	*500*																		-	
2a	*373*																			-
		3	4b	4a	4d	4c	1c	1a	1b	5c	6b	6a	5b	5a	6d	7a	6c	7b	2b	2a

c. Stormwater runoff reduction

		3	4a	1c	4b	4d	4c	6a	1a	6b	5b	6d	1b	5c	5a	6c	7a	2b	2a	7b
3	*1664*	-																		
4a	*1105*		-																	
1c	*1260*			-																
4b	*1067*				-															
4d	*1079*					-														
4c	*1125*						-													
6a	*909*							-												
1a	*878*								-											
6b	*1031*									-										
5b	*962*										-									
6d	*1034*											-								
1b	*818*												-							
5c	*936*													-						
5a	*917*														-					
6c	*414*															-				
7a	*405*																-			
2b	*363*																	-		
2a	*594*																		-	
7b	*369*																			-
		3	4a	1c	4b	4d	4c	6a	1a	6b	5b	6d	1b	5c	5a	6c	7a	2b	2a	7b

Table 11.6 continued.

d. Carbon storage

		3	4b	4d	4c	4a	1c	1b	1a	6b	5c	6d	5b	6a	5a	7a	2b	7b	6c	2a
3	*1642*	-																		
4b	*975*		-																	
4d	*1084*			-																
4c	*1084*				-															
4a	*345*					-														
1c	*463*						-													
1b	*1086*							-												
1a	*1253*								-											
6b	*1155*									-										
5c	*1203*										-									
6d	*751*											-								
5b	*847*												-							
6a	*894*													-						
5a	*831*														-					
7a	*910*															-				
2b	*366*																-			
7b	*848*																	-		
6c	*485*																		-	
2a	*369*																			-
		3	4b	4d	4c	4a	1c	1b	1a	6b	5c	6d	5b	6a	5a	7a	2b	7b	6c	2a

e. Opportunities for cultural ecosystem services

		3	7a	5c	4a	1a	5a	6b	4c	1c	7b	6a	1b	4d	5b	6c	2a	6d	4b	2b
3	*1436*	-																		
7a	*904*		-																	
5c	*858*			-																
4a	*880*				-															
1a	*825*					-														
5a	*716*						-													
6b	*920*							-												
4c	*792*								-											
1c	*887*									-										
7b	*833*										-									
6a	*902*											-								
1b	*832*												-							
4d	*946*													-						
5b	*875*														-					
6c	*891*															-				
2a	*831*																-			
6d	*814*																	-		
4b	*981*																		-	
2b	*879*																			-
		3	7a	5c	4a	1a	5a	6b	4c	1c	7b	6a	1b	4d	5b	6c	2a	6d	4b	2b

Table 11.6 continued.

f. Habitat provision

		3	4a	4c	4b	4d	1c	6a	6d	1a	1b	6b	5c	5b	5a	7a	2b	6c	7b	2a
3	*1651*	-																		
4a	*944*		-																	
4c	*943*			-																
4b	*1076*				-															
4d	*336*					-														
1c	*527*						-													
6a	*1172*							-												
6d	*1093*								-											
1a	*1108*									-										
1b	*1090*										-									
6b	*789*											-								
5c	*857*												-							
5b	*864*													-						
5a	*999*														-					
7a	*939*															-				
2b	*517*																-			
6c	*948*																	-		
7b	*535*																		-	
2a	*422*																			-
		3	4a	4c	4b	4d	1c	6a	6d	1a	1b	6b	5c	5b	5a	7a	2b	6c	7b	2a

production are: 2a (transient communities), 2b (settled in the city), 6c (young families in terraced homes), 7a (Asian communities) and 7b (Afro-Caribbean communities). Groups 2a and 2b include a high frequency of junior professionals living in flats, while groups 7a and 7b are the multicultural communities, who have high proportions of residents occupying flats and terraced housing (Office for National Statistics 2005). Output Areas belonging to all of these groups tend to have a higher than average population density, probably resulting in the opposite situation to that of supergroup 3, i.e. more built infrastructure and correspondingly low levels of ecosystem service-providing greenspace.

Groups in supergroup 5 (senior communities, older workers and public housing), and the other groups from supergroup 6 (settled households, typical traits and aspiring households) typically have intermediate levels of ecosystem service production. It is not surprising that Output Areas in supergroup 6, which is described as “typical” – most likely meaning that individuals in a wide variety of circumstances live there – have average levels of production. It is, however, interesting to note that Output Areas typified as being "constrained by circumstances" (supergroup 5) do not in fact have the lowest levels of ecosystem service production but, in fact, are fairly average.

Output Areas in groups belonging to supergroup 4 (prospering suburbs) and, to a slightly (but generally not significantly) lesser extent, supergroup 1 (blue collar communities), have higher levels of ecosystem service production than the other groups except supergroup 3 (Table 11.5). Supergroup 4 includes the four suburban household groups, with the profiles of fairly well-off families, e.g. detached housing, multiple car ownership (Office for National Statistics 2005). Yet the population density for the profile of this supergroup is very close to the UK mean (Office for National Statistics 2005), indicating that high population density cannot account for all variation in levels of ecosystem service production. Indeed, the profiles of the groups in supergroup 1, i.e. the blue collar communities, all have population densities above the UK mean (Office for National Statistics 2005).

11.4. Discussion

11.4.1. Evaluation of datasets

The datasets used to assess social equality in levels of ecosystem service production paint a picture of who lives where in Sheffield. The approximated social grade is based on occupation and is also correlated with income, while the area classification includes categories that describe profiles of stereotypically more and less well-off households in a variety of circumstances (Office for National Statistics 2005, National Readership Survey, no date). There are patterns of spatial clustering in both of these datasets, and thus, together, they are able to show spatial variation in levels of affluence, deprivation, and other aspects of social circumstances.

The preponderance of non-significant and quite weak correlations in Table 11.4 (contrast with the correlations found between ecosystem service production and urban morphology metrics in Section 10.3) suggests that approximated social grade is not as good a discriminator of access to ecosystem services as area classification group (Table 11.5), which finds significant differences for a variety of groups (although certain groups tend to be similar to each other). This is probably because the area classification is based on a wide selection of population variables and uses a clustering approach to define a large typology; whereas the social grade is based only on details of occupation.

The area classification is therefore able to a greater extent to discriminate levels of ecosystem service production in terms of demographics, household composition, housing, employment and socioeconomic details (Table).

11.4.2. Social inequality in ecosystem service production levels

One point on which results from both datasets agree is that opportunities for cultural ecosystem services are fairly equally distributed across social conditions in Sheffield: the production of this service is not correlated with the proportion of residents in any approximated social grade (Table 11.4), and only Output Areas classified as countryside (supergroup 3) have a different level of provision to those in other categories (Table 11.5). This is despite variation in the spatial distribution of social grades and area classifications (Map 36 to Map 41), and probably occurs because, although the types of publicly accessible greenspace change across the study area (e.g. public parks and playing fields/recreation groups being more common in the city centre, with woodlands more common on the urban outskirts), the total amount of these different types of greenspace does not change consistently with patterns of socioeconomic conditions.

Interestingly, this result contradicts the finding of a similar study from Sheffield, which found that neighbourhoods with more older and more deprived people tended to have greater access to parks (Barbosa et al. 2007). However, the two analyses are not directly comparable for a number of reasons, including use of different spatial boundaries and extent, and measurement of distance to parks via the road network (rather than distance as the crow flies, as used here).

The result from this study is however similar to that of a similar study from New York City, which found “[socioeconomically] unpatterned inequality” in access to parks (Maroko et al. 2009). Although there was wide variation in how well areas were provisioned with parks, there was no association with neighbourhood ethnic composition or income. In contrast, a study of Phoenix, Arizona, found that some ethnic groups were more likely to live in areas with good access to parks; and, significantly, the parts of Phoenix with large populations of youth had the lowest park access – especially to large, high quality parks (Cutts et al. 2009). The fact that the production of opportunities for cultural ecosystem services seems to be socioeconomically equitably distributed in Sheffield indicates that no particular group bears an above-average risk of lifelong social and physical health issues associated with inability to access public greenspace (Cutts et al. 2009). However, it has been noted that different social groups (e.g. people of different ethnicities) use parks in different ways, meaning that they have different requirements of public greenspace (Loukaitou-Sideris 1995). This aspect of socioeconomic equity could not be addressed here.

There is evidence for social inequality in the production of all five of the other ecosystem services, although the results are contradictory between the approximated social grade and the area classification. This seems to be largely due to the consideration of multiple variables in the area classification, meaning that the distribution of people of each social grade amongst Output Areas of different classification groups is wide (Figure 11.1) – although there are some trends (Section 11.2.4). The patterns of inequality are discussed in the following two sections, which look at the profiles of the ecosystem service-deprived and -privileged socioeconomic groups respectively.

11.4.3. Ecosystem service-deprived groups

The proportion of residents in social grade D (the second least affluent grade, although the lowest paid grade that is not welfare-dependent) is negatively correlated with ecosystem service production (Table 11.4), indicating that the residents of Output Areas with the high proportions of unskilled manual workers tend to be deprived in terms of ecosystem services. Yet the area classification supergroup that includes groups with the first and third highest proportions of residents in group D, i.e. blue-collar communities, has moderately high levels of the production of most ecosystem services. High proportions of residents of social grade D are also found in several other groups, however, including both multicultural groups (supergroup 7) and one of the city living groups (transient communities), both of which have generally low levels of ecosystem service production (Figure 11.2, Table 11.5). These associations probably explain the apparent contradiction in the results between the two datasets.

There are also quite high numbers of people of social grade D in Output Areas of supergroup 5, which is described as being "constrained by circumstances" and includes profiles of less well-off households (Table 11.3, Table 11.5). There are also higher proportions of people of social grade E in these groups than any other, confirming that the typical resident of these Output Areas is indeed not very well off (Table 11.5). Despite this, groups in supergroup 5 have intermediate levels of ecosystem service production (Figure 11.2, Table 11.5), indicating that the least well-off individuals in terms of socioeconomics are not the least well-off in terms of ecosystem services. The spatial distribution of Output Areas of supergroup 5 is quite wide across the study area (Map 41), and although there is a gradient of decreasing proportions of social grade E away from the city centre this pattern is fairly weak (Map 40). This is in contrast to patterns of ecosystem service production, which for most ecosystem services have quite a strong tendency to increase with distance from the urban core (Map 9, Map 12, Map 15, Map 21). This explains the proximate cause of the fact that ecosystem service production is not lowest in the least affluent areas of Sheffield, but not the ultimate cause.

One suggestion for the ultimate cause is that these portions of the population are restricted to certain types of housing (renting publicly owned flats is very common for supergroup 5) that are located in areas of reasonably high ecosystem service production – perhaps due to the inclusion of greenspace infrastructure in such areas – or very variable production. This is an idea that is developed further below and in the general discussion (Chapter 12).

In summary, the Output Areas that are most deprived in terms of ecosystem services are not, as was expected, those classified as being “constrained by circumstances”, or the least affluent. Rather, low levels of ecosystem service production are found in the following types of Output Areas:

- Output Areas whose population contains many unskilled manual workers;
- Multicultural communities with a high proportion of immigrants, where a high proportion of housing is rented flats and terraced, and where unemployment is high and car ownership/use is low;
- Output Areas where education levels are typically high and families with children are rare, and many residents occupy rented flats;
- Output Areas with many young families in rented terraced homes without central heating.

Although not many previous studies of the relationships between local socioeconomic conditions and ecosystem services exist, the fact that the least affluent group live in areas of intermediate ecosystem service production is not generally in agreement with the results that have been reported, but the general association between lower affluence and lower ecosystem service production is. Several studies from various North American and European cities have found associations between low socioeconomic status and exposure to outdoor pollution (although this study did not measure exposure to pollution, but rather pollution reduction by vegetation), including fine particulate matter (diameter < 2.5 μm) and nitrogen oxides (Rotko et al. 2000, Ponce et al. 2005, Schikowski et al. 2008, Marshall et al. 2009). This increased exposure is also linked to a higher burden of health problems, such as hospitalisations due to asthma and reduced lung function (Solorzano et al. 2005, Schikowski et al. 2008). The link between low socioeconomic status and health problems caused by air pollution has been attributed to factors such as poor housing conditions, less access to healthcare and lower levels of education (Ponce et al. 2005, Solorzano et al. 2005).

Low income has also been found to correlate with higher urban temperatures in Phoenix, Arizona, at least during the daytime (Harlan et al. 2006, Buyantuyev and Wu 2010). People living in poverty are, furthermore, unlikely to have the material resources to cope with heat and are therefore even more likely to suffer health problems caused by high temperatures (Harlan et al. 2006). Indeed, poorer people have a higher risk of death in extreme heat events (Johnson and Wilson 2009).

Several studies have also investigated the relationships between plant and avian species richness average neighbourhood income. In Leipzig, Germany, high levels of diversity of both of these taxa tend to be found in areas with high income and socioeconomic status and low unemployment rates (Kinzig et al. 2005, Strohbach et al. 2009). For plants, this relationship is stronger in gardens than in parks, perhaps because garden planting is an individual decision whereas park planting is more influenced by central policies (Kinzig et al. 2005). Plant diversity is also related to income in Phoenix, Arizona; in this case, the relationship may partly be driven by wealthy people’s preference for new housing, developments of which have been planted with a more diverse flora (Hope et al. 2003). Part of this relationship may be determined by differences in behaviours that encourage wildlife: a study has found that providing food for garden birds is more common in the less deprived areas of Sheffield (Fuller et al. 2008). Other authors have suggested that there is a “luxury effect”, whereby the wealthiest parts of a city tend to occur in areas with more remnant greenspace, and also have a more wildlife-friendly neighbourhood design (Strohbach et al. 2009). Regardless of the reasons, it is significant because of both the environmental justice implications and the consequences for larger scale biodiversity (Kinzig et al. 2005).

Interestingly, in the few studies that show the shape of the relationship between socioeconomic status and ecosystem services, the two that investigate US cities indicate that ecosystem services increase with socioeconomic status from low to medium status (Hope et al. 2003, Buyantuyev and Wu 2010). In contrast, the one study investigating a European city (Leipzig) show no consistent increase in the ecosystem service (biodiversity) from low to medium socioeconomic status (Kinzig et al. 2005). Germany, like the UK, has a higher expenditure on income support, housing benefits and other social benefits than the US (Organisation for Economic Co-operation and Development 2007, Adema and Ladaique 2009). It is possible that these benefits counteract some of the consequences of poverty and enable people to afford better housing than they otherwise would, meaning that ecosystem service production is more equitable amongst the less affluent sectors of society than might otherwise be the case. This is however a very small sample size from which to draw firm conclusions.

There is also evidence for associations between the ethnic composition of neighbourhoods and ecosystem services, as found in this study. For example, in the city of Hamilton, Ontario, high levels of air pollution are positively correlated with the proportion of the local population comprising Latin Americans, but negatively associated with the proportion comprising Asian Canadians (Buzzelli and Jerrett 2004). Buzzelli and Jerret suggest that this is caused by variation in affluence at the time of immigration to Canada: Asian immigrants from, for example, Hong Kong and South Korea tend to be relatively wealthy, while Latin Americans are comparatively poor.

These associations between socioeconomic status and ecosystem services are not, however, universal: some studies fail to find any link. For example, socioeconomic group is not associated with nitrogen dioxide exposure in Helsinki (Rotko et al. 2001). In Chicago, local average incomes are not related to total bird species richness, although they are related to the proportion of the total richness that comprises native versus exotic birds (Loss et al. 2009). A study of five UK cities found no significant relationships between the proportion of an area classed as being relatively affluent and highly professionally qualified, and stormwater runoff reduction or heat island mitigation (Tratalos et al. 2007); and in the same study, only one of eight metrics of biodiversity potential showed a significant positive relationship. Nevertheless, the bulk of the evidence indicates that the general pattern is for lower socioeconomic status to be associated with lower local levels of ecosystem services, even if it is not necessarily the case that the lowest status equates with lowest ecosystem service levels, as was found for five of the six services in this study.

11.4.4. Ecosystem service-affluent groups

Interestingly, the approximated social grade suggests that economically affluent areas, i.e. areas with high proportions of the population in grade AB, do not have higher levels of ecosystem service production than areas with lower proportions of the population in grade AB (Table 11.4): these variables are not significantly correlated, except weakly in the case of habitat provision. In fact, only the proportion of residents in social grade C2 (skilled and responsible manual workers) is positively correlated with ecosystem service production. The proportion in grade C1, who have a similar average income to grade C2 (Market Research Society, no date), but undertake non-manual work, is also not correlated with ecosystem services. This could mean that non-manual workers do not tend to make decisions about where to live according to landscape characteristics that influence ecosystem service production levels, whereas the opposite is true for more affluent manual workers. Indeed, high frequencies of people of these social grades are spread amongst area classification groups with profiles indicating a wide range of housing types (Figure 11.1), and therefore probably also a wide variety of landscape types.

There are high proportions of people of social grade C2 in several area classification categories, including supergroup 1 (blue collar communities), and groups 4a (prospering younger families), 4c (thriving suburbs) and 6a (settled households) (Figure 11.1). This is generally in agreement with the results from the area classification pairwise comparison tests, which consistently finds that supergroups 4 and 1 (prospering suburbs and blue collar communities respectively) have high levels of ecosystem service production compared to other groups except supergroup 3 (Table 11.5).

Supergroup 3, i.e. countryside communities, clearly have far higher levels of production of all ecosystem services (Figure 11.2). This is unsurprising, given that the profile of this supergroup indicates low population densities and high frequencies of detached housing (Office for National Statistics 2005). Residents of Output Areas in these groups also tend not to use public transport to travel to work, but are likely to come from households owning multiple cars; and are more likely than the national average to work in agriculture (Office for National Statistics 2005). All of these factors indicate relatively rural areas with a lot of greenspace infrastructure. Given the well known relationship between degree of urbanisation and ecosystem services (Walsh et al. 2005, Chadwick et al. 2006, McKinney 2006, Tratalos et al. 2007, McKinney 2008, Pickett et al. 2009), it is expected that such areas would have relatively high levels of service provision.

The profiles of other types of Output Area that can be considered ecosystem service-affluent are as follows. Residents of Output Areas in supergroup 1 (blue collar communities) are more likely than the national average to live in publicly rented terraced housing, but unlikely to live in flats; and are less likely to have higher education qualifications (Office for National Statistics 2005). There are three distinct groups in this supergroup (see Table 11.3 for details), but all have similar levels of ecosystem service production (Table 11.5).

Output Areas in supergroup 1 tend to have quite high proportions of residents in manual working social grades (C2 and D). Given the correlations between the proportions in these social grades and ecosystem service production, it can also be deduced that within the Output Areas in supergroup 1, those with higher proportions of residents in social grade C2 and lower proportions in grade D have the highest production of ecosystem services.

Groups in supergroup 4 (prospering suburbs) have similar levels of ecosystem service production to supergroup 1. The profile of supergroup 4 indicates relatively affluent areas: there are high frequencies of households in detached houses with multiple cars, with rented housing, terraced houses and flats being relatively rare (Office for National Statistics 2005). In addition, central heating is more common than the national average (Office for National Statistics 2005). Table 11.3 describes some of the key differences between the groups in this supergroup, although these differences do not coincide with significant differences in levels of ecosystem service production (Table 11.5). These groups have some of the highest proportions of non-manual workers, especially social grade AB, although the spread of non-manual workers amongst other area classification groups means that the proportion of residents in these groups does not appear to be correlated with ecosystem service production.

Supergroup 4 has a relatively high proportion of residents in the more affluent social grades (Figure 11.1), and the pattern of these Output Areas having high levels of ecosystem services fits in with the findings of most other similar studies (see discussion in the previous section). The fact that Output Areas in supergroup 1 also have high ecosystem service production is in contrast to these studies, however, as this supergroup has high proportions of less affluent residents. It is unclear why this pattern should arise in Sheffield, although it may perhaps be due to the urban morphology of these neighbourhoods.

11.4.5. The question of choice

In the previous section, the idea that non-manual workers do not make decisions about where to live according to ecosystem service providing landscape features was suggested to explain the lack of a correlation between the proportion of the population in social grades AB and C1, and ecosystem services. Similarly, it could be posited that skilled and more responsible manual workers (grade C2) are the only group who are able to, and actively chose to, live in areas with the greenspace infrastructure necessary to provide ecosystem services, hence the positive correlation between the frequency of such people in an Output Area and ecosystem service production (Table 11.4); whereas people of social grade D, who are less affluent, choose housing according to features that coincide with less ecosystem service providing infrastructure (possibly cost, for example), causing the negative relationship.

Implicit in this line of reasoning is that occupation and/or affluence influence choices about what kind of locale to live in, i.e. that causality for the correlations lies with social grade (or correlated attributes). However, this is not the only possible logic explaining the relationships. Section 11.4.3 suggested that people in social grade E are unable to make a choice about where to live but are forced into living in particular types of accommodation that are located in particular landscape types. More specifically, the infrastructure for communities of less affluent people, with particular types of housing and landscape settings (designed, probably amongst other things, to be lower cost) was laid down, and then less affluent people moved in. According to this reasoning, the causal factor lies with physical infrastructure that was designed and built by urban planners. It can also be imagined that many different types of physical infrastructure are specifically designed to be targeted to various particular sectors of the population. Indeed, the fact that housing types play such a prominent role in the profiles of area classification groups suggests that this could well be the case (Table 11.3).

In reality these explanations are probably not mutually exclusive, and there are likely to be feedback loops in which changes in infrastructure generate changes in the socioeconomic profile of an area, which in turn allow, or require, further infrastructure changes. It is difficult to find evidence to back up this reasoning, as there appears to have been little work investigating whether the quality of the environment influences choices about where to live in different ways for people of different socioeconomic classes. One source of evidence comes from an investigation of residential developments designed according to sustainability principles (Williams et al. 2010). The sustainability of developments was not found to be an important reason for choosing to live in such a development for most of the residents: features such as the size and type of home, parking space and development appearance were more frequently important (Williams et al. 2010). As building homes to sustainability standards creates a cost premium (Communities and Local Government 2008a), it is likely that less affluent people will be financially excluded from these developments. If this type of behaviour holds true in other situations, it may explain why there is no relationship between approximated social grade AB and ecosystem service production, and why the profiles of the most ecosystem service-affluent area classification groups do not indicate the highest financial affluence. However, no evidence could be found to indicate whether other socioeconomic groups make decisions according to different criteria.

The main point to remember here is that while the analyses in this chapter have been able to examine the relationships between the local social and economic situation and the production of ecosystem services, they cannot confirm why these relationships exist. It has been possible to identify which sectors of society are the most deprived and affluent in terms of ecosystem services, but not to say whether causality lies with socioeconomic factors, or with physical infrastructure factors related to ecosystem service production.

11.5. Summary

This chapter has used analyses of the ecosystem service model output, and datasets describing multiple aspects of local socioeconomic conditions, to examine relationships between levels of ecosystem service production and socioeconomic profiles at a small spatial scale. The results form a body of evidence for the existence of significant social inequalities in ecosystem service production levels. In particular, areas where high frequencies of unskilled manual workers or people of Asian and Afro-Caribbean ethnic identity reside, and areas occupied by young professionals without families and renting flats, or with families and renting terraced houses, tend to be most deprived of ecosystem services (Section 11.4.3). At the other end of the scale, the most ecosystem service affluent areas are typically countryside communities in agricultural regions (Section 11.4.4). Suburban communities, with many residents living in owned detached houses, and communities with many manual workers and many residents living in rented terraced houses, also tend to have quite high levels of ecosystem service production (Section 11.4.4). These findings satisfy the sixth research aim listed in Section 1.4.

12. General discussion

12.1. Aims of the thesis and principal findings

The spatially variable nature of ecosystem services and the fact that production of ecosystem services can be profoundly influenced by environmental management actions mean that the ecosystem service concept has potential to be used to inform planning and decision making. The central theme of this thesis has been an exploration of this potential in the urban context (Section 1.4). The core aim of the thesis was therefore to model urban ecosystem services in Sheffield in such a way that the output has the potential to be used by urban planners decision makers; and then to explore the output to demonstrate some of this potential.

Several requirements of the models were identified in order to make the output useful (Section 1.3.3): output must be spatially explicit, because ecosystem services and urban planning are inherently spatial issues; output must be produced at a local scale relevant to urban decision making; and the selection of models should achieve an appropriate balance between inclusion of as many ecosystem services as possible and the accuracy/depth of individual models. The "production function" approach, which combines multiple spatial data inputs, was identified as an appropriate way to achieve these criteria. A final requirement was that the models should use widely available data inputs so that the developed models could be implemented across a wide geographic range.

Six specific aims for this thesis were identified (Section 1.4), which when achieved also satisfy the core aim. The ways in which these aims were met, and key findings from analyses, are detailed in the following sections.

12.1.1. Aim 1: identification of a suite of ecosystem services suitable for modelling at the neighbourhood scale within a city using existing data sets and methods

Six ecosystem services were identified, and were first presented in Section 2.1. The ecosystem services are as follows: reduction of air pollution levels by deposition of nitrogen dioxide (NO_2) and particulate matter (PM_{10}) to vegetation surfaces; mitigation of the heat island effect through evapotranspirative cooling by vegetation; reduction of stormwater runoff by retention in vegetation and abstraction into non-sealed surfaces; carbon storage in vegetation and less disturbed soils; opportunities for cultural ecosystem services in publicly accessible greenspace; and the provision of habitat for biodiversity. The first five of these ecosystem services provide known direct benefits to human health and well-being, which are reviewed in Chapter 3 to Chapter 7 and summarised in Table 9.1. In the case of habitat provision, biodiversity is thought to be critically important to many ecosystem services (Naeem 1998, Folke et al. 2004, Millennium Ecosystem Assessment 2005a); providing space for biodiversity to exist thus also contributes to human well-being, albeit indirectly (Chapter 8, Section 1.2.6).

As well as providing a diversity of benefits, these ecosystem services differ in several other important features. The ecosystem services are affected by different aspects of the natural and built environments, and this is reflected in the data inputs used for each model (although some data inputs, especially land cover, are also used across multiple models). The ecosystem services also include examples categorically different types, with cultural ecosystem services providing intangible, personal benefits; habitat provision being an example of a “supporting”, or indirectly beneficial, service; and the remaining four regulating the physical and chemical environment (De Groot et al. 2002, Chee 2004, Millennium Ecosystem Assessment 2005b). Different factors influence levels of demand for the ecosystem services, and the spatiotemporal characteristics of supply also vary (Table 2.1). Thus these six ecosystem services, identified as being suitable and feasible for inclusion in this thesis, represent a diverse range of services.

12.1.2. Aim 2: collation of input data requirement by the models from existing data sources

Before the models could be implemented, it was necessary to collate the data required for the models to function. Model-specific parameters were collated from a range of sources, but chiefly the documentation of the adapted models. These parameters are listed in Appendix A, which gives technical details of each of the models, and the models themselves are described in Chapter 3 to Chapter 8.

In order to make the models spatially explicit, several spatial datasets were also required; namely land cover, land use, soils and air pollution maps (Section 2.2). GIS layers of land use, soils and air pollution were acquired for the study area from published sources. However, a suitable GIS layer of land cover data could not be found, so one was constructed using two pre-existing datasets: the Land Cover Map 2000 and the Ordnance Survey MasterMap topography layer (Section 2.2.1). This key advantage of this new land cover dataset is that it maps individual landscape features at a fine scale, with each feature being classified as having one of thirteen land cover types or as having unknown land cover or natural land cover of an unknown type. Although the spatial distributions of land use, land cover and soils are not wholly independent, for example with some land uses being restricted to particular soil types and requiring certain land covers, these datasets each provide information used in the production functions of the various ecosystem service models.

12.1.3. Aim 3: implementation of the ecosystem service models

Chapter 3 to Chapter 8 each present one ecosystem service model, with technical details provided in Appendix A. The models produced spatially explicit datasets about the production of the six ecosystem services across Sheffield, i.e. where the entity or process that ultimately provides a benefit to human welfare is produced by the biophysical infrastructure of the environment. The output was produced at three different spatial resolutions (Section 2.3): arbitrarily drawn 500m grid squares, which mirror the way study plots are usually designed in ecology and shows area-standardised levels of ecosystem services; Historic Environment Character areas, which shows areas with a relatively homogeneous land use history; and Output Areas, which reflect socioeconomic conditions and are drawn to maximise social homogeneity. These different spatial resolutions are suited to investigating different types of questions.

The model output is shown in Map 5 to Map 22 in Appendix D. It is obvious from the visual output that the spatial distribution of ecosystem services differs between ecosystem services: reduction of air pollution and habitat provision are greatest in the agricultural belt between the moorlands and the urban core of Sheffield; heat island mitigation and carbon storage generally increase with distance from the urban centre; storm water runoff reduction is greatest in the heath (but not bog) moorlands; and opportunities for cultural ecosystem services are low in most areas except the moorlands. The fact that the distributions are different shows that the models are reflecting different functions of the land. The three spatial resolutions also reveal different patterns, for example with the Output Areas showing greater detail in the areas where more people are resident, which further indicates the suitability of the resolutions for addressing different questions.

Modelling the supply of benefits from the ecosystem services, i.e. the spatial distribution of welfare benefits to humans (which may be different to the distribution of production due to the flow of entities/processes, for example down-catchment or by diffusion) proved more difficult (Section 9.5.5). Supply modelling was possible for heat island mitigation, for which the distribution is the same as for production, and opportunities for cultural ecosystem services, for which supply distribution was considered as a simple function of distance from production. The supply of benefits from carbon storage, i.e. reduction of climate change risk, is non-spatial as all humans benefit; however, quantifying reduction of these risks was not possible. The supply of reduction of air pollution levels and lowered flood risk from storm runoff reduction could not be modelled because it was impossible to couple the ecosystem service production models used here to further models required to predict the supply. (Habitat provision, which is considered here as an indirect "supporting" ecosystem service, does not have a supply distribution in the same sense as the other, direct ecosystem services.)

The fact that ecosystem service supply could not be modelled has implications for the ability of this suite of models to address certain types of question. It is important to bear in mind that analytical results from maps of ecosystem service production will not necessarily show where the benefits of ecosystem services are being received. Nevertheless, it seems likely that, in the case of these six services, at least some of the total benefit will be provided at the site of production. It was also not possible to model the actual benefits to humans supplied by any of these ecosystem services, due to a paucity of quantitative evidence linking ecosystem services to health or well-being.

Modelling this flow is something that appears rarely to have been tackled previously. A global scale study found only four ecosystem services that could be mapped using this production to supply 'flow' approach (Naidoo et al. 2008). Of these, carbon sequestration and carbon storage have global flow (i.e. equal supply of benefits everywhere) and grassland production of livestock has negligible flow (i.e. supply is at site of production). The remaining ecosystem service, water provision, was mapped from the point of production to the point of supply using a process-based hydrological model. No other studies applying this type of approach could be identified from the literature.

The difficulty encountered when trying to link ecosystem service production to supply has implications for how the model output can be used to inform urban planning and management. The output datasets produced by the models in this thesis proved to be suitable for investigation in and of themselves (Chapter 9), and in relation to urban morphology (Chapter 10) and socioeconomics (Chapter 11), producing interesting results about the spatial patterning and coincidence of ecosystem services with the natural, built and social environments. However, where ecosystem service supply and/or health benefits cannot be quantified it is only possible to draw conclusions about what is good for ecosystem services in explicit areas, rather than what is good for the people living there. In other words, although higher levels of ecosystem service production at a spatially explicit site can be taken to be better for people in general, it cannot be said that the people who benefit maximally are those at that site; the exception is for ecosystem services for which benefits are obtained at the site of production, such as with the heat island mitigation model used here.

12.1.4. Aim 4: investigation of relationships between the modelled ecosystem services in the study area in order to contribute to understanding of how to undertake ecosystem service assessments

Understanding the spatial similarities and differences between multiple ecosystem services is essential to management and decision making, for example in order to avoid damaging the production of one ecosystem services when intervening to improve another (Anderson et al. 2009). Chapter 9 presents two analyses undertaken to develop such an understanding, namely correlation analysis and hotspot analysis. These analyses quantify covariation between ecosystem services and identify priority areas for conservation. There is also a more heuristic discussion of the general patterns observed.

The investigations make several pertinent findings. There is a general increase in the production of ecosystem services from the urban core of Sheffield outwards, which is the pattern expected along urbanisation gradients (Oke 1987, Gill 2006, Tratalos et al. 2007), but there are also exceptions that show that local, small-scale factors play an important role in determining ecosystem service production. The small- and larger-scale patterns also differ between ecosystem services, as shown by visual inspection of the ecosystem service maps and the correlations coefficients between ecosystem service pairs (Section 9.4.2).

The hotspot analysis found that there are few areas within the urbanised region that produce high levels of multiple ecosystem services, despite high levels of individual ecosystem services frequently occurring in this region. The urban hotspots that are present can be identified mainly as public parks (Map 27, Map 28), highlighting the importance of parks for providing ecosystem services to urban citizens.

Perhaps the most important finding from Chapter 9, which is relevant not only to Sheffield but also has implications for other ecosystem service studies, is that the choice of spatial units of analysis has a profound effect on analytical results (Section 9.5.4). The effect of spatial units on results does not appear to have been studied previously in relation to ecosystem services. However, this study has found that the different spatial units produce consistently different results, presumably because the units identify different types of detail "on the ground"; for example, Output Areas and Historic Environment Character areas identify neighbourhoods of socioeconomic and land use homogeneity respectively. The implication of this finding is that the spatial units used in any given study should be chosen carefully and with reference to the question(s) being addressed. This is especially important given ecologists' habit of arbitrarily assigning study plots, which may have little relevance to the social and cultural questions frequently addressed in ecosystem service science.

12.1.5. Aim 5: analysis of relationships between urban morphology and ecosystem services in order to make urban planning recommendations

There are theoretical expectations that urban morphology, which is defined as the spatial structure and composition of human settlements, is a strong influence on ecosystem service production (Alberti 2005). At present, however, there is very little empirical evidence with which to evaluate these expectations (Section 10.1, Table 10.2). Urban morphology includes features such as housing density and type, land use patterns and transport infrastructure. Chapter 10 uses a number of metrics representing these features to investigate how urban morphology, land use and ecosystem service production covary spatially within Sheffield, and in doing so demonstrates that the models implemented in this thesis can be used to inform urban planning.

The analysis was used to produce recommendations for the improvement of ecosystem services via urban morphology. There were found to be no situations where changing one aspect of urban morphology would improve some ecosystem services while degrading others, i.e. the direction of correlations was the same for all ecosystem services. The absolute and relative correlation strengths varied between services, however, meaning that the security of achieving improvements from changing a particular urban morphology characteristic varies between services (Section 10.4.8). Thus the provision of all ecosystem services could be improved (in decreasing order of priority) by decreasing the proportion of impervious cover, reducing the population density, and building more detached houses (as opposed to terraces/flats). Constructing smaller buildings at lower density and building a less extensive road network also help, but to a lesser degree (Table 10.8). Urban morphology characteristics appear to have far more influence than land use type on levels of production when all land uses are included in the analysis.

Recommendations for the improvement of individual ecosystem services vary in terms of which urban morphology characteristics should be considered as the highest priority (Table 10.9). The highest priority for reducing air pollution and mitigating the heat island effect is decreasing population density, while runoff reduction, carbon storage and habitat provision are most likely to be improved by reducing the proportion of impervious cover. In contrast, land use is the most important factor influencing opportunities for cultural ecosystem services, reflecting the model construction: levels of this ecosystem service can be improved by converting land use to water bodies, unenclosed land, woodland, and ornamental, parkland and recreational uses.

Chapter 10 also analysed ecosystem service – urban morphology patterns for two individual land uses: residential and industrial. The recommendations for these land uses were each different to each other and to the recommendations across all land uses (Table 10.9). These differences probably reflecting constraints imposed on urban morphology by land use, indicating that although land use is not the most important factor in improving ecosystem service production, it is important to consider the practicality of particular morphologies to each land use when making recommendations.

12.1.6. Aim 6: analysis of social inequity in access to ecosystem services at the neighbourhood scale, in order to identify any sectors of society with limited provision

It is known that, at a global scale, relatively poorer urban communities are more likely to face environmental health hazards (Millennium Ecosystem Assessment 2005c). There are a few within-city studies that indicate the same is, at least in general, true at the neighbourhood scale (Table 11.1). The possible exception to this is access to parks/public greenspace, which in some cities appears to be higher in less affluent areas. Achieving socioeconomic equity in ecosystem services is important because ecosystem services are essential for health and welfare, levels of which tend to be lower amongst less affluent sectors of society.

Chapter 11 uses datasets quantifying different aspects of local-scale socioeconomic conditions to explore patterns in ecosystem service production (approximated social grade and area classification of Output Areas; Section 11.2). The datasets both identified strata of society that are affluent and deprived in terms of ecosystem services. The exception was opportunities for cultural ecosystem services: these services were relatively equitably distributed amonst socioeconomic groups (Section 11.3).

The approximated social grade uses census data to assign households to one of five groups reflecting the occupation of the main household earner; the grade is also correlated with affluence and media and consumer behaviour (Section 11.2.2). The proportion of the total population of an area comprising each approximated social grade was correlated against ecosystem service production. Only two grades were significantly correlated with ecosystem services (Table 11.4): the proportion of the population made up of the group representing skilled manual workers was positively correlated with ecosystem services, while the opposite was true for unskilled manual workers. There were no strong patterns for non-manual workers or people dependent on welfare.

The area classification of Output Areas used data from the 2001 Census to assign each Output Area to one of 21 categories (hierarchically organised into seven groups) using multivariate methods (Section 11.2.3). Output Areas classified as ‘countryside’, which are typically fund in rural areas and have low population density, have the highest levels of ecosystem services; followed by ‘properous suburbs’ (households typically owning detached housing) and ‘blue collar communities’ (typically living in publicly rented terraced houses, and with relatively low prevalence of higher education). ‘City living’ and ‘multicultural’ Output Areas have the lowest levels of ecosystem services; these Output Areas, which are common in the central urban area, have high proportions of young professionals renting flats and people from ethnic minorities respectively.

It can be concluded from these analyses that unskilled manual workers, multicultural communities, and young households are deprived of ecosystem services, especially when compared to rural residents and more affluent and established households (Section 11.4.3, Section 11.4.4). Interestingly, however, it is not the least affluent who live in areas with the lowest levels of ecosystem service production. It may be that this is due to interplay between the ability to afford accommodation in particular areas/housing types, and covariation between socioeconomic status and decisions about where to live (Section 11.4.5). This idea is developed further in Section 12.2 below.

12.2. Achieving social equality via urban morphology

The principal findings of Chapter 10 and Chapter 11 were that levels of ecosystem service production vary with patterns of urban morphology and with socioeconomic conditions respectively. It might be expected that urban morphology and socioeconomic conditions also vary with each other. The analyses in Chapter 10 and Chapter 11 were performed at different spatial resolutions in an attempt to describe homogeneous units first of urban morphology (Historic Environment Character areas) and then of socioeconomic conditions (Output Areas) and thus cannot be directly compared. Figure 12.1 shows, however, that there is considerable variation in urban morphology between Output Areas of different area classifications (Section 11.2.3).

Even excluding the countryside Output Areas (group 3), which have drastically different patterns of urban morphology, there are clear differences between Output Areas of different classes. Figure 12.1 shows the area classes in order of decreasing average ecosystem service production. Most of the metrics, especially the proportion of impervious cover, proportion of detached houses, and the road LDI, show general patterns with decreasing levels of ecosystem service production (Figure 11.2); although the metrics do not covary exactly. It is thus apparent that Output Areas of different classes not only have varying levels of ecosystem service production, but also unique patterns of urban morphology.

In Section 11.4.5 I proposed a possible cause of the relationship between socioeconomic conditions and ecosystem service production; namely, that all except the most affluent people are limited in their choice of where to live by housing cost. There are many determinants of housing cost, a review of which is beyond the scope of this discussion. Suffice it to say components of urban morphology are reflected in the cost of a residence (Jones et al. 2009, Jones et al. 2010b). In order to provide housing opportunities for less affluent people, particular types of housing must be available: for

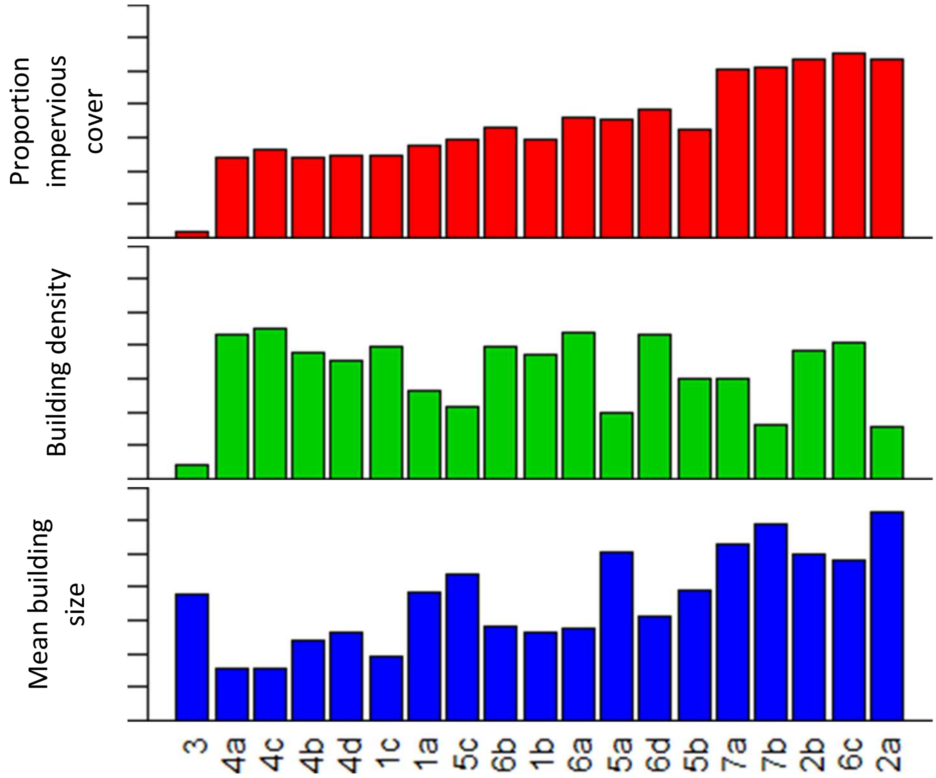


Figure 12.1. Average rank of urban morphology metric values across Output Areas with different area classifications. Area classifications ordered by decreasing rank averaged across all ecosystem services. (See Chapter 11 for further information about area classifications, especially Table 11.3 for descriptions of groups and Figure 11.2 for ecosystem service production.)

example, blocks of flats and terraced houses with courtyards or small gardens, rather than detached houses with large gardens. More expensive, lower density housing is also likely to be found in particular types of areas, e.g. away from industrial estates or very busy roads, and close to parks or the countryside – in other words, away from the city centre (Jones et al. 2009, Jones et al. 2010b).

The least affluent people will likely have the least choice about where to live, and it seems probably that more affluent people will choose more "desirable" (expensive) areas; and the respective neighbourhoods will tend to have different but to some degree consistent types of morphology. These choices (or otherwise) are reflected in the fact that housing type is a distinguishing feature for many of the Output Area classes (Table 11.3). The Output Area classes are also variously distinguished by characteristics such as living arrangements, family situation, age composition, field of employment and ethnicity (Vickers and Rees 2007), some of which would be expected to reflect affluence. This suggests that affluence may indeed be an important determinant of where to live, although not the only one (Jones et al. 2010b). A previous study from the UK has also found that the socioeconomic structure of a city is tightly linked to its

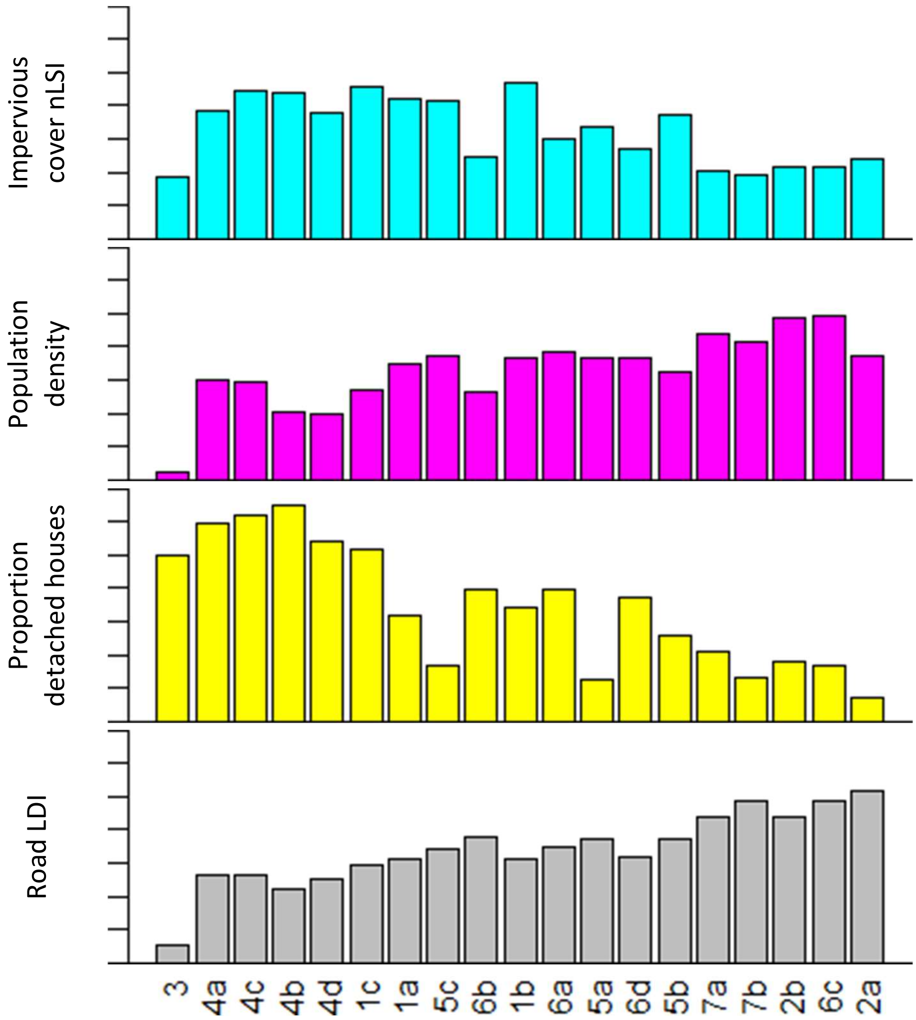


Figure 12.1 continued.

physical structure via the housing market, noting that income is key to a household's choice of location within a city (Jones et al. 2009, Jones et al. 2010b).

In addition to the work in this thesis, previous studies from various cities in developed countries have found evidence that ecosystem service provision is not socioeconomically equitable (Table 11.1), with income commonly being a significant influence, but also other factors such as neighbourhood ethnic and age composition (see references in Table 11.1). If housing cost is a barrier to achieving social equity of ecosystem service production, and ecosystem services also vary strongly with patterns of urban morphology, then the challenge is to change the way in which low cost housing is designed in a way that is conducive to improved levels of ecosystem service production.

Table 10.9 lists urban morphology recommendations for improving ecosystem service production in residential areas. The recommendations vary for different ecosystem services but include reducing population density, reducing the proportion of impervious cover, and increasing the proportion detached houses. These recommendations are suggestive of suburban and rural neighbourhoods, being typical of Output Areas in supergroups 3 and 4 (Figure 12.1). The socioeconomic profiles and approximated social grade composition of these groups, however, indicates quite well-off residents and typically expensive types of housing (Table 11.3). A further problem with this type of low density housing is that the population becomes spread out over a greater area, leaving less uninhabited countryside in which to conserve sensitive biodiversity and the production of ecosystem services that are produced within cities poorly or not at all (Burton 2000, McKinney 2006). The tools developed in this thesis would be suitable for examining this kind of trade-off, if manipulated spatial data were used as input to the models.

It is thus apparent that, at least in the present circumstances, optimal ecosystem service production and lowest house price are mutually exclusive goals at the neighbourhood scale. Yet neither are these goals entirely incompatible: according to the differences in ecosystem service production between Output Area classes, it is not the least well-off classes (those in supergroup 5) who live in areas with the lowest levels of ecosystem service production (Section 11.4.3). The types of housing occupied by these groups may therefore be a good model to at least reduce social inequalities.

Figure 12.1 describes the typical urban morphology characteristics of Output Areas in supergroup 5. These groups tend to have moderate proportions of impervious cover and road LDI; low building density and proportion detached houses; and quite large building size and high impervious surface nLSI and population density. The census profiles of these groups indicate that flats are more common than average, and detached housing is rare.

Figure 12.2 illustrates what this kind of urban morphology looks like "on the ground" by means of aerial photography of two supergroup 5 Output Areas. These Output Areas were chosen to be representative of their group and have levels of ecosystem service production typical of this supergroup. The first photograph shows an area of mixed terraced and semi-detached housing with gardens and blocks of flats with communal greenspace. The second photograph also includes terraced and semi-detached housing with some publicly accessible greenspace.

While housing of this type may promote ecosystem service production, it may not however be socially acceptable, i.e. it may not be the type of housing that people want

Figure 12.2. Aerial photography of example Output Areas (darker areas are outside Output Area boundaries), classified as supergroup 5, which have average levels of ecosystem service production. Photography source: Google Maps (accessed 27/07/2010).

to live in. Very little research has looked at the social aspects of urban form at a local/neighbourhood scale. Bramley et al. (2010) investigated relationships between urban morphology and 'social sustainability', or "the continuing ability of a city to function as a long-term, viable setting for human interaction, communication and cultural development" (Yiftachel & Hedgcock 1993, cited in Bramley et al. 2010) across five UK cities, including Sheffield. They found that a sense of pride and attachment to, and satisfaction with, one's neighbourhood and home is best promoted by low density suburbs (Bramley et al. 2010). This finding reflects far older evidence from the United States, where consumer surveys have repeatedly found a preference for suburban residence (Gordon and Richardson 1997).

Interestingly, however, there is also evidence that lower density housing types, at least at the city scale, are themselves associated with higher levels of socioeconomic segregation (Burton 2000). This is argued to be a result of the inability of lower income groups to be able to afford low density housing types, while all income groups are found in high density housing types (Burton 2000). Consequently, in cities with a low proportion of low density housing, socioeconomic segregation remains low (Burton 2000). Thus it appears that the social preference for suburban living is itself the cause of socioeconomic segregation and the concurrent social inequity in ecosystem service production.

The wider problem of social equity more generally is beyond the scope of this discussion. The key message of this section is that access to high levels of ecosystem service production is one amongst many things that suffers as a consequence of socioeconomic deprivation. The analyses performed in this thesis have identified some potential starting points for improving the situation for the most ecosystem service deprived groups of society by changing the way in which low cost housing neighbourhoods are designed. A major challenge in implementing such designs, however, is that present cultural preferences do not highly rate the types of neighbourhood that have been suggested here as models. It is possible that educating people about ecosystem service benefits may aid changing these preferences.

12.3. Methodological evaluation

12.3.1. Statistics versus heuristics

The implementation and interpretation of the analyses in this thesis have been impeded by several features of the data that present barriers to the use of conventional statistical procedures. These features are spatial autocorrelation, irregular frequency distributions and the inclusion of multiple variables both as predictors (socioeconomic conditions/urban morphology) and responses (ecosystem services).

Statistically significant spatial autocorrelation was found to be present in levels of production of all ecosystem services, and was ascribed to a tendency for similar land uses, and thus also types of land covers, to be spatially hierarchically clustered (Section 9.4.1, Section 9.5.2). The usual approach to dealing with spatial autocorrelation in statistical analysis is to use algorithms that calculate the reduced number of degrees of freedom, and then use this number to find the correct probability value (Rangel et al. 2006). However, the implementation of this approach was not possible because of the non-normal frequency distributions found in the data: as discussed more full in Section 9.2.1, existing algorithms do not work with rank data (Clifford et al. 1989, Dutilleul et al. 1993).

A further consequence of needing to use rank data is that multiple regression could not be used. Thus the analyses in this thesis usually comprise tables of large numbers of single variable correlations between all possible pairs of variables (e.g. Table 9.2, Table 10.5, Table 11.4). This means that the chances that Type I errors will occur become unacceptably high. This is also an issue in the case of the large tables of pairwise multiple comparison tests (e.g. Table , Table 11.5). The usual approach to identifying a more appropriately conservative significance threshold is to use Bonferroni corrections (see Section 9.2.2); yet this approach also has philosophical and practical problems (Perneger 1998, Cabin and Mitchell 2000, Moran 2003).

Thus the inherent characteristics of the datasets used in this thesis causes problems in the interpretation of the significance of results for two reasons: firstly, because spatial autocorrelation means that each data point is not full independent; and secondly, because the number of tests performed means that the probability of false conclusions is high. It has been difficult to find guidance in how to proceed in such a situation. The eventual approach taken, as described fully in Section 9.2, was:

1. To present non-spatial statistics but to use a more conservative significant threshold of 0.001 (compared to the usual 0.05) to try to prevent Type I errors due to spatial autocorrelation. This approach follows, but is more conservative than, Dale & Zbigniewicz (1991, cited in Fortin and Dale 2005), who used a threshold of 0.01.
2. To present both Bonferroni-corrected and uncorrected results to indicate how multiple tests might affect interpretation, as suggested by Cabin & Mitchell (2000).

3. Rather than relying on the significance of single results, to look for patterns of significance across tables, following the advice of Moran (2003).

It might be argued that this approach was theoretically overly conservative. Nevertheless, the analyses presented found many patterns that were considered statistically highly significant even following this procedure, suggesting that the associations are very strong and thus important in reality. Weaker relationships were also interpreted as non-significant (e.g. Table , Table 11.5e), indicating that the level of statistical conservativism adopted was appropriate in the present situation.

Cabin & Mitchell (2000) pointed out with regards to Bonferroni corrections that there is a need to use common sense in the interpretation of statistical tests. The success of the idiosyncratic approach used here suggests that this is true more widely of all situations in which standard statistical implementation and interpretation procedures do not apply. In essence, it has been critical not to rely on the results of individual statistical tests as the ultimate determinant of whether or not a pattern is significant, but to use them as heuristic tools that, in synthesis, aid the understanding of complex real-world situations.

A further reason why the statistics presented in this thesis should be used only as heuristic tools lies with the more fundamental problem that the uncertainty associated with the ecosystem service model output is unknown. The input data for land cover and land use is recent and is as accurate as any such dataset is likely to get. This is because the MasterMap topography layer, the Land Cover Map 2000 and the Historic Environment Character datasets were constructed using a combination of aerial photography and ground truthing in order to establish spatial boundaries and accurate classification (Fuller et al. 2002, South Yorkshire Archaeology Service 2005, Ordnance Survey 2008). However, the soils data are probably less reliable due to the spatial variability of soil properties and the fact that these datasets are necessarily constructed from point, rather than continuous, sampling.

A further issue for the air pollution reduction, heat island mitigation, stormwater runoff reduction and carbon storage models is uncertainty associated with input parameters. Many of the parameters used in the air pollution reduction model have known (but unquantified) uncertainty or variability (Zhang et al. 2001, Zhang et al. 2002, Zhang et al. 2003). Less information was available about the uncertainty associated with input parameters for the other models, but the use of single values in generating the model output means that there will inevitably be some inaccuracy. Similarly, the index used to quantify habitat provision has not been empirically grounded, but it was based on a combination of ecological characteristics that, at least singly, are known in general to affect biodiversity (see Section 8.2, Appendix A.5).

For these reasons the model output is unreliable as a statement that, for example "the surface temperature reduction in place x is y degrees". As discussed in Section 9.5.1, the model output should only be used to make statements about relative levels of ecosystem service production. Furthermore, the urban morphology metrics and the datasets used to indicate socioeconomic conditions are, at best, indicators of certain aspects of complex concepts. The conservative approach to statistical significance should provide some security against this uncertainty, but it is nevertheless important to remember these facts in the interpretation of what the results mean, as has been done in the discussions in this thesis; and, again, to use the statistical results not as black-and-white figures but as heuristic tools that contribute to understanding the real world.

12.3.2. Production function approach

Another point to consider is that some of the ecosystem service models may be more accurate than others. Air pollution reduction, heat island mitigation and stormwater runoff reduction are all quantified from process-based models of varying complexity (air pollution reduction being the most complex and runoff reduction being the least). Although, as already mentioned, the use of empirical parameters with unknown degrees of uncertainty/variability introduces some inaccuracy to the output from these models, the fact that the models are based on relatively well understood science lends credibility.

The inventory approach used in the carbon storage model is simpler and does not explicitly consider any processes such as photosynthesis, soil respiration, and soil disturbance. These processes are heavily influenced by factors such as local meteorological conditions and land use (Pouyat et al. 2006, Jandl et al. 2007, Byrne et al. 2008). The fact that these processes are not considered in the model (in contrast to meteorological data for the air pollution reduction and heat island mitigation models, for example) will introduce inaccuracy to the model from an alternative source. Similarly, the model of opportunities for cultural ecosystem services is very simple, consisting of a binary decision over whether a particular land use type is suitable for providing these opportunities. All other factors influencing cultural ecosystem services, such as recreation facilities and security, are ignored. These models simplify reality more than may desirable for some applications, although they represent the most tractable levels of complexity possible in the circumstances of this thesis: more complex models could not be developed due to the unavailability of other data that would have been required and/or due to a lack of specialist technical expertise.

The habitat provision model represents a different approach to modelling, i.e. construction of indices. This approach uses metrics calculated from easily observed/recorded data (in this case land cover) to indicate something that is associated with those data, but more difficult to record (in this case, biodiversity). Biodiversity is difficult to model directly at the spatial scales studies used in this study, requiring habitat models for each species; but the association of species richness with particular aspects of land cover is quite strong (Section 8.2). Therefore the use of indicator metrics is justified in order that biodiversity can be included in the portfolio of ecosystem services. However, the use of indices rather than direct modelling, i.e. quantifying habitat provision rather than biodiversity itself, introduces a different source of uncertainty, namely imperfect correlations between the index metrics and species richness.

The idea of the production function approach adopted in this thesis is to strike a balance between greater reliability/depth/accuracy of single models (for example, with the more complex air pollution model) and greater breadth of different ecosystem services. This balance is probably best met (for the purposes of this thesis) by models such as the heat island mitigation and stormwater runoff reduction models, which are not so complex as to require a large time investment to understand and implement the models, but include a variety of factors that influence the output.

The fact that the models have different degrees of accuracy potentially has consequences for the interpretation of both statistical and heuristic analyses. Unreliable models may not place the correct emphasis on different factors affecting the ecosystem services and thus introduce patterns that, in reality, do not occur; it is equally possible that failure to include factors may obscure real patterns. For this reason it may be desirable to place greater weight on the models more likely to produce accurate results when making decisions using the output of a suite of production function models.

Another consequence of the production function approach, at least as far as the models included in this thesis are concerned, is that the same spatial input data (land cover, soils) are used in multiple ecosystem services. This is liable to introduce similar spatial patterning into the output of different models. It does not mean that areas with the lowest levels of one ecosystem service will necessarily have the lowest levels of other ecosystem services, but it does mean that many of the areas with low levels of one ecosystem service are likely to have low levels of other ecosystem services.

This fact does not invalidate the production function approach. Although multiple models use the same spatial input data, each model uses these in combination with different, independently derived variables and parameters, i.e. each model uses these spatial data in different ways, thereby producing different results. In reality, certain land cover types, for example, are associated with higher levels of several ecosystem services (e.g. woodlands), which introduces strong correlations between the services and also similar patterns of spatial autocorrelation within services (Chapter 9). However, it is important to realise that this is not an artefact of the use of the same input data for multiple models.

12.4. Beyond the case study

This thesis has used a case study to demonstrate the possibility of modelling urban ecosystem services at local scales, and the potential of the resulting model outputs to address urban planning and socioeconomic issues. The models presented in Chapter 3 to Chapter 8 can, however, be implemented for any location dependent upon the availability of the following spatial data:

- Land cover (necessary for all models except opportunities for cultural ecosystem services).
- Land use (necessary for opportunities for cultural ecosystem services model).
- Soils data:
    - Texture (highly recommended for heat island mitigation model, although it is possible to run the model without specific knowledge of soils; and necessary for stormwater runoff reduction model).
    - Density, temperature (recommended for heat island mitigation model, but could use estimates).
    - Carbon content (necessary for the soil component of carbon storage).

- Air pollution levels (optional for air pollution reduction model – is important to understanding the ecosystem service, but not important for the model results as presented in this thesis).
- (If the location is outside of the UK, alternative non-spatial parameters might also be required, e.g. climate data for the air pollution model, vegetation carbon storage.)

With the exception of the Historic Environment Character and (optional) AIRVIRO air pollution datasets, all of the datasets used in this thesis are available UK-wide. The Historic Environment Character dataset is part of an on-going programme that aims to cover all of England (South Yorkshire Archaeology Service 2005). The census data used as socioeconomic descriptors and for some of the urban morphology metrics is also UK-wide, meaning that the approach used here to measure urban morphology and socioeconomic conditions can also be implemented for any place in the UK. Thus, in terms of the aims of the thesis, the results from this case study are applicable across all English urban areas.

In satisfying the research aims, this thesis has also presented a number of interesting findings about patterns of ecosystem service production, and relationships between urban morphology/socioeconomic conditions and ecosystem services, which relate specifically to Sheffield. These findings arise because of Sheffield's unique patterns of land cover and land use (including patterns of residence), as well as its geographical and geological setting. The present configuration of these factors has been determined by historical factors that may not be applicable elsewhere; and consequently, it is not immediately clear whether the conclusions drawn can be generalised to other urban locations.

Sheffield developed where it is because of the abundance of raw materials for iron and steel production, and rivers suitable for industry and energy generation (Hey 1998, Sheffield City Council 2009a). The boundaries of Sheffield as used in this study, i.e. the boundaries of the metropolitan borough, exist as they are due to somewhat arbitrary political decisions to divide up land and population amongst appropriate administrative regions. It is as a consequence of these decisions that the metropolitan borough of Sheffield includes a large expanse of agricultural land and peaty moorland to the west, as well as quite a large city to the east.

The heavy industry and mining that caused Sheffield to expand and later to suffer a severe recession is also partly responsible for many of the city's idiosyncratic features (Hey 1998). The need to provide the population with respite from the polluted city saw the origins of many parks, which make Sheffield renowned for being a "green city" to this day (Hey 1998, Sheffield City Council 2009a). It is also at least partially as a consequence of the industry that areas to the west of the city centre tend to house more affluent residents: when Sheffield was heavily polluted, richer residents built their houses in the areas receiving fresh air on the westerly winds from the Peak District (Hey 1998). Even today, the areas with an industrial heritage tend to be more socioeconomically deprived (Hey 1998). However, the urban renewal prompted by the recession of the 1970s is one reason why there are some less deprived neighbourhoods in traditionally industrial areas.

It seems likely that the hilly topographical setting is also a direct cause of some of Sheffield's characteristics. Many of the city's parks and greenspaces are located on steep hills and cliffs: in other words, some of the greenspace exists where it is only because it would be impossible to build there. The upland nature of the Peak District is probably also one reason why the western part of Sheffield never became heavily populated, although latterly this is also due to the creation of the National Park.

It is obvious, then, that the high levels of ecosystem service production to the west of the study area, and many of the urban morphology and socioeconomic patterns observed in the city, are the consequence of a combination of historical events unique to Sheffield. This might suggest that the specific findings of the analyses performed in this thesis are not applicable elsewhere.

On the other hand, there are features of socioeconomic conditions, urban morphology, and factors related to ecosystem service provision that tend to show similar gradients from the centre of a city to its suburbs, at least in the UK (Tratalos et al. 2007, Dempsey et al. 2010). Investigations of one inner, one middle and one outer city neighbourhood from each of five UK cities (including Sheffield) found a number of consistent trends from inner to outer areas (Tratalos et al. 2007, Dempsey et al. 2010). These trends, summarised in Table 12.1, are also applicable to the trends seen in the urbanised part of the city of Sheffield (see Appendix D).

If only this part of Sheffield, and not its more rural regions, is considered, then the results from the present thesis might become generalisable. Moreover, the Output Area and Historic Environment Character GIS layers have a relatively small proportion of the

Table 12.1. Trends from inner to outer city neighbourhoods, identified from five UK cities. (Source: Tratalos et al. 2007, Dempsey et al. 2010)

	Inner neighbourhoods	**Outer neighbourhoods**
Relating to urban morphology		
Population density	Higher	Lower
Transport infrastructure	Well-connected, grid-like	Tree-like with a few “spine” main roads
Housing type	Detached/semi-detached rarer	Detached/semi-detached more common
Relating to socioeconomic conditions		
Social grade composition	No pattern	No pattern
Demographic composition	More young families without children	More older families
Housing tenure	Private renting common	Almost exclusively owner occupied
Social housing	Spatially concentrated, mostly in inner neighbourhoods	-
Relating to ecosystem services		
Proportion of greenspace	Lower	Higher
Carbon storage	Lower	Higher (often highest in middle neighbourhoods)
Stormwater runoff	Higher	Lower
Maximum temperature	Higher	Lower

polygons in rural areas (Figure 2.4, Figure 2.5). This means that rural areas comprise only a small part of the total sample size in statistical analyses and thus have little effect on the results, compared to suburban and rural areas.

Specific patterns will vary between cities, but there might also be some trends expected. For example, the east half of Sheffield is more deprived, whereas it might be the west in another city; although in the UK, where the prevailing winds (which bring clean air) are from the west, this particular pattern may be more common. Localised urban regeneration, which has been necessary in the past century in industrial cities throughout the UK, will cause unique small scale trends. In Sheffield this regeneration is occurring largely along its inner city rivers; this might also be seen in other cities where industry was historically focused along geographically central canals or navigable rivers.

Nevertheless, the existence of larger scale general patterns in urban morphology and socioeconomics that are consistent between cities (Table 12.1), in combination with the fact that the majority of the sample size in the present study is made up of urban and suburban data points, means that the main conclusions about relationships between ecosystem service production, urban morphology and socioeconomic conditions probably to some extent hold for other UK cities.

12.5. Implications and future directions

This thesis has proven the potential of using combinations of existing datasets and methods to spatially explicitly model a variety of ecosystem services, produce urban planning recommendations and investigate important social issues. This is in spite of difficulties in implementing and interpreting statistical analysis, and the fact that the model output is associated with unquantified uncertainty (Section 12.3). This research contributes to efforts to undertake spatial ecosystem service assessments using the "production function" approach (Section 1.3.3.4) by compiling a toolbox that can be used in urban contexts. As spatially explicit assessments at relevant scales are important for integrating ecosystem services into management and decision making processes (Chan et al. 2006, Naidoo et al. 2008, Nelson et al. 2009), and because urbanisation poses such a threat to ecosystem services (Section 1.2) yet urban areas are home to more than half the world's population (United Nations Population Fund 2007), it is critical that urban ecosystem services are represented in this field of research.

Naturally, this toolbox is a work in progress and many challenges remain. In particular, there are a variety of urban ecosystem services that were not modelled in the present thesis (see Table 1.2 for examples) due either to lack of data or methods; conclusions drawn about ecosystem services in general from the existing models may be erroneous when different ecosystem services are also taken into consideration. Furthermore, it has also not been possible to go beyond production functions to what might be called supply functions, i.e. linking the production of ecosystem services to the needs for, and values of, those services to the urban population via socioeconomic data. Thus, although the approaches developed in this thesis have succeeded in generating informative results, the potential capabilities of the toolbox are far from being attained.

Perhaps a more philosophical issue with the approach taken in this thesis, and which applies to spatial assessments more generally, is the use of a spatial boundary to the modelled area. The boundary used here, i.e. the extent of Sheffield's administrative boundary, was chosen as it is relevant to the scale at which decisions are made. However, the spatiotemporal characteristics of ecosystem services mean that the beneficiaries of a service are not necessarily located at the site of production (Section 1.2.1). For example, the beneficiaries of stormwater runoff reduction are the down-catchment population, while those of carbon storage are the global population (Table 9.1). In both cases, the beneficiaries do not reside within the boundaries of the assessment.

Moreover, urban areas have a large ecosystem service “footprint” due to the consumption of food, water, and other material products originating from outside of the urban boundaries (Rees and Wackernagel 2008). Trade-offs mean that the production of these services for consumption by urban residents can be detrimental to other services produced in rural areas (Rodríguez et al. 2006). Similarly, some ecosystem services (especially cultural services) can be consumed by urban residents temporarily leaving the urban area to visit sites of service production. In both situations, ecosystem services are effectively being imported into cities, with the result that cities influence areas far beyond their boundaries, often detrimentally (Rodríguez et al. 2006, Rees and Wackernagel 2008).

It is also possible for ecosystem services produced within a city to benefit people living outside a city. The reduction of air pollution by urban vegetation, for example, improves air quality downwind of the city as well (although it is likely that the city is a substantial net source of air pollution). Similarly, rural residents may visit urban parks and thereby benefit from cultural ecosystem services.

For these reasons, it is ultimately desirable to be able to include multiple spatial scales in ecosystem service assessments, so that the consequences of changes at any one scale can be assessed at other scales. The Multi-scale Integrated Models of Ecosystem Services (MIMES) project has this as its aim (http://www.uvm.edu/giee/mimes/ accessed 29/03/2010), although it is at present at a very early stage.

A final key area that might be highlighted for future work is that of sustainability. A quotation used in Chapter 1, from Williams et al. (2000b), is repeated here:

> “A form [i.e. urban morphology] is taken to be sustainable if it: enables the city to function within its natural and man-made carrying capacities; is ‘user-friendly’ for its occupants; and promotes social equity.”

The analyses performed in this thesis have not addressed the issue of whether the urban morphology of Sheffield is sustainable in terms of whether the consequential levels of ecosystem services provide an adequate quality of life for its residents, or whether the current levels of social inequity in ecosystem service production are sufficient to inhibit the social functioning of the city. This is partially due to the inability to link ecosystem service production to supply and beneficiaries, but also due to the difficulties in defining thresholds for sustainability. Should threshold sustainable levels of ecosystem service supply be defined by stakeholder involvement or by expert assessment? How can interpersonal variations in ecosystem service requirements be accounted for?

Nevertheless, at the simplest level, in the context of this study it can be assumed that higher levels of ecosystem service production are "more" sustainable. In this light, perhaps the most relevant conclusion is that urban morphologies of the kind typically implemented in areas of housing for some of the least well-off groups of society in fact provide average levels of ecosystem services (Section 12.2). This indicates that socioeconomic inequity and ecosystem service inequity are not necessarily exactly spatially correlated, and provides a starting point for designing more sustainable cities.

References

Abrahams, M. J., Price, J., Whitlock, F. A. & Williams, G. (1976) The Brisbane floods, January 1974: their impact on health. *The Medical Journal of Australia,* **2,** 936-9.

Adema, W. & Ladaique, M. (2009) How expensive is the welfare state? Gross and net indicators in the OECD Social Expenditure Database (SOCX). OEDC Social, Employment and Migration WOrking Papers, no. 92. Retrieved from http://www.oecd.org/document/9/0,3343,en_2649_34637_38141385_1_1_1_1,00.html, accessed 15/07/2010. OECD Publishing, Paris.

Adler, N. E. & Ostrove, J. M. (1999) Socioeconomic status and health: what we know and what we don't. *Annals of the New York Academy of Sciences,* **896,** 3-15.

Alberti, M. (2005) The effects of urban patterns on ecosystem function. *International Regional Science Review,* **28,** 168.

Alexandri, E. & Jones, P. (2008) Temperature decreases in an urban canyon due to green walls and green roofs in diverse climates. *Building and Environment,* **43,** 480-493.

Anderson, B. J., Armsworth, P. R., Eigenbrod, F., Thomas, C. D., Gillings, S., Heinemeyer, A., Roy, D. B. & Gaston, K. J. (2009) Spatial covariance between biodiversity and other ecosystem service priorities. *Journal of Applied Ecology,* **46,** 888-896.

Andersson, E., Barthel, S. & Ahrné, K. (2007) Measuring social-ecological dynamics behind the generation of ecosystem services. *Ecological Applications,* **17,** 1267-1278.

Armsworth, P. R., Chan, K. M. A., Daily, G. C., Ehrlich, P. R., Kremen, C., Ricketts, T. H. & Sanjayan, M. A. (2007) Ecosystem-service science and the way forward for conservation. *Conservation Biology,* **21,** 1383-1384.

Arnfield, A. J. (2003) Two decades of urban climate research: a review of turbulence, exchanges of energy and water, and the urban heat island. *International Journal of Climatology,* **23,** 1-26.

Arnold, C. L. & Gibbons, C. J. (1996) Impervious surface coverage: the emergence of a key environmental indicator. *Journal of the American Planning Association,* **62,** 243-258.

Balvanera, P., Pfisterer, A. B., Buchmann, N., He, J. S., Nakashizuka, T., Raffaelli, D. & Schmid, B. (2006) Quantifying the evidence for biodiversity effects on ecosystem functioning and services. *Ecology Letters,* **9,** 1146-1156.

Barbosa, O., Tratalos, J. A., Armsworth, P. R., Davies, R. G., Fuller, R. A., Johnson, P. & Gaston, K. J. (2007) Who benefits from access to green space? A case study from Sheffield, UK. *Landscape and Urban Planning,* **83,** 187-195.

Barthel, S., Colding, J., Elmqvist, T. & Folke, C. (2005) History and local management of a biodiversity-rich, urban cultural landscape. *Ecology and Society,* **10,** 10.

Bass, B., Krayenhoff, E. S., Martilli, A., Stull, R. B. & Auld, H. (2003) The impact of green roofs on Toronto's urban heat island. *Proceedings of the First North American Green Roof Conference: Greening Rooftops for Sustainable Communities, 26-30 May 2003, Chicago*. Cardinal Group.

BBC News (2007) Floods 'will cost Sheffield millions'. *BBC News Online, 12/07/2007*. Retrieved from http://news.bbc.co.uk/1/hi/england/south_yorkshire/6896710.stm, accessed 19/08/2009.

Beer, A. R. (2003) Greenstructure and Urban Planning - Case Study - Sheffield, UK. Report for COST C11 Project. Retrieved from http://www.greenstructureplanning.eu/COSTC11/sheffield-case.htm, accessed 13/04/2010.

BenDor, T., Brozovic, N. & Pallathucheril, V. G. (2008) The social impacts of wetland mitigation policies in the United States. *Journal of Planning Literature,* **22,** 341-357.

Bennet, G. (1970) Bristol floods 1968. Controlled survey of effects on health of local community disaster. *British Medical Journal,* **3,** 454-458.

Bierwagen, B. G. (2005) Predicting ecological connectivity in urbanizing landscapes. *Environment and Planning B: Planning and Design,* **32,** 763-776.

Bisgrove, R. & Hadley, P. (2002) Gardening in the Global Greenhouse. The Impacts of Climate Change on Gardens in the UK. Summary Report. UKCIP, Oxford.

Bivand, R., with contributions by Anselin, L., Assunçaõ, R., Berke, O., Bernat, A., Carvalho, M., Chun, Y., Dormann, C., Dray, S., Halbersma, R., Krainski, E., Lewin-Koh, N., Li, H., Ma, J., Millo, G., Mueller, W., Ono, H., Peres-Neto, P., Reder, M., Tiefelsdorf, M. & Yu, D. (2009) spdep: Spatial dependence: weighting schemes, statistics and models. R package version 0.4-34.

Bolger, D. T., Suarez, A. V., Crooks, K. R., Morrison, S. A. & Case, T. J. (2000) Arthropods in urban habitat fragments in southern California: area, age, and edge effects. *Ecological Applications,* **10,** 1230-1248.

Bolund, P. & Hunhammar, S. (1999) Ecosystem services in urban areas. *Ecological Economics,* **29,** 293-301.

Booth, D. B., Visitacion, B. & Steinemann, A. C. (2006) Damages and Costs of Stormwater Runoff in the Puget Sound Region. *The Water Center, Department of Civil and Environmental Engineering, University of Washington*. Seattle, WA.

Booth, M. L., Owen, N., Bauman, A., Clavisi, O. & Leslie, E. (2000) Social–cognitive and perceived environment influences associated with physical activity in older Australians. *Preventive Medicine,* **31,** 15-22.

Bouchard, J., Ford, A. T., Eigenbrod, F. E. & Fahrig, L. (2009) Behavioral responses of northern leopard frogs (Rana pipiens) to roads and traffic: implications for population persistence. *Ecology and Society,* **14,** 23.

Bramley, G., Brown, C., Dempsey, N., Power, S. & Watkins, D. (2010) Chapter 5. Social acceptability. *Dimensions of the Sustainable City* (eds M. Jenks & C. Jones). Springer, London.

Breiman, L. (2001) Random forests. *Machine Learning,* **45,** 5-32.

Brimblecombe, P. (2001) Ch 1. Urban Air Pollution. *The urban atmosphere and its effects. Air pollution reviews - Vol. 1* (eds P. Brimblecombe & R. L. Maynard). Imperial College Press, London.

Brook, J. R., Zhang, L., Di-Giovanni, F. & Padro, J. (1999) Description and evaluation of a model of deposition velocities for routine estimates of air pollutant dry deposition over North America. Part I: model development. *Atmospheric Environment,* **33,** 5037-5051.

Buck, A. L. (1981) New equations for computing vapor pressure and enhancement factor. *Journal of Applied Meteorology,* **20,** 1527-1532.

Buck Research Manual (1996) Updated equation from Buck (1981), cited at http://en.wikipedia.org/wiki/Arden_Buck_Equation, accessed 06/11/2009.

Burton, E. (2000) The compact city: Just or just compact? A preliminary analysis. *Urban Studies,* **37,** 1969-2006.

Buyantuyev, A. & Wu, J. (2010) Urban heat islands and landscape heterogeneity: linking spatiotemporal variations in surface temperatures to land-cover and socioeconomic patterns. *Landscape ecology,* **25,** 17-33.

Buzzelli, M. & Jerrett, M. (2004) Racial gradients of ambient air pollution exposure in Hamilton, Canada. *Environment and Planning A,* **36,** 1855-1876.

Byrne, L. B., Bruns, M. A. & Kim, K. C. (2008) Ecosystem properties of urban land covers at the aboveground–belowground interface. *Ecosystems,* **11,** 1065-1077.

Cabin, R. J. & Mitchell, R. J. (2000) To Bonferroni or not to Bonferroni: when and how are the questions. *Bulletin of the Ecological Society of America,* **81,** 246-248.

Caliman, A., Pires, A. F., Esteves, F. A., Bozelli, R. L. & Farjalla, V. F. (2010) The prominence of and biases in biodiversity and ecosystem functioning research. *Biodiversity and Conservation,* **19,** 651-664.

Cambridge Systematics (2005) Cool Pavement Report. EPA Cool Pavements Study - Task 5. Draft report. United States Environmental Protection Agency.

Cardinale, B. J., Srivastava, D. S., Duffy, J. E., Wright, J. P., Downing, A. L., Sankaran, M. & Jouseau, C. (2006) Effects of biodiversity on the functioning of trophic groups and ecosystems. *Nature,* **443,** 989-992.

Centers for Disease Control and Prevention. (2000) Morbidity and mortality associated with hurricane Floyd - North Carolina, September-October 1999. *Morbidity and Mortality Weekly Report,* **49,** 369-370.

Chadwick, M. A., Dobberfuhl, D. R., Benke, A. C., Huryn, A. D., Suberkropp, K. & Thiele, J. E. (2006) Urbanization affects stream ecosystem function by altering hydrology, chemistry, and biotic richness. *Ecological Applications,* **16,** 1796-1807.

Chamberlain, D. E., Cannon, A. R., Toms, M. P., Leech, D. I., Hatchwell, B. J. & Gaston, K. J. (2009) Avian productivity in urban landscapes: a review and meta-analysis. *Ibis,* **151,** 1-18.

Chan, K. M. A., Shaw, M. R., Cameron, D. R., Underwood, E. C. & Daily, G. C. (2006) Conservation planning for ecosystem services. *PLoS Biology,* **4,** e379.

Chee, Y. E. (2004) An ecological perspective on the valuation of ecosystem services. *Biological Conservation,* **120,** 549-565.

Chen, W. Y. & Jim, C. Y. (2008) Cost–benefit analysis of the leisure value of urban greening in the new Chinese city of Zhuhai. *Cities,* **25,** 298-309.

Cho, S., Makar, P. A., Lee, W. S., Herage, T., Liggio, J., Li, S. M., Wiens, B. & Graham, L. (2009) Evaluation of a unified regional air-quality modeling system (AURAMS) using PrAIRie2005 field study data: The effects of emissions data accuracy on particle sulphate predictions. *Atmospheric Environment,* **43,** 1864-1877.

Christensen, N. L., Bartuska, A. M., Brown, J. H., Carpenter, S., D'Antonio, C., Francis, R., Franklin, J. F., MacMahon, J. A., Noss, R. F. & Parsons, D. J. (1996) The report of the Ecological Society of America committee on the scientific basis for ecosystem management. *Ecological Applications,* **6,** 665-691.

CIBSE. (1982) *Guide A2 - Weather and Solar Data,* Chartered Institution of Building Services Engineers, London.

CIBSE. (1999) *Guide A - Environmental Design,* Chartered Institution of Building Services Engineers, London.

Clarke, A. G. & Hamilton, R. S. (no date) URGENT Project. Evolution of the particle size distribution of vehicular emissions in the urban atmosphere. Project report. Retrieved from http://badc.nerc.ac.uk/data/urgent/Projects-Doc/vehiem-report.pdf, accessed 27/07/2009.

Clergeau, P., Jokimäki, J. & Savard, J. P. L. (2001) Are urban bird communities influenced by the bird diversity of adjacent landscapes? *Journal of Applied Ecology,* **38,** 1122-1134.

Clergeau, P., Savard, J. P. L., Mennechez, G. & Falardeau, G. (1998) Bird abundance and diversity along an urban-rural gradient: a comparative study between two cities on different continents. *The Condor,* **100,** 413-425.

Cliff, A. D. & Ord, J. K. (1981) *Spatial processes: models and applications,* Page Bros., Norwich, UK.

Clifford, P., Richardson, S. & Hémon, D. (1989) Assessing the significance of the correlation between two spatial processes. *Biometrics*, 123-134.

Cohen, D. A., Ashwood, J. S., Scott, M. M., Overton, A., Evenson, K. R., Staten, L. K., Porter, D., McKenzie, T. L. & Catellier, D. (2006) Public parks and physical activity among adolescent girls. *Pediatrics,* **118,** e1381.

Colding, J. & Folke, C. (2009) The role of golf courses in biodiversity conservation and ecosystem management. *Ecosystems,* **12,** 191-206.

Colding, J., Lundberg, J. & Folke, C. (2006) Incorporating green-area user groups in urban ecosystem management. *AMBIO: A Journal of the Human Environment,* **35,** 237-244.

Coles, R. W. & Bussey, S. C. (2000) Urban forest landscapes in the UK - progressing the social agenda. *Landscape and Urban Planning,* **52,** 181-188.

Coley, R. L., Kuo, F. E. & Sullivan, W. C. (1997) Where does community grow? *Environment and Behavior,* **29,** 468-494.

Communities and Local Government (2008a) Cost analysis of the code for sustainable homes. Final report. Communities and Local Government, London.

Communities and Local Government (2008b) Planning Policy Statement 12: creating strong safe and prosperous communities through Local Spatial Planning. The Stationery Office, London.

Costanza, R. (2000) Social goals and the valuation of ecosystem services. *Ecosystems,* **3,** 4-10.

Costanza, R. (2008) Ecosystem services: multiple classification systems are needed. *Biological Conservation,* **141,** 350-352.

Costanza, R., d'Arge, R., De Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R. V. & Paruelo, J. (1997) The value of the world's ecosystem services and natural capital. *Nature,* **387,** 253-260.

Costanza, R., d'Arge, R., De Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R. V. & Paruelo, J. (1998) The value of the world's ecosystem services and natural capital. *Ecological Economics,* **25,** 3-15.

Crabtree, S. (2009) Floods bring more misery to Sheffield. *The Star, 10/06/2009.* Sheffield. Retrieved from http://www.thestar.co.uk/news/LATEST-Floods-bring-more-misery.5354677.jp, accessed 27/04/2010.

Cranfield University (2009) LandIS - Land Information System - digital soils data. Retrieved from http://www.landis.org.uk/data/index.cfm, accessed 13/11/2009.

Crooks, K. R. (2002) Relative sensitivities of mammalian carnivores to habitat fragmentation. *Conservation Biology,* **16,** 488-502.

Cruickshank, M. M., Tomlinson, R. W. & Trew, S. (2000) Application of CORINE land-cover mapping to estimate carbon stored in the vegetation of Ireland. *Journal of Environmental Management,* **58,** 269-287.

Currie, D. J. (2007) Disentangling the roles of environment and space in ecology. *Journal of Biogeography,* **34,** 2009-2011.

Cutler, D. R., Edwards Jr, T. C., Beard, K. H., Cutler, A., Hess, K. T., Gibson, J. & Lawler, J. J. (2007) Random forests for classification in ecology. *Ecology,* **88,** 2783-2792.

Cutts, B. B., Darby, K. J., Boone, C. G. & Brewis, A. (2009) City structure, obesity, and environmental justice: An integrated analysis of physical and social barriers to walkable streets and park access. *Social Science & Medicine,* **69,** 1314-1322.

Daily, G. C., Alexander, S., Ehrlich, P. R., Goulder, L., Lubchenco, J., Matson, P. A., Mooney, H. A., Postel, S., Schneider, S. H. & Tilman, D. (1997) Ecosystem services: benefits supplied to human societies by natural ecosystems. *Issues in Ecology,* **1,** 1-18.

Daily, G. C., Polasky, S., Goldstein, J., Kareiva, P. M., Mooney, H. A., Pejchar, L., Ricketts, T. H., Salzman, J. & Shallenberger, R. (2009) Ecosystem services in decision making: time to deliver. *Frontiers in Ecology and the Environment,* **7,** 21-28.

Davies, R. G., Barbosa, O., Fuller, R. A., Tratalos, J., Burke, N., Lewis, D., Warren, P. H. & Gaston, K. J. (2008) City-wide relationships between green spaces, urban land use and topography. *Urban Ecosystems,* **11,** 269-287.

De'ath, G. & Fabricius, K. E. (2000) Classification and regression trees: a powerful yet simple technique for ecological data analysis. *Ecology,* **81,** 3178-3192.

De Deyn, G. B., Cornelissen, J. H. C. & Bardgett, R. D. (2008) Plant functional traits and soil carbon sequestration in contrasting biomes. *Ecology Letters,* **11,** 516-531.

De Groot, R. (2006) Function-analysis and valuation as a tool to assess land use conflicts in planning for sustainable, multi-functional landscapes. *Landscape and Urban Planning,* **75,** 175-186.

De Groot, R. S., Wilson, M. A. & Boumans, R. M. J. (2002) A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecological Economics,* **41,** 393-408.

de Vries, S., Verheij, R. A., Groenewegen, P. P. & Spreeuwenberg, P. (2003) Natural environments-healthy environments? An exploratory analysis of the relationship between greenspace and health. *Environment and Planning A,* **35,** 1717-1732.

DEFRA (2009) Local air quality management technical guidance LAQM.TG(09). Department for Environment Food and Rural Affairs.

Del Barrio, E. P. (1998) Analysis of the green roofs cooling potential in buildings. *Energy and Buildings,* **27,** 179-193.

Dempsey, N., Brown, C., Raman, S., Porta, S., Jenks, M., Jones, C. & Bramley, G. (2010) Chapter 2. Elements of urban form. *Dimensions of the Sustainable City* (eds M. Jenks & C. Jones). Springer, London.

Diniz-Filho, J. A. F., Bini, L. M. & Hawkins, B. A. (2003) Spatial autocorrelation and red herrings in geographical ecology. *Global Ecology and Biogeography,* **12,** 53-64.

Dormann, C. F., McPherson, J. M., Araújo, M. B., Bivand, R., Bolliger, J., Carl, G., Davies, R. G., Hirzel, A., Jetz, W. & Danielkissling, W. (2007) Methods to account for spatial autocorrelation in the analysis of species distributional data: a review. *Ecography,* **30,** 609-628.

Duffy, J. E. (2009) Why biodiversity is important to the functioning of real-world ecosystems. *Frontiers in Ecology and the Environment,* **7,** 437-444.

Dutilleul, P., Clifford, P., Richardson, S. & Hemon, D. (1993) Modifying the t test for assessing the correlation between two spatial processes. *Biometrics*, **305-314.**

Egoh, B., Reyers, B., Rouget, M., Richardson, D. M., Le Maitre, D. C. & van Jaarsveld, A. S. (2008) Mapping ecosystem services for planning and management. *Agriculture, Ecosystems & Environment,* **127,** 135-140.

Ehrenfeld, J. G. (2004) The expression of multiple functions in urban forested wetlands. *Wetlands,* **24,** 719-733.

Eigenbrod, F., Armsworth, P. R., Anderson, B. J., Heinemeyer, A., Gillings, S., Roy, D. B., Thomas, C. D. & Gaston, K. J. (2010) The impact of proxy-based methods on mapping the distribution of ecosystem services. *Journal of Applied Ecology,* **47,** 377-385.

Elleker, A. (2009) 2009 air quality updating and screening assessment for Sheffield City Council. Sheffield City Council, Sheffield.

Environment Agency (2004) Review of building parameters for development of a soil vapour intrusion model. R&D project record p5-079. Technical report. Environment Agency, Bristol.

Environmental Protection Agency (2004) AERMOD deposition algorithms - science document (revised draft). United States Environmental Protection Agency.

Escobedo, F. J., Wagner, J. E., Nowak, D. J., De la Maza, C. L., Rodriguez, M. & Crane, D. E. (2008) Analyzing the cost effectiveness of Santiago, Chile's policy of using urban forests to improve air quality. *Journal of Environmental Management,* **86,** 148-157.

Evans, K. L., Newson, S. E. & Gaston, K. J. (2009) Habitat influences on urban avian assemblages. *Ibis,* **151,** 19-39.

Fenger, J. (2009) Ch 10. Urban air pollution. *Atmospheric Science for Environmental Scientists* (eds C. N. Hewitt & A. V. Jackson). Wiley-Blackwell, Chichester, UK.

Fine, D. (2003) *History and Guide: Sheffield. Revised edition,* Tempus Publishing Ltd., Stroud, Gloustershire.

Fisher, P., Comber, A. J. & Wadsworth, R. (2005) Chapter 6. Land use and land cover: contradiction or complement. *In: Re-presenting GIS* (eds P. Fisher & D. Unwin). John Wiley & Sons, Chicester, UK.

Folke, C., Carpenter, S., Walker, B., Scheffer, M., Elmqvist, T., Gunderson, L. & Holling, C. S. (2004) Regime shifts, resilience, and biodiversity in ecosystem management. *Annual Review of Ecology, Evolution, and Systematics,* **35,** 557-581.

Fortin, M.-J. & Dale, M. R. T. (2005) *Spatial analysis: a guide for ecologists,* Cambridge University Press, Cambridge, UK.

Frank, L. D., Sallis, J. F., Conway, T. L., Chapman, J. E., Saelens, B. E. & Bachman, W. (2006) Many pathways from land use to health: associations between neighborhood walkability and active transportation, body mass index, and air quality. *Journal of the American Planning Association,* **72,** 75-87.

Frumkin, H. (2002) Urban sprawl and public health. *Public Health Reports,* **117,** 201-207.

Fuller, R., Smith, G., Sanderson, R., Hill, A., Thomson, R., Cox, N., Brown, R., Clarke, P., Rothery, P. & Gerard, F. (2002) Land Cover Map 2000: A guide to the classification system. Extract from the draft final report. Technical report CSLCM/Final. Centre for Ecology and Hydrology, Monk's Wood.

Fuller, R. A., Irvine, K. N., Devine-Wright, P., Warren, P. H. & Gaston, K. J. (2007) Psychological benefits of greenspace increase with biodiversity. *Biology Letters,* **3,** 390-394.

Fuller, R. A., Tratalos, J., Warren, P. H., Davies, R. G., Pępkovska, A. & Gaston, K. J. (2010) Chapter 4. Environment and biodiversity. *Dimensions of the Sustainable City. Future City 2.* (eds M. Jenks & C. Jones). Springer, London.

Fuller, R. A., Warren, P. H., Armsworth, P. R., Barbosa, O. & Gaston, K. J. (2008) Garden bird feeding predicts the structure of urban avian assemblages. *Diversity and Distributions,* **14,** 131-137.

Gaffield, S. J., Goo, R. L., Richards, L. A. & Jackson, R. J. (2003) Public health effects of inadequately managed stormwater runoff. *American Journal of Public Health,* **93,** 1527-1533.

Garden, J., McAlpine, C., Peterson, A., Jones, D. & Possingham, H. (2006) Review of the ecology of Australian urban fauna: A focus on spatially explicit processes. *Austral Ecology,* **31,** 126-148.

Gartland, L. (2008) *Heat Islands: Understanding and Mitigating Heat in Urban Areas,* Earthscan, London.

Gates, P. (2002) Gardening in the global greenhouse. The impacts of climate change on gardens in the UK. Summary report. UK Climate Impacts Programme, Oxford.

Gerber, H. E. (1985) Relative-humidity parameterization of the Navy Aerosol Module (NAM). NRL report 8956. Naval Research Laboratory, Washington D.C.

Getter, K. L., Rowe, D. B. & Andresen, J. A. (2007) Quantifying the effect of slope on extensive green roof stormwater retention. *Ecological Engineering,* **31,** 225-231.

Giles-Corti, B., Broomhall, M. H., Knuiman, M., Collins, C., Douglas, K., Ng, K., Lange, A. & Donovan, R. J. (2005) Increasing walking: how important is distance to, attractiveness, and size of public open space? *American Journal of Preventive Medicine,* **28,** 169-176.

Gill, S. E. (2006) Climate Change and Urban Greenspace. PhD thesis. *School of Environment and Development*. University of Manchester.

Gill, S. E., Handley, J. F., Ennos, A. R. & Pauleit, S. (2007) Adapting cities for climate change: the role of the green infrastructure. *Built Environment,* **33,** 115-133.

Giridharan, R., Lau, S. S. Y., Ganesan, S. & Givoni, B. (2007) Urban design factors influencing heat island intensity in high-rise high-density environments of Hong Kong. *Building and Environment,* **42,** 3669-3684.

Golubiewski, N. E. (2006) Urbanization increases grassland carbon pools: effects of landscaping in Colorado's front range. *Ecological Applications,* **16,** 555-571.

Gong, W., Dastoor, A. P., Bouchet, V. S., Gong, S., Makar, P. A., Moran, M. D., Pabla, B., Ménard, S., Crevier, L. P. & Cousineau, S. (2006) Cloud processing of gases and aerosols in a regional air quality model (AURAMS). *Atmospheric Research,* **82,** 248-275.

Gordon, P. & Richardson, H. W. (1997) Are compact cities a desirable planning goal? *Journal of the American Planning Association,* **63,** 95-106.

Grimm, N. B., Grove, J. M., Pickett, S. T. A. & Redman, C. L. (2000) Integrated approaches to long-term studies of urban ecological systems. *Urban Ecology***,** 123-141.

Guardian.co.uk (2008) Insurance premiums rise with flood waters. *Guardian Online, 12/01/2008*. Retrieved from http://www.guardian.co.uk/money/2008/jan/17/homeinsurance.insurance, accessed 19/08/2009.

Guo, L. B. & Gifford, R. M. (2002) Soil carbon stocks and land use change: a meta analysis. *Global Change Biology,* **8,** 345-360.

Haines-Young, R. (2009) Land use and biodiversity relationships. *Land Use Policy,* **26,** 178-186.

Haines-Young, R. & Potschin, M. (2008) England's terrestrial ecosystem services and the rationale for an ecosystem approach. Full technical report, DEFRA project code NR0107. Department for Environment, Food and Rural Affairs.

Hampicke, U. (1994) Ethics and economics of conservation. *Biological Conservation,* **67,** 219-231.

Handley, J., Pauleit, S., Slinn, P., Barber, A., Baker, M., Jones, C. & Lindley, S. (2003) Accessible natural green space standards in towns and cities: a review and toolkit for their implementation. English Nature research reports number 526. English Nature, Peterborough, UK.

Harlan, S. L., Brazel, A. J., Prashad, L., Stefanov, W. L. & Larsen, L. (2006) Neighborhood microclimates and vulnerability to heat stress. *Social Science & Medicine,* **63,** 2847-2863.

Hartig, T., Evans, G. W., Jamner, L. D., Davis, D. S. & Gärling, T. (2003) Tracking restoration in natural and urban field settings. *Journal of Environmental Psychology,* **23,** 109-123.

Haynie, F. H. & Lemmons, T. J. (1990) Particulate matter soiling of exterior paints at a rural site. *Aerosol Science & Technology,* **13,** 356-367.

Heckmann, K. E., Manley, P. N. & Schlesinger, M. D. (2008) Ecological integrity of remnant montane forests along an urban gradient in the Sierra Nevada. *Forest Ecology and Management,* **255,** 2453-2466.

Hector, A., Joshi, J., Scherer-Lorenzen, M., Schmid, B., Spehn, E. M., Wacker, L., Weilenmann, M., Bazeley-White, E., Beierkuhnlein, C. & Caldeira, M. C. (2007) Biodiversity and ecosystem functioning: reconciling the results of experimental and observational studies. *Functional Ecology,* **21,** 998-1002.

Herzog, T. R., Black, A. M., Fountaine, K. A. & Knotts, D. J. (1997) Reflection and attentional recovery as distinctive benefits of restorative environments. *Journal of Environmental Psychology,* **17,** 165-170.

Hey, D. (1998) *A History of Sheffield,* Carnegie, Lancaster.

Hicks, B. B., Baldocchi, D. D., Meyers, T. P., Hosker, R. P. & Matt, D. R. (1987) A preliminary multiple resistance routine for deriving dry deposition velocities from measured quantities. *Water, Air, & Soil Pollution,* **36,** 311-330.

Hogan, D. M. & Walbridge, M. R. (2007) Urbanization and nutrient retention in freshwater riparian wetlands. *Ecological Applications,* **17,** 1142-1155.

Holman-Dodds, J. K., Bradley, A. A. & Potter, K. W. (2003) Evaluation of hydrologic benefits of infiltration based urban storm water management. *Journal of the American Water Resources Association,* **39,** 205-215.

Holman, J. P. (1997) *Heat Transfer. 8th edn.,* McGraw-Hill, New York.

Holtslag, A. A. M. & Van Ulden, A. P. (1983) A simple scheme for daytime estimates of the surface fluxes from routine weather data. *Journal of Applied Meteorology,* **22,** 517-529.

Hope, D., Gries, C., Zhu, W., Fagan, W. F., Redman, C. L., Grimm, N. B., Nelson, A. L., Martin, C. & Kinzig, A. (2003) Socioeconomics drive urban plant diversity. *Proceedings of the National Academy of Sciences of the United States of America,* **100,** 8788-8792.

Hothorn, T., Hornik, K. & Zeileis, A. (2006) Unbiased recursive partitioning: A conditional inference framework. *Journal of Computational and Graphical Statistics,* **15,** 651-674.

Howard, M. J., Brillman, J. C. & Burkle Jr., F. M. (1996) Infectious disease emergencies in disasters. *Emergency Medicine Clinics of North America,* **14,** 413-428.

Ichinose, T., Shimodozono, K. & Hanaki, K. (1999) Impact of anthropogenic heat on urban climate in Tokyo. *Atmospheric Environment,* **33,** 3897-3909.

IPCC (2007). *Climate Change 2007: Synthesis Report. IPCC Fourth Assessment Report (AR4)* (eds R. K. Pachauri & A. Reisinger). Intergovernmental Panel on Climate Change, Geneva, Switzerland.

Irvine, K. N., Fuller, R. A., Devine-Wright, P., Tratalos, J., Payne, S. R., Warren, P. H., Lomas, K. J. & Gaston, K. J. (2010) Chapter 10. Ecological and psychological value of urban green space. *Dimensions of the Sustainable City* (eds M. Jenks & C. Jones). Springer, London.

Jackson, D. L. (2000) Guidance on the interpretation of the Biodiversity Broad Habitat Classification (terrestrial and freshwater types): Definitions and the relationships with other habitat classifications. Technical report, JNCC Report no. 307. Joint Nature Conservation Committee, Peterborough.

Jandl, R., Lindner, M., Vesterdal, L., Bauwens, B., Baritz, R., Hagedorn, F., Johnson, D. W., Minkkinen, K. & Byrne, K. A. (2007) How strongly can forest management influence soil carbon sequestration? *Geoderma,* **137,** 253-268.

Jenerette, G. D., Harlan, S. L., Brazel, A., Jones, N., Larsen, L. & Stefanov, W. L. (2007) Regional relationships between surface temperature, vegetation, and human settlement in a rapidly urbanizing ecosystem. *Landscape Ecology,* **22,** 353-365.

Jenks, M. & Jones, C. (2010) Future City 2: Dimensions of the Sustainable City. Springer, London.

Jiang, L., Wan, S. & Li, L. (2009) Species diversity and productivity: why do results of diversity-manipulation experiments differ from natural patterns? *Journal of Ecology,* **97,** 603-608.

Jim, C. Y. & Chen, W. Y. (2006a) Perception and attitude of residents toward urban green spaces in Guangzhou (China). *Environmental Management,* **38,** 338-349.

Jim, C. Y. & Chen, W. Y. (2006b) Recreation–amenity use and contingent valuation of urban greenspaces in Guangzhou, China. *Landscape and Urban Planning,* **75,** 81-96.

Jim, C. Y. & Chen, W. Y. (2008) Assessing the ecosystem service of air pollutant removal by urban trees in Guangzhou (China). *Journal of Environmental Management,* **88,** 665-676.

Jo, H. K. (2002) Impacts of urban greenspace on offsetting carbon emissions for middle Korea. *Journal of Environmental Management,* **64,** 115-126.

Jo, H. K. & McPherson, G. E. (1995) Carbon storage and flux in urban residential greenspace. *Journal of Environmental Management,* **45,** 109-133.

Johansson, E. & Emmanuel, R. (2006) The influence of urban design on outdoor thermal comfort in the hot, humid city of Colombo, Sri Lanka. *International Journal of Biometeorology,* **51,** 119-133.

Johnson, D. P. & Wilson, J. S. (2009) The socio-spatial dynamics of extreme urban heat events: the case of heat-related deaths in Philadelphia. *Applied Geography,* **29,** 419-434.

Johnson, D. W. & Curtis, P. S. (2001) Effects of forest management on soil C and N storage: meta analysis. *Forest Ecology and Management,* **140,** 227-238.

Johnstone, C. & Markandya, A. (2006) Valuing river characteristics using combined site choice and participation travel cost models. *Journal of Environmental Management,* **80,** 237-247.

Jones, C., Jenks, M. & Bramley, G. (2010a) Chapter 11. Complementarities and contradictions. *Dimensions of the Sustainable City* (eds M. Jenks & C. Jones). Springer, London.

Jones, C., Leishman, C. & MacDonald, C. (2009) Sustainable urban form and residential development viability. *Environment and Planning A,* **41,** 1667-1690.

Jones, C., Leishman, C., MacDonald, C., Orr, A. & Watkins, D. (2010b) Chapter 7. Economic Viability. *Dimensions of the Sustainable City* (eds M. Jenks & C. Jones). Springer, London.

Jorgensen, A., Hitchmough, J. & Calvert, T. (2002) Woodland spaces and edges: their impact on perception of safety and preference. *Landscape and Urban Planning,* **60,** 135-150.

Kaczynski, A. T., Johnson, A. J. & Saelens, B. E. (2009) Neighborhood land use diversity and physical activity in adjacent parks. *Health & Place,* **16,** 413-415.

Kantha, L. H. & Clayson, C. A. (2000) *Small scale processes in geophysical fluid flows,* Academic Press, London.

Kaplan, G. A., Haan, M. N., Syme, S. L., Minkler, M. & Winkleby, M. (1987) Socioeconomic status and health. *American Journal of Preventive Medicine,* **3,** 125-131.

Kareiva, P., Watts, S., McDonald, R. & Boucher, T. (2007) Domesticated nature: shaping landscapes and ecosystems for human welfare. *Science,* **316,** 1866-1869.

Kellett, J. E. (1982) The private garden in England and Wales. *Landscape Planning,* **9,** 105-123.

Kim, J. & Kaplan, R. (2004) Physical and psychological factors in sense of community: new urbanist Kentlands and nearby Orchard Village. *Environment and Behavior,* **36,** 313-340.

Kinzig, A. P., Warren, P., Martin, C., Hope, D. & Katti, M. (2005) The effects of human socioeconomic status and cultural characteristics on urban patterns of biodiversity. *Ecology and Society,* **10,** 23.

Kline, J. D. (2006) Public demand for preserving local open space. *Society & Natural Resources,* **19,** 645-659.

Korpela, K. & Hartig, T. (1996) Restorative qualities of favorite places. *Journal of Environmental Psychology,* **16,** 221-233.

Korpela, K. M. (1989) Place-identity as a product of environmental self-regulation. *Journal of Environmental Psychology,* **9,** 241-256.

Korpela, K. M. (1992) Adolescents' favourite places and environmental self-regulation. *Journal of Environmental Psychology,* **12,** 249-258.

Korpela, K. M. (2003) Negative mood and adult place preference. *Environment and Behavior,* **35,** 331-346.

Korpela, K. M., Hartig, T., Kaiser, F. G. & Fuhrer, U. (2001) Restorative experience and self-regulation in favorite places. *Environment and Behavior,* **33,** 572.

Korpela, K. M. & Ylén, M. (2007) Perceived health is associated with visiting natural favourite places in the vicinity. *Health and Place,* **13,** 138-151.

Krygier, J. & Wood, D. (2005) *Making maps: a visual guide to map design for GIS,* Guilford Press, New York.

Kuo, F. E. (2001) Coping with poverty: Impacts of environment and attention in the inner city. *Environment and Behavior,* **33,** 5-34.

Kuo, F. E. & Sullivan, W. C. (2001a) Aggression and violence in the inner city: effects of environment via mental fatigue. *Environment and Behavior,* **33,** 543-571.

Kuo, F. E. & Sullivan, W. C. (2001b) Environment and crime in the inner city: Does vegetation reduce crime? *Environment and Behavior,* **33,** 343-367.

Kuo, F. E. & Taylor, F. (2004) A potential natural treatment for attention-deficit/hyperactivity disorder: evidence from a national study. *American Journal of Public Health,* **94,** 1580-1586.

Larsen, J. (2006) Setting the record straight: more than 52,000 Europeans died from heat in summer 2003. July 28, 2006. *Earth Policy Institute. Plan B Updates*. Retrieved from http://www.earth-policy.org/index.php?/plan_b_updates/2006/update56, accessed 28/10/2009.

Lindemann-Matthies, P., Junge, X. & Matthies, D. (2010) The influence of plant diversity on people's perception and aesthetic appreciation of grassland vegetation. *Biological Conservation,* **143,** 195-202.

Lodge, D. M. & Shrader-Frechette, K. (2003) Nonindigenous species: ecological explanation, environmental ethics, and public policy. *Conservation Biology,* **17,** 31-37.

Loreau, M., Naeem, S., Inchausti, P., Bengtsson, J., Grime, J. P., Hector, A., Hooper, D. U., Huston, M. A., Raffaelli, D. & Schmid, B. (2001) Biodiversity and ecosystem functioning: current knowledge and future challenges. *Science,* **294,** 804-808.

Lorenz, K. & Lal, R. (2009) Biogeochemical C and N cycles in urban soils. *Environment International,* **35,** 1-8.

Loss, S. R., Ruiz, M. O. & Brawn, J. D. (2009) Relationships between avian diversity, neighborhood age, income, and environmental characteristics of an urban landscape. *Biological Conservation,* **142,** 2578-2585.

Loukaitou-Sideris, A. (1995) Urban form and social context: cultural differentiation in the uses of urban parks. *Journal of Planning Education and Research,* **14,** 89.

Lovatt, R. (2007) Developments in the Sheffield Population. Available from http://www.sheffield.gov.uk/your-city-council/sheffield-facts-figures/population-information, accessed 10/05/2010. Corporate Policy Unit, Sheffield City Council, Sheffield.

Maas, J., Verheij, R. A., Groenewegen, P. P., De Vries, S. & Spreeuwenberg, P. (2006) Green space, urbanity, and health: how strong is the relation? *Journal of Epidemiology and Community Health,* **60,** 587-592.

Maheswaran, R., Haining, R. P., Brindley, P., Law, J., Pearson, T., Fryers, P. R., Wise, S. & Campbell, M. J. (2005a) Outdoor air pollution and stroke in Sheffield, United Kingdom: a small-area level geographical study. *Stroke,* **36,** 239-243.

Maheswaran, R., Haining, R. P., Brindley, P., Law, J., Pearson, T., Fryers, P. R., Wise, S. & Campbell, M. J. (2005b) Outdoor air pollution, mortality, and hospital admissions from coronary heart disease in Sheffield, UK: a small-area level ecological study. *European Heart Journal,* **26,** 2543-2549.

Market Research Society. (2006) *Occupation Groupings: A Job Dictionary. 6th edition.,* Market Research Society, London.

Market Research Society (no date) 2001 Census Outputs by the MRS Social Grade Approximation. Retreived from http://www.mrs.org.uk/networking/cgg/cggsocialgrade.htm, accessed 09/03/2010.

Maroko, A. R., Maantay, J. A., Sohler, N. L., Grady, K. L. & Arno, P. S. (2009) The complexities of measuring access to parks and physical activity sites in New York City: a quantitative and qualitative approach. *International Journal of Health Geographics,* **8,** 34.

Marshall, J. D., Brauer, M. & Frank, L. D. (2009) Healthy neighborhoods: walkability and air pollution. *Environmental Health Perspectives,* **117,** 1752-1759.

Matthias, A. D., Comrie, A. C. & Musil, S. A. (2006) Ch 23. Atmospheric pollution. *Environmental and Pollution Science, 2nd Edition.* (eds I. L. Pepper, C. P. Gerba & M. L. Brusseau), pp. 377-394. Elsevier Science/Academic Press, San Diego, California.

Matzarakis, A., Mayer, H. & Iziomon, M. G. (1999) Applications of a universal thermal index: physiological equivalent temperature. *International Journal of Biometeorology,* **43,** 76-84.

Mayer, H. (1999) Air pollution in cities. *Atmospheric Environment,* **33,** 4029-4037.

Maynard, R. L. (2001) Particulate air pollution. *The urban atmosphere and its effects. Air Pollution Reviews - Vol. 1* (eds P. Brimblecombe & R. L. Maynard). Imperial College Press, London.

McDonald, A. G., Bealey, W. J., Fowler, D., Dragosits, U., Skiba, U., Smith, R. I., Donovan, R. G., Brett, H. E., Hewitt, C. N. & Nemitz, E. (2007) Quantifying the effect of urban tree planting on concentrations and depositions of PM10 in two UK conurbations. *Atmospheric Environment,* **41,** 8455-8467.

McGarigal, K., Cushman, S. A., Neel, M. C. & Ene, E. (2002) FRAGSTATS: spatial pattern analysis program for categorical maps. Computer software, available from www.umass.edu/landeco/research/fragstats/fragstats.html, accessed 13/11/2009. University of Massachusetts, Amherst.

McKinney, M. L. (2006) Urbanization as a major cause of biotic homogenization. *Biological Conservation,* **127,** 247-260.

McKinney, M. L. (2008) Effects of urbanization on species richness: a review of plants and animals. *Urban Ecosystems,* **11,** 161-176.

Memmott, J., Craze, P. G., Waser, N. M. & Price, M. V. (2007) Global warming and the disruption of plant–pollinator interactions. *Ecology Letters,* **10,** 710-717.

Meteorological Synthesizing Centre - East (no date) Dry deposition. Documentation of the MSCE-HM-Hem (hemispheric modelling heavy metals) transport model. Retrieved from http://msceast.org/hemispheric/hm/description_hem.html, accessed 01/09/2009. Meteorological Synthesizing Centre - East.

Millennium Ecosystem Assessment (2003) Ecosystems and human well-being: a framework for assessment. Island Press, Washington D.C.

Millennium Ecosystem Assessment. (2005a) *Ecosystems and human well-being: biodiversity synthesis,* Island Press, Washington, DC.

Millennium Ecosystem Assessment. (2005b) *Ecosystems and human well-being: synthesis,* Island Press, Washington D.C.

Millennium Ecosystem Assessment (2005c) Global assessment reports volume 1: current state & trends. Island Press, Washington D.C.

Mills, L. S., Soulé, M. E. & Doak, D. F. (1993) The keystone-species concept in ecology and conservation. *BioScience,* **43,** 219-224.

Moors for the Future (2007) Peak District moorland. Moors for the Future research note no 12. Moors for the Future, Edale, Hope Valley, UK.

Moran, M. D. (2003) Arguments for rejecting the sequential Bonferroni in ecological studies. *Oikos,* **100,** 403-405.

Naeem, S. (1998) Species redundancy and ecosystem reliability. *Conservation Biology,* **12,** 39-45.

Nagendra, H. (2002) Opposite trends in response for the Shannon and Simpson indices of landscape diversity. *Applied Geography,* **22,** 175-186.

Naidoo, R., Balmford, A., Costanza, R., Fisher, B., Green, R. E., Lehner, B., Malcolm, T. R. & Ricketts, T. H. (2008) Global mapping of ecosystem services and conservation priorities. *Proceedings of the National Academy of Sciences,* **105,** 9495.

National Readership Survey (no date) Lifestyle data. Retrieved from http://www.nrs.co.uk/lifestyle.html, accessed 09/03/2010.

Natural England (2009) Sites of Special Scientific Interest (SSSIs): England's special wildlife and geological sites. Available from http://naturalengland.etraderstores.com/NaturalEnglandShop/NE54, accessed 03/05/2010. Natural England Publications, Newcastle-upon-Tyne.

Nelson, E., Mendoza, G., Regetz, J., Polasky, S., Tallis, H., Cameron, D. R., Chan, K. M. A., Daily, G. C., Goldstein, J. & Kareiva, P. M. (2009) Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. *Frontiers in Ecology and the Environment,* **7,** 4-11.

Newton, P. (2000) Urban form and environmental performance. *Achieving Sustainable Urban Form* (eds K. Williams, E. Burton & M. Jenks). E&FN Spon, London.

Niemelä, J., Kotza, D. J. & Yli-Pelkonen, V. (2009) Comparative urban ecology: challenges and possibilities. *Ecology of Cities and Towns: A Comparative Approach* (eds M. J. McDonnell, A. K. Hahs & J. H. Breuste). Cambridge University Press, Cambridge.

Norderhaug, A., Ihse, M. & Pedersen, O. (2000) Biotope patterns and abundance of meadow plant species in a Norwegian rural landscape. *Landscape Ecology,* **15,** 201-218.

Nowak, D. J. & Crane, D. E. (2002) Carbon storage and sequestration by urban trees in the USA. *Environmental Pollution,* **116,** 381-389.

Nowak, D. J., Crane, D. E., Stevens, J. C., Hoehn, R. E., Walton, J. T. & Bond, J. (2008) A ground-based method of assessing urban forest structure and ecosystem services. *Arboriculture and Urban Forestry,* **34,** 347-358.

Nowak, D. J., Crane, D. E., Stevens, J. C. & Ibarra, M. (2002a) Brooklyn's urban forest. *USDA Forest Service general technical report NE-290*. US Department of Agriculture, Newtown Square, PA.

Nowak, D. J., McHale, P. J., Ibarra, M., Crane, D., Stevens, J. & Luley, C. (1998) Modeling the effects of urban vegetation on air pollution. *Air Pollution Modeling and its Application,* **XII,** 399–407.

Nowak, D. J., Stevens, J. C., Sisinni, S. M. & Luley, C. J. (2002b) Effects of urban tree management and species selection on atmospheric carbon dioxide. *Journal of Arboriculture,* **28,** 113-122.

Oberndorfer, E., Lundholm, J., Bass, B., Coffman, R. R., Doshi, H., Dunnett, N., Gaffin, S., Koehler, M., Liu, K. K. Y. & Rowe, B. (2007) Green roofs as urban ecosystems: ecological structures, functions, and services. *BioScience,* **57,** 823-833.

Offerle, B., Grimmond, C. S. B. & Fortuniak, K. (2005) Heat storage and anthropogenic heat flux in relation to the energy balance of a central European city centre. *International Journal of Climatology,* **25,** 1405-1419.

Office for National Statistics (2004) Census 2001 OA Boundaries for England and Wales in MID/MIF and Shapefile format. Supplied on computer DVD. Office for National Statistics.

Office for National Statistics (2005) Area classification for output areas - cluster summaries. Retrieved from http://www.statistics.gov.uk/about/

methodology_by_theme/area_classification/oa/cluster_summaries.asp, accessed 10/03/2010.
Office for National Statistics (2007) Census Geography. Retrieved from http://www.statistics.gov.uk/geography/census_geog.asp#oa, accessed 14/12/2009.
Office for National Statistics (no date) About ONS. Retrieved from http://www.ons.gov.uk/about/index.html, accessed 17/03/2010.
Office of the Deputy Prime Minister (2004) Creating better places to live: A guide to the planning system in England. ODPM Publications, Wetherby, West Yorkshire.
Ohl, C. A. & Tapsell, S. (2000) Flooding and human health. *British Medical Journal,* **321,** 1167-1168.
Oke, T. R. (1987) *Boundary layer climates. 2nd edition.,* Routledge.
Ordnance Survey (2008) OS MasterMap topography layer. User guide and technical specification. Technical report, Ordnance Survey. Available from http://www.ordnancesurvey.co.uk/oswebsite/. accessed 25/08/2009.
Ordnance Survey (no date) OS Land-Form PANORAMA DTM, 1:50000 Ordnance Survey.
Organisation for Economic Co-operation and Development (2007) Tax/benefit policies: Overview tables for selected years. Available from http://www.oecd.org/document/3/0,3343,en_2649_34637_39617987_1_1_1_1,00.html#policy, accessed 15/07/2010. OECD.
Ozawa, C. P. & Yeakley, J. A. (2007) Performance of management strategies in the protection of riparian vegetation in three oregon cities. *Journal of Environmental Planning and Management,* **50,** 803-822.
Paine, R. (1989) User's Guide to the CTDM Meteorological Preprocessor (METPRO) Program. US Environmental Protection Agency, Research Triangle Park, NC.
Paradis, E. (2009) Moran's autocorrelation coefficient in comparative methods. CRAN R package 'ape' vignette. Available from http://cran.r-project.org/web/packages/ape/vignettes/MoranI.pdf, accessed 12/01/2010.
Parsons, R., Tassinary, L. G., Ulrich, R. S., Hebl, M. R. & Grossman-Alexander, M. (1998) The view from the road: implications for stress recovery and immunization. *Journal of Environmental Psychology,* **18,** 113-140.
Patton, D. R. (1975) A diversity index for quantifying habitat "edge". *Wildlife Society Bulletin,* **3,** 171-173.
Paul, M. J. & Meyer, J. L. (2001) Streams in the urban landscape. *Annual Review of Ecology and Systematics,* **32,** 333-365.
Paul, M. J., Meyer, J. L. & Couch, C. A. (2006) Leaf breakdown in streams differing in catchment land use. *Freshwater Biology,* **51,** 1684-1695.
Pauleit, S., Ennos, R. & Golding, Y. (2005) Modeling the environmental impacts of urban land use and land cover change - a study in Merseyside, UK. *Landscape and Urban Planning,* **71,** 295-310.
Pavao-Zuckerman, M. A. (2008) The nature of urban soils and their role in ecological restoration in cities. *Restoration Ecology,* **16,** 642-649.
Payne, L., Orsega-Smith, B., Roy, K. & Godbey, G. (2005) Local park use and personal health among older adults: an exploratory study. *Journal of Park and Recreation Administration,* **23,** 1-20.
Peak District National Park Authority (2003) Recreation Strategy for the Peak District National Park. Available from http://resources.peakdistrict.gov.uk/pubs/

recreation/recreationstrategy.pdf, accessed 25/05/2010. Peak District National Park Authority, Bakewell, UK.

Peak District National Park Authority (no date) Biodiversity Action Plan. Available from http://www.peakdistrict.gov.uk/bap.pdf, accessed 25/05/2010. Peak District National Park Authority, Bakewell, UK.

Perneger, T. V. (1998) What's wrong with Bonferroni adjustments. *British Medical Journal,* **316,** 1236.

Peters, J., Baets, B. D., Verhoest, N. E. C., Samson, R., Degroeve, S., Becker, P. D. & Huybrechts, W. (2007) Random forests as a tool for ecohydrological distribution modelling. *Ecological Modelling,* **207,** 304-318.

Pickett, S. T. A. & Cadenasso, M. L. (2002) The ecosystem as a multidimensional concept: meaning, model, and metaphor. *Ecosystems,* **5,** 1-10.

Pickett, S. T. A., Cadenasso, M. L., Grove, J. M., Nilon, C. H., Pouyat, R. V., Zipperer, W. C. & Costanza, R. (2001) Urban ecological systems: linking terrestrial ecological, physical, and socioeconomic components of metropolitan areas. *Annual Review of Ecology and Systematics,* **32,** 127-157.

Pickett, S. T. A., Cadenasso, M. L., McDonnell, M. J. & Burch Jr., W. R. (2009) Frameworks for urban ecosystem studies: gradients, patch dynamics and the human ecosystem in the New York metropolitan area and Baltimore, USA. *Ecology of Cities and Towns: A Comparative Approach* (eds M. J. McDonnell, A. K. Hahs & J. H. Breuste). Cambridge University Press, Cambridge.

Pikora, T., Giles-Corti, B., Bull, F., Jamrozik, K. & Donovan, R. (2003) Developing a framework for assessment of the environmental determinants of walking and cycling. *Social Science & Medicine,* **56,** 1693-1703.

Pino, J., Rodą, F., Ribas, J. & Pons, X. (2000) Landscape structure and bird species richness: implications for conservation in rural areas between natural parks. *Landscape and Urban Planning,* **49,** 35-48.

Ponce, N. A., Hoggatt, K. J., Wilhelm, M. & Ritz, B. (2005) Preterm birth: the interaction of traffic-related air pollution with economic hardship in Los Angeles neighborhoods. *American Journal of Epidemiology,* **162,** 140-148.

Pouyat, R. V., Yesilonis, I. D. & Nowak, D. J. (2006) Carbon storage by urban soils in the United States. *Journal of Environmental Quality,* **35,** 1566-1575.

R Development Core Team (2008) R: a language and environment for statistical computing (http://www.R-project.org). R Foundation for Statistical Computing, Vienna, Austria.

Raffaelli, D., Wiegand, J., Ashmore, M., Smart, J. & White, P. (2007) Assessing and monitoring the provision of services by ecosystems. Work package 3 of England's terrestrial ecosystem services and the rationale for an ecosystem-based approach. DEFRA report NR0107. Department for Environment, Food and Rural Affairs.

Rangel, T. F., Diniz-Filho, J. A. F. & Bini, L. M. (2006) Towards an integrated computational tool for spatial analysis in macroecology and biogeography. *Global Ecology and Biogeography,* **15,** 321-327.

Rees, W. & Wackernagel, M. (2008) Urban ecological footprints: why cities cannot be sustainable - and why they are a key to sustainability. *Urban ecology: an international perspective on the interaction between humans and nature* (ed E. S. John M. Marzluff, Wilfried Endlicher, Marina Alberti, Gordon Bradley, Clare Ryan, Ute Simon and Craig ZumBrunnen). Springer, New York.

Rodríguez, J. P., Beard Jr, T. D., Bennett, E. M., Cumming, G. S., Cork, S., Agard, J., Dobson, A. P. & Peterson, G. D. (2006) Trade-offs across space, time, and ecosystem services. *Ecology and Society,* **11,** 28.

Roed, J. (1987) Dry deposition in rural and in urban areas in Denmark. *Radiation Protection Dosimetry,* **21,** 33-36.

Roetzer, T., Wittenzeller, M., Haeckel, H. & Nekovar, J. (2000) Phenology in central Europe–differences and trends of spring phenophases in urban and rural areas. *International Journal of Biometeorology,* **44,** 60-66.

Rotko, T., Koistinen, K., Haenninen, O. & Jantunen, M. (2000) Sociodemographic descriptors of personal exposure to fine particles (PM2.5) in EXPOLIS Helsinki. *Journal of Exposure Analysis and Environmental Epidemiology,* **10,** 385-393.

Rotko, T., Kousa, A., Alm, S. & Jantunen, M. (2001) Exposures to nitrogen dioxide in EXPOLIS-Helsinki: microenvironment, behavioral and sociodemographic factors. *Journal of Exposure Analysis and Environmental Epidemiology,* **11,** 216-223.

Sandler, R. (2010) The value of species and the ethical foundations of assisted colonization. *Conservation Biology,* **24,** 424-431.

Sanford, M. P., Manley, P. N. & Murphy, D. D. (2009) Effects of urban development on ant communities: implications for ecosystem services and management. *Conservation Biology,* **23,** 131-141.

Sansalone, J. J. (2003) The physical and chemical nature of urban stormwater runoff pollutants. *Wet-weather flow in the urban watershed: Technology and management* (eds R. Field & D. Sullivan). Lewis, Boca Raton, Florida, USA.

Santamouris, M. (2009) *Advances in Building Energy Research, Volume 3,* Earthscan, London.

Satterthwaite, D. (2008) Cities' contribution to global warming: notes on the allocation of greenhouse gas emissions. *Environment and Urbanization,* **20,** 539-549.

Schikowski, T., Sugiri, D., Reimann, V., Pesch, B., Ranft, U. & Krämer, U. (2008) Contribution of smoking and air pollution exposure in urban areas to social differences in respiratory health. *BMC Public Health,* **8,** 179.

Schlesinger, M. D., Manley, P. N. & Holyoak, M. (2008) Distinguishing stressors acting on land bird communities in an urbanizing environment. *Ecology,* **89,** 2302-2314.

Schmid, B. (2002) The species richness-productivity controversy. *Trends in Ecology & Evolution,* **17,** 113-114.

Scottish Executive Development Department (2008) Planning and open space. Planning advice note 65. Scottish Executive Development Department, Edinburgh.

Shandas, V., Graybill, J. K. & Ryan, C. M. (2008) Incorporating ecosystem-based management into urban environmental policy: a case study from western Washington. *Journal of Environmental Planning and Management,* **51,** 647-662.

Sheffield City Council (2003) 2001 census topic reports. Economic activity. Retrieved from http://www.sheffield.gov.uk/your-city-council/sheffield-facts-figures/sheffield-economy/employment--jobs/economic-activity, accessed 10/05/2010. Corporate Policy Unit, Sheffield City Council, Sheffield.

Sheffield City Council (2007) Employment by Business Sector. Excel spreadsheet document, retrieved from http://www.sheffield.gov.uk/your-city-council/sheffield-facts-figures/sheffield-economy/businesses, accessed 11/05/2010.

Sheffield City Council (2008a) Air Quality Progress Report August 2008. Sheffield City Council, Sheffield.

Sheffield City Council (2008b) Detailed Asessment for PM10. Sheffield City Council, Sheffield.

Sheffield City Council (2009a) Sheffield Development Framework. Core Strategy. Sheffield City Council, Sheffield, UK.

Sheffield City Council (2009b) State of Sheffield 2009. Retrieved from http://www.sheffield.gov.uk/your-city-council/sheffield-facts-figures/state-of-sheffield, accessed 27/04/2010. Sheffield City Council, Sheffield.

Sheffield City Council (no date) Sheffield's Great Outdoors: Green & Open Space Strategy 2010-2030. Sheffield City Council, Sheffield.

Shuster, W. D., Gehring, R. & Gerken, J. (2007) Prospects for enhanced groundwater recharge via infltration of urban storm water runoff: a case study. *Journal of Soil and Water Conservation,* **62,** 129-137.

Silva, J., Ribiero, C. & Guedes, R. (2007) Roughness length classification of CORINE land cover classes. *European Wind Energy Conference*.

Simmonds, D. & Coombe, D. (2000) The transport implications of alternative urban forms. *Achieving Sustainable Urban Form* (eds K. Williams, E. Burton & M. Jenks). E&FN Spon, London.

Singer, G. A. & Battin, T. J. (2007) Anthropogenic subsidies alter stream consumer-resource stoichiometry, biodiversity, and food chains. *Ecological Applications,* **17,** 376-389.

Slabbekoorn, H. & Ripmeester, E. A. P. (2008) Birdsong and anthropogenic noise: implications and applications for conservation. *Molecular Ecology,* **17,** 72-83.

Smith, J. P. (1998) Socioeconomic status and health. *American Economic Review,* **88,** 192-196.

Smith, R. M., Thompson, K., Hodgson, J. G., Warren, P. H. & Gaston, K. J. (2006) Urban domestic gardens (IX): composition and richness of the vascular plant flora, and implications for native biodiversity. *Biological Conservation,* **129,** 312-322.

Soil Survey Division Staff. (1993) *Soil survey manual. Handbook 18,* Soil Conservation Service, U.S. Department of Agriculture.

Solorzano, M., Stingone, J., Rentas, A. R., Claudio, L., Aral, M. M., Brebbia, C. A., Sinks, T. & Maslia, M. L. (2005) Comparison of fine particulate matter and ozone levels in zip code areas of New York City with different child asthma burdens. *Environmental Exposure and Health,* **85,** 23-32.

South Yorkshire Archaeology Service (2005) South Yorkshire Environment Characterisation. Revised project design. South Yorkshire Archaeology Service, Sheffield.

St Leger, L. (2003) Health and nature - new challenges for health promotion. *Health Promotion International,* **18,** 173-175.

Stokols, D., Grzywacz, J. G., McMahan, S. & Phillips, K. (2003) Increasing the health promotive capacity of human environments. *American Journal of Health Promotion,* **18,** 4-13.

Stone, B. & Bullen, J. L. (2006) Urban form and watershed management: how zoning influences residential stormwater volumes. *Environment and Planning B: Planning and Design,* **33,** 21-37.

Stone Jr, B. (2008) Urban sprawl and air quality in large US cities. *Journal of Environmental Management,* **86,** 688-698.

Strobl, C. (2009) E-mail to M. Mears, 23/02/2009.

Strobl, C., Boulesteix, A. L., Kneib, T., Augustin, T. & Zeileis, A. (2008) Conditional variable importance for random forests. *BMC Bioinformatics,* **9,** 307.
Strobl, C., Boulesteix, A. L., Zeileis, A. & Hothorn, T. (2007) Bias in random forest variable importance measures: illustrations, sources and a solution. *BMC Bioinformatics,* **8,** 25.
Strohbach, M. W., Haase, D. & Kabisch, N. (2009) Birds and the city: urban biodiversity, land use, and socioeconomics. *Ecology and Society,* **14,** 31.
Swanwick, C., Dunnett, N. & Woolley, H. (2003) Nature, role and value of green space in towns and cities: an overview. *Built Environment,* **29,** 94-106.
Takahashi, T., Amano, Y., Kuchimura, K. & Kobayashi, T. (2008) Carbon content of soil in urban parks in Tokyo, Japan. *Landscape and Ecological Engineering,* **4,** 139-142.
Takano, T., Nakamura, K. & Watanabe, M. (2002) Urban residential environments and senior citizens' longevity in megacity areas: the importance of walkable green spaces. *Journal of Epidemiology and Community Health,* **56,** 913-918.
Tang, U. W. & Wang, Z. S. (2007) Influences of urban forms on traffic-induced noise and air pollution: results from a modelling system. *Environmental Modelling & Software,* **22,** 1750-1764.
Taylor, A. F., Kuo, F. E. & Sullivan, W. C. (2001) Coping with ADD: the surprising connection to green play settings. *Environment and Behavior,* **33,** 54-77.
Taylor, A. F., Kuo, F. E. & Sullivan, W. C. (2002) Views of nature and self-discipline: evidence from inner city children. *Journal of environmental psychology,* **22,** 49-63.
Thompson, K., Askew, A. P., Grime, J. P., Dunnett, N. P. & Willis, A. J. (2005) Biodiversity, ecosystem function and plant traits in mature and immature plant communities. *Functional Ecology,* **19,** 355-358.
Thuiller, W., Lavorel, S., Sykes, M. T. & Araújo, M. B. (2006) Using niche-based modelling to assess the impact of climate change on tree functional diversity in Europe. *Diversity and Distributions,* **12,** 49-60.
Tong, C., Feagin, R. A., Lu, J., Zhang, X., Zhu, X., Wang, W. & He, W. (2007) Ecosystem service values and restoration in the urban Sanyang wetland of Wenzhou, China. *Ecological Engineering,* **29,** 249-258.
Tratalos, J., Fuller, R. A., Warren, P. H., Davies, R. G. & Gaston, K. J. (2007) Urban form, biodiversity potential and ecosystem services. *Landscape and Urban Planning,* **83,** 308-317.
Tsihrintzis, V. A. & Hamid, R. (1997) Modeling and management of urban stormwater runoff quality: a review. *Water Resources Management,* **11,** 136-164.
Tso, C. P., Chan, B. K. & Hashim, M. A. (1991) Analytical solutions to the near-neutral atmospheric surface energy balance with and without heat storage for urban climatological studies. *Journal of Applied Meteorology,* **30,** 413-424.
Tunnicliffe, W. S. & Ayres, J. G. (2001) Trends in air pollution related disease. *The urban atmosphere and its effects. Air pollution reviews - Vol. 1* (eds P. Brimblecombe & R. L. Maynard). Imperial College Press, London.
Tzoulas, K., Korpela, K., Venn, S., Yli-Pelkonen, V., Ka mierczak, A., Niemela, J. & James, P. (2007) Promoting ecosystem and human health in urban areas using green infrastructure: a literature review. *Landscape and Urban Planning,* **81,** 167-178.
UK Biodiversity Group (1999) Habitat action plan: blanket bog. Retrieved from http://www.ukbap.org.uk/ukplans.aspx?id=21, accessed 13/05/2010. Originally published in: UK Biodiversity Group tranche 2 action plans - volume VI:

terrestrial and freshwater species and habitats (October 1999, Tranche 2, Vol VI, p205).

UK Meteorological Office (2006) MIDAS land surface stations data (1853-current). Retrieved from http://badc.nerc.ac.uk/data/ukmo-midas, accessed 01/09/2009. British Atmospheric Data Centre.

UKCIP (2002) Warming up the region: Yorkshire and Humber climate change impact scoping study. Available from http://www.ukcip.org.uk/index.php?option=com_content&task=view&id=196&Itemid=318, accessed 27/04/2010.

Ulrich, R. S. (1984) View through a window may influence recovery from surgery. *Science,* **224,** 420-421.

Ulrich, R. S., Simons, R. F., Losito, B. D., Fiorito, E., Miles, M. A. & Zelson, M. (1991) Stress recovery during exposure to natural and urban environments. *Journal of Environmental Psychology,* **11,** 201-230.

United Nations Population Fund (2007) State of the world population 2007. Unleashing the potential of urban growth. United Nations Population Fund, New York.

USDA-NRCS (1986) Urban hydrology for small watersheds. TR55. United States Department of Agriculture Natural Resources Conservation Service - Conservation Engineering Division.

USDA-NRCS (2005) Soil map of the world: global suborders. GIS dataset. USDA-NRCS, Soil Survey Division, World Soil Resources, Washington D.C.

VanWoert, N. D., Rowe, D. B., Andresen, J. A., Rugh, C. L., Fernandez, R. T. & Xiao, L. (2005) Green roof stormwater retention: effects of roof surface, slope, and media depth. *Journal of Environmental Quality,* **34,** 1036-1044.

Verburg, P. H., van de Steeg, J., Veldkamp, A. & Willemen, L. (2009) From land cover change to land function dynamics: a major challenge to improve land characterization. *Journal of Environmental Management,* **90,** 1327-1335.

Vickers, D. & Rees, P. (2007) Creating the UK National Statistics 2001 output area classification. *Journal of the Royal Statistical Society series A,* **170,** 379-403.

Villarreal, E. L. & Bengtsson, L. (2005) Response of a Sedum green-roof to individual rain events. *Ecological Engineering,* **25,** 1-7.

Visser, M. E., Van Noordwijk, A. J., Tinbergen, J. M. & Lessells, C. M. (1998) Warmer springs lead to mistimed reproduction in great tits (Parus major). *Proceedings of the Royal Society B: Biological Sciences,* **265,** 1867-1870.

Volchenkov, D. & Blanchard, P. (2007) Ghetto of Venice: access to the target node and the random target access time. *Arxiv preprint arXiv:0710.3021.*

Walker, B. H. (1992) Biodiversity and ecological redundancy. *Conservation Biology,* **6,** 18-23.

Wallace, K. J. (2007) Classification of ecosystem services: problems and solutions. *Biological Conservation,* **139,** 235-246.

Walsh, C. J., Roy, A. H., Feminella, J. W., Cottingham, P. D., Groffman, P. M. & Morgan Ii, R. P. (2005) The urban stream syndrome: current knowledge and the search for a cure. *Journal of the North American Benthological Society,* **24,** 706-723.

Walworth, J. & Pepper, I. L. (2006) Physical contaminants. *Environmental and Pollution Science, 2nd Edition.* (eds I. L. Pepper, C. P. Gerba & M. L. Brusseau), pp. 121-131. Elsevier Science/Academic Press, San Diego, California.

Ward, A., Trimble, S. & Wolman, M. (2003) *Environmental Hydrology,* Lewis Publishers, Boca Raton, Florida, USA.

WebMET (1999) Met monitoring guide. Retrieved from http://www.webmet.com/met_monitoring/toc.html, accessed 13/07/2009.

Weiss, A. & Norman, J. M. (1985) Partitioning solar radiation into direct and diffuse, visible and near-infrared components. *Agricultural and Forest meteorology,* **34,** 205-213.

Wells, N. M. (2000) At home with nature: effects of "greenness" on children's cognitive functioning. *Environment and Behavior,* **32,** 775-795.

Wheater, H. S. (2006) Flood hazard and management: a UK perspective. *Philosophical Transactions A,* **364,** 2135-2145.

White, M. A., Nemani, R. R., Thornton, P. E. & Running, S. W. (2002) Satellite evidence of phenological differences between urbanized and rural areas of the eastern United States deciduous broadleaf forest. *Ecosystems,* **5,** 260-273.

Whitford, V., Ennos, A. R. & Handley, J. F. (2001) "City form and natural process" - indicators for the ecological performance of urban areas and their application to Merseyside, UK. *Landscape and Urban Planning,* **57,** 91-103.

Wilby, R. L. & Perry, G. L. W. (2006) Climate change, biodiversity and the urban environment: a critical review based on London, UK. *Progress in Physical Geography,* **30,** 73-98.

Williams, K., Burton, E. & Jenks, M. (2000a) Achieving sustainable urban form. E & FN Spon, London.

Williams, K., Burton, E. & Jenks, M. (2000b) Achieving sustainable urban form: an introduction. *Achieving sustainable urban form* (eds K. Williams, E. Burton & M. Jenks). E & EF Spon, London.

Williams, K., Burton, E. & Jenks, M. (2000c) Part 2 Urban Form and Transport: New Dimensions. Introduction. *Achieving sustainable urban form* (eds K. Williams, E. Burton & M. Jenks). E & EF Spon, London.

Williams, K., Dair, C. & Londsay, M. (2010) Chapter 9. Neighbourhood design and sustainable lifestyles. *Dimensions of the Sustainable City. Future City 2* (eds M. Jenks & C. Jones). Springer, London.

Wong, N. H., Chen, Y., Ong, C. L. & Sia, A. (2003) Investigation of thermal benefits of rooftop garden in the tropical environment. *Building and Environment,* **38,** 261-270.

Wood, L., Frank, L. D. & Giles-Corti, B. (2010) Sense of community and its relationship with walking and neighborhood design. *Social Science & Medicine,* **70,** 1381-1390.

World Health Organization (2006) Air quality guidelines for particulate matter, ozone, nitrogen dioxide and sulfur dioxide. Global update 2005. Summary of risk assessment. World Health Organization.

Worrall, F., Burt, T., Adamson, J., Reed, M., Warburton, J., Armstrong, A. & Evans, M. (2007) Predicting the future carbon budget of an upland peat catchment. *Climatic Change,* **85,** 139-158.

Yang, J., Yu, Q. & Gong, P. (2008) Quantifying air pollution removal by green roofs in Chicago. *Atmospheric Environment,* **42,** 7266-7273.

Yli-Pelkonen, V. (2006) Use of ecological information in urban planning. PhD thesis. *Faculty of Biosciences, Department of Biological and Environmental Sciences*. University of Helsinki.

Zannetti, P. (1990) *Air pollution modelling. Theories, computational methods and available software,* Computational Mechanics Publications, Southampton, UK.

Zhang, L., Brook, J. R. & Vet, R. (2003) A revised parameterization for gaseous dry deposition in air-quality models. *Atmospheric Chemistry and Physics,* **3,** 2067–2082.

Zhang, L., Gong, S., Padro, J. & Barrie, L. (2001) A size-segregated particle dry deposition scheme for an atmospheric aerosol module. *Atmospheric Environment,* **35,** 549-560.

Zhang, L., Moran, M. D., Makar, P. A., Brook, J. R. & Gong, S. (2002) Modelling gaseous dry deposition in AURAMS: a unified regional air-quality modelling system. *Atmospheric Environment,* **36,** 537-560.

Appendix A. Ecosystem modelling technical details

A.1 Reduction of air pollution model

Air pollution reduction is modelled using published pollutant deposition models, namely the Meteorological Service of Canada's AURAMS (A Unified Regional Air-quality Modelling System), with some equations replaced by the equivalent equations from the US EPA's AERMOD where the AURAMS specification could not be implemented due to non-availability of input data or poor model reporting. AURAMS is specified in publications by Zhang et al. (2001, 2002, 2003), and AERMOD by the Environmental Protection Agency (2004). In essence, these models use pollutant concentration data (the output from pollution dispersion models), land cover specific parameters and meteorological data to calculate the rate at which pollutants are deposited from the atmosphere to the surface, in dry weather conditions.

A.1.1 Model formulation

This section will describe the mathematical formulation of the air pollution removal model. Symbols are described as they are encountered, and summarised in Table 0.1.

The model proceeds by calculation of the pollutant flux to surfaces (F; in units of g m^{-2} s^{-1}) as the product of the deposition velocity (the effective velocity at which particles are deposited; V_d; m s^{-1}) and the ambient pollutant concentration (C; g m^{-3}) (Zannetti 1990, Nowak et al. 1998):

$$F = V_d\, C \tag{7}$$

Values of C are derived from output from Sheffield City Council's AIRVIRO model, which is described in Section 2.2.4. The method for determining V_d depends on whether the pollutant is gaseous or particulate. The formulations for NO_2 (gaseous) and PM_{10} (particulate) are described in Sections A.1.1.2 and A.1.1.3 respectively. First, some meteorological pre-processing is required to determine parameters used in V_d calculation.

Table 0.1. Symbols used in air pollution removal model formulation. Data source details the sources of input data, or equations where values are calculated within the model; Met 1 and Met 2 indicate the source as the Sheffield City Council meteorological records and the Derbyshire Lane weather station respectively.

Symbol	Meaning	Units	Data source
A	Characteristic radius of particle collectors	m	Zhang et al. (2001)
a	Empirical constant used in Eqn. (8)	none	Zannetti (1990)
b	Empirical constant used in Eqn. (8)	none	Zannetti (1990)
b_{rs}	Empirical light response constant used in Eqns. (28) and (29)	W m^{-2}	Zhang et al. (2002)
b_{vpd}	Empirical constant used in Eqn. (41)	kPa^{-1}	Zhang et al. (2002)
C	Ambient pollutant concentration	µg m^{-3}	AIRVIRO
D	Brownian diffusivity of particles	m^2 s^{-1}	Eqn. (59)
D_{H20}	Molecular diffusivity of water vapour in air, taken as 2.2×10^{-5}	m^2 s^{-1}	Hicks et al. (1987)
D_{NO2}	Molecular diffusivity of NO_2 in air, taken as 1.4×10^{-5}	m^2 s^{-1}	Hicks et al. (1987)
d	Day of year	days	n/a
d_p, d_{p0}	Particle diameter, after and before hygroscopic growth respectively	m	Simulated size distribution, Eqn. (51)
E_B, E_{IM}, E_{IN}	Particle collection efficiency by Brownian diffusion, impaction and interception respectively	none	Eqns. (56), (57), (58)
F	Pollutant flux to surfaces	µg m^{-2} s^{-1}	Eqn. (7)
f_D f_T f_ψ	Fractional reduction of G_{st} due to water vapour pressure deficit, air temperature and water stress respectively	none	Eqns. (41), (40), (44)
f_p	Proportion of particulate matter in size band	none	Simulated size distribution
g	Acceleration due to gravity, taken as 9.80616	m s^{-2}	Environmental Protection Agency (2004)
G_{st}	Unstressed canopy stomatal conductance	m s^{-1}	Eqn. (20)
h	Hour angle	radians	Eqn. (25)
L	Monin-Obukhov length	none	Eqn. (8)
LAI	Total leaf area index	m^2 m^{-2}	Zhang et al. (2003)
LAI_{shade}, LAI_{sun}	Leaf area index of shaded and sunlit leaves respectively	m^2 m^{-2}	Eqns. (22), (21)
m	Optical air mass	none	Eqn. (38)
P	Air pressure	kPa	Met 2
p	Density of particulate matter	g m^{-3}	Zhang et al. (2001)
PAR_{diff}, PAR_{dir}	Downward visible radiation fluxes above the canopy from diffuse and direct beam radiation respectively	W m^{-2}	Eqns. (37), (36)

Table 12.2 continued.

Symbol	Meaning	Units	Data source
PAR_{shade}, PAR_{sun}	Photosynthetically active radiation received by shaded and sunlit leaves respectively	W m^{-2}	Eqns. (30), (31)
PR_{DN}, PR_{dN}, PR_{DV}, PR_{dV}	Potential near infrared direct beam, near infrared diffuse, visible direct beam and visible diffuse radiation	W m^{-2}	Eqns. (34), (35), (32), (33)
R_1	Fraction of particles that do not rebound	none	Eqn. (60)
R_a	Aerodynamic resistance	s m^{-1}	Eqn. (14)
R_{ac}	In-canopy aerodynamic resistance	s m^{-1}	Eqn. (45)
R_{ac0}	Reference value of R_{ac}	s m^{-1}	Zhang et al. (2003), Eqn. (46)
R_b	Quasi-laminar sublayer resistance	s m^{-1}	Eqn. (15)
R_c	Canopy resistance	s m^{-1}	Eqn. (17)
R_{cut}	Cuticle resistance	s m^{-1}	Eqn. (48)
R_{cutd0}, R_{cutw0}	Reference values of R_{cut} for dry and wet conditions respectively	s m^{-1}	Zhang et al. (2003)
R_g	Ground resistance	s m^{-1}	Zhang et al. (2002, 2003)
R_m	Mesophyll resistance	s m^{-1}	Zhang et al. (2002)
R_s	Surface resistance	s m^{-1}	Eqn. (55)
R_{st}	Stomatal resistance	s m^{-1}	Eqn. (19)
r_{stmin}	Minimum possible leaf stomatal resistance	s m^{-1}	Zhang et al. (2003)
$r_{stshade}$, r_{stsun}	Unstressed stomatal leaf resistance for shaded and sunlit leaves respectively	s m^{-1}	Eqns. (29), (28)
RH	Relative humidity as a fraction	none	Met 2
S_{CF}	Slip correction factor	none	Eqn. (53)
SL	Solar longitude	radians	Eqn. (27)
SR	Solar radiation	W m^{-2}	Met 1+2
St	Stokes number	none	Eqn. (61)
T	Air temperature	K	Met 1+2
t	Universal time	hours	n/a
T_C	Air temperature	C	Met 1+2
T_{max}, T_{min}	Maximum and minimum air temperature outside which stomata are completely closed	K	Zhang et al. (2003)
T_{opt}	Air temperature at which maximal stomatal opening occurs	K	Zhang et al. (2003)
U	Wind speed at height z	m s^{-1}	Met 1+2
u_*	Friction velocity	m s^{-1}	Eqn. (9)
v, v^*	Ambient and saturation water vapour pressure respectively	kPa	Eqns. (43), (42)
V_d	Deposition velocity	m s^{-1}	Eqns. (13), (50)

Table 12.2 continued.

Symbol	Meaning	Units	Data source
V_g	Gravitational settling velocity	m s^{-1}	Eqn. (52)
w	Water absorption in the near infrared	none	Eqn. (39)
W_{st}	Fraction of stomatal closure due to canopy wetness	none	Eqn. (18)
x	Parameter used in Eqn. (10)	none	Eqn. (11)
z	Height of wind speed measurements	m	n/a
z_0	Roughness length	m	Zhang et al. (2003), Eqn. (12)
α	Parameter used in Eqn. (57)	none	Zhang et al. (2001)
β	Scaling parameter for converting values determined for O_3 to values for NO_2	none	Zhang et al. (2002)
γ	Parameter used in Eqn. (56)	none	Zhang et al. (2001)
δ	Solar declination	radians	Eqn. (26)
η	Absolute viscosity of air	g m^{-1} s^{-1}	Eqn. (54)
ϑ	Solar zenith angle	radians	Eqn. (23)
κ	Von Kármán constant, taken to be 0.41	none	Kantha and Clayson (2000)
λ	Mean free path of air molecules	m	Environmental Protection Agency (2004)
ν	Kinematic viscosity of air	m^2 s^{-1}	Eqn. (16)
φ	Solar elevation angle	radians	Eqn. (24)
χ	Angle between the leaf and the sun	radians	Zhang et al. (2002)
Ψ_{c1}, Ψ_{c2}	Leaf water dependency parameters used in Eqn. (44)	MPa	Zhang et al. (2003)
ψ_m	Stability function in Eqn. (9)	none	Eqn. (10)

A.1.1.1 Meteorological pre-processing

Two derived meteorological parameters are important to deposition processes: friction velocity (u_*; m s^{-1}), which is a reference wind velocity describing the relationship between the surface stress (or the downward flux of momentum along the main wind direction) and the air density; and the Monin-Obukhov stability length (L; dimensionless), which "characterises the 'stability' of the surface layer" (Zannetti 1990). Given data on cloud cover, incoming solar radiation, air temperature and wind speed, u_* and L can be estimated according to procedures described by Paine (1989). However, the only available cloud cover data were modelled long term average monthly estimates, and estimates made using these data were outside the limits of expected values. Therefore an alternative approach was used.

Empirical constants have been developed that relate L to a simple classification of meteorological stability conditions (i.e. the temperature profile from the surface to the top of the planetary boundary layer) by equations of the form (Zannetti 1990):

$$L = \frac{1}{a\, z_0^b} \tag{8}$$

Where z_0 is the roughness length (m; see Eqn. (12)), or the height above the ground below which turbulence is not fully developed. z_0 is related to the height of the structures interfering with turbulent processes at the land surface and consequently depends on land cover and seasonal changes (Zannetti 1990, Zhang et al. 2002). a and b are empirical constants that vary with the Pasquill stability class, a simple classification of atmospheric stability conditions (Zannetti 1990), details of which are given in Table 0.2. When conditions are neutral, $L = \infty$, while stable conditions have $\infty > L > 0$ and unstable conditions have $L < 0$. Neutral conditions are characterised by an adiabatic (decreasing with height) vertical temperature profile, and are typical of daytime–nighttime transitions, overcast conditions or strong winds. Stable conditions are characterised by a positive heat flux at the ground and are typical of daytime; and unstable conditions by a ground-based temperature inversion, typical of clear nights with weak winds (Zannetti 1990).

Daytime Pasquill stability classes were determined using the solar radiation/delta-T method (WebMET 1999), with wind speed and solar radiation as input. Nighttime stability requires cloud cover data for accurate classification. Suitable cloud cover data was not available, so nighttime stability was classified according to the Turner method assuming constant cloud cover > 4/10, which the modelled long term average monthly estimates suggested to be the average case for all months. This assumption caused a possible difference of a maximum of one stability class, the potential effects of which were investigated during sensitivity testing (Section A.1.4.3). Both classification methods are recommended by the US EPA (WebMET 1999). Table 0.3 shows the classification criteria as modified from WebMET (1999).

u_* was calculated from L using the following equation (Zannetti 1990):

$$u_* = \frac{\kappa\, U}{\ln\frac{z}{z_0} - \psi_m} \tag{9}$$

Where κ is the Von Kármán constant (dimensionless), which current estimates take to be 0.41 (Kantha and Clayson 2000), U is the wind speed (m s^{-1}) at height z (m) and

Table 0.2. Pasquill stability classes, their description, and values of coefficients used in estimation of the Monin-Obukhov stability length.

Class	Description	a	b
A	Very stable	-0.08750	-0.1029
B	Unstable	-0.03849	-0.1714
C	Slightly unstable	-0.00807	-0.3049
D	Neutral	0	0
E	Slightly stable	0.00807	-0.3049
F/G	Stable	0.03849	-0.1714

Table 0.3. Criteria used for classification of Pasquill stability classes (see Table 0.2).

a. Daytime criteria

Wind speed ($m\ s^{-1}$)	Solar radiation ($W\ m^{-2}$)			
	≥925	**675-925**	**175-675**	**<175**
<2	A	A	B	D
2-3	A	B	C	D
3-5	B	B	C	D
5-6	C	C	D	D
≥6	C	D	D	D

b. Nighttime criteria

Wind speed ($m\ s^{-1}$)	Category
<3.4	F/G
3.4-5.5	E
≥5.5	D

ψ_m is a stability function, the form of which depends on stability conditions (Zannetti 1990):

$$\psi_m = \begin{cases} 0, & \text{if neutral } (L = \infty) \\ \dfrac{-5z}{L}, & \text{if stable } (L > 0) \\ 2\ \ln\left(\dfrac{1+x}{2}\right) + \ln\left(\dfrac{1+x^2}{2}\right) - \tan^{-1} x + \dfrac{\pi}{2}, & \text{if unstable } (L < 0) \end{cases} \tag{10}$$

Where, for the case $L < 0$, x is:

$$x = \left(1 - \frac{16z}{L}\right)^{0.25} \tag{11}$$

As these equations show, u_* is zero when there is no wind, and increases as wind speed increases; and as well as being dependent upon stability conditions, it is also influenced by the nature of the land cover.

z_0 is a seasonally varying parameter in these equations, and is determined for any time period (t) as a function of the seasonal (t), annual minimum (min) and annual maximum (max) leaf area index (Zhang et al. 2003):

$$z_0(t) = z_0(min) + \left(z_0(max) - z_0(min)\right)\left(\frac{LAI(t) - LAI(min)}{LAI(max) - LAI(min)}\right) \tag{12}$$

A.1.1.2 NO₂ deposition velocity

The deposition velocity of gaseous pollutants such as NO_2 is a function of three separate resistances. The aerodynamic resistance (R_a; s m^{-1}) is the resistance through the surface layer, which is characterised by atmospheric turbulence properties and typically extends to a height of 20m (Zannetti 1990). The quasilaminar sublayer resistance (R_b; s m^{-1}) is the resistance through the deposition layer, which is the thin layer above the deposition surface where molecular diffusion processes dominate, and turbulence is intermittent (Zannetti 1990). Finally, the canopy resistance (R_c; s m^{-1}) describes processes at the canopy or vegetation surface that remove the pollutant from the air, such as uptake into the plant or chemical reaction on plant or ground surfaces (Hicks et al. 1987, Zannetti 1990). Each of these resistances is calculated separately, and V_d is then the inverse sum of these components:

$$V_d = \frac{1}{R_a + R_b + R_c} \tag{13}$$

R_a is calculated differently for neutral and stable conditions compared to unstable conditions (Environmental Protection Agency 2004):

$$R_a = \begin{cases} \dfrac{1}{\kappa\, u_*}\left[\ln\left(\dfrac{z}{z_0}\right) + \dfrac{5z}{L}\right], & \text{if neutral or stable} \\ \dfrac{1}{\kappa\, u_*}\left[\ln\dfrac{\left(\sqrt{1-\dfrac{16z}{L}}-1\right)\left(\sqrt{1-\dfrac{16z_0}{L}}+1\right)}{\left(\sqrt{1-\dfrac{16z}{L}}+1\right)\left(\sqrt{1-\dfrac{16z_0}{L}}-1\right)}\right], & \text{if unstable} \end{cases} \tag{14}$$

The quasilaminar sublayer resistance is calculated as follows (Environmental Protection Agency 2004):

$$R_b = \frac{2.2}{\kappa\, u_*}\left(\frac{\upsilon}{D_{NO_2}}\right)^{\frac{2}{3}} \tag{15}$$

Where υ is the kinematic viscosity of air ($m^2\ s^{-1}$) calculated by Eqn. (16), and D_{NO_2} is the molecular diffusivity of NO_2 in air, taken as $1.4 \times 10^{-5}\ m^2\ s^{-1}$ (Hicks et al. 1987).

$$\upsilon = 0.1505 \times 10^{-4} \left(\frac{T}{273.15}\right)^{1.772} \left(\frac{P}{101.325}\right) [1 + 0.0132\,(P - 101.325)] \tag{16}$$

Where T is the air temperature in Kelvin (K), 273.15K is the reference temperature (taken to be the freezing point of water), P is the air pressure (kPa) and 101.325kPa is the reference pressure (taken to be average sea-level pressure).

Parameterisation of R_c involves computation of multiple resistance components: stomatal (R_{st}), mesophyll (R_m), in-canopy aerodynamic (R_{ac}), ground (R_g) and cuticle (R_{cut}), all measured in $m\ s^{-1}$. R_{st}, R_m, R_{ac} and R_{cut} are not applicable where there is no vegetation (Zhang et al. 2003). The formulation is as follows (Zhang et al. 2003):

$$R_c = \left(\frac{1 - W_{st}}{R_{st} + R_m} + \frac{1}{R_{ac} + R_g} + \frac{1}{R_{cut}}\right)^{-1} \tag{17}$$

Where W_{st} is the fraction of stomatal closure when the canopy is wet. In the published model formulation, the canopy is treated as wet when rain or dew occurs. However, due to the unavailability of cloud cover data for the present study area there is no way to determine when dew is occurring. Therefore the canopy is treated as dry except when at least 0.1mm rainfall has occurred during a given hour or the hour preceeding it (following Environmental Protection Agency (2004)). W_{st} depends on whether the canopy is wet or dry, and the amount of solar radiation (SR, in $W\ m^{-2}$), such that greater SR results in greater stomatal closure (Zhang et al. 2003):

$$W_{st} = \begin{cases} 0, & \text{if dry} \\ 0, & \text{if wet and } SR \leq 200 \\ \dfrac{SR - 200}{800}, & \text{if wet and } 200 < SR \leq 600 \\ 0.5, & \text{if wet and } SR > 600 \end{cases} \tag{18}$$

Stomatal resistance is calculated as a "sunlit/shaded big leaf" or "two big leaf" model (Zhang et al. 2002):

$$R_{st} = \left(G_{st}\, f_T\, f_D\, f_\Psi\, \frac{D_{NO_2}}{D_{H_2O}}\right)^{-1} \tag{19}$$

G_{st} is the unstressed canopy stomatal conductance ($m\ s^{-1}$); f_T, f_D and f_Ψ are functions describing the conductance-reducing effects of air temperature, water vapour pressure deficit and water stress respectively (dimensionless); and D_{H_2O} is the molecular diffusivity of water vapour in air, taken as $2.2 \times 10^{-5}\ m^2\ s^{-1}$.

G_{st} is a function of PAR (photosynthetically active radiation). At nighttime when there is no solar radiation, the stomata are assumed to be completely closed, and R_{st} has an infinite value; finite values only occur when the sun is above the horizontal. G_{st} is computed as the weighted sum of conductances for sunlit and shaded leaves (Zhang et al. 2002):

$$G_{st} = \frac{LAI_{sun}}{r_{stsun}} + \frac{LAI_{shade}}{r_{stshade}} \tag{20}$$

Where LAI_{sun} and LAI_{shade} are the LAI (leaf area index) of sunlit and shaded leaves (m^2 m^{-2}), and r_{stsun} and r_{stsha} are the unstressed leaf stomatal resistances for sunlit and shaded leaves (s m^{-1}).

LAI_{sun} and LAI_{shade} are calculated from total LAI and the solar zenith angle (θ, in radians) (Zhang et al. 2002). θ is approximated from the day of the year (d), universal time (t, in hours), the western longitude (2.618×10^{-2} radians, equal to 1.5°W) and latitude (0.932 radians, equal to 53.4°N) of Sheffield. The procedure is according to Holtslag and Van Ulden (1983).

$$LAI_{sun} = \begin{cases} 2\cos\theta\, 1 - \exp\left(-0.5\,\dfrac{LAI}{\cos\theta}\right), & \text{if} \cos\theta \geq 0 \\ 0, & \text{if } cos\theta < 0 \end{cases} \tag{21}$$

$$LAI_{shade} = LAI - LAI_{sun} \tag{22}$$

$$\theta = 1.570796 - \Phi \tag{23}$$

$$\Phi = \sin^{-1}(\sin\delta \sin 0.932 + \cos\delta \cos 0.932 \cos h) \tag{24}$$

$$h = -2.618 \times 10^{-2} + 0.043\sin(2\,SL) - 0.033\sin(0.0175\,d) + 0.262\,t - \pi \tag{25}$$

$$\delta = \sin^{-1}(0.398 \sin SL) \tag{26}$$

$$SL = 4.871 + 0.0175\,d + 0.033\sin(0.0175\,d) \tag{27}$$

Where Φ is the solar elevation, δ is the solar declination, h is the hour angle through which the earth must turn to bring the meridian of the location directly under the sun, and SL is the solar longitude, all in radians. The limit of 0 in Equation (21) prevents negative LAI values from being computed. The value of θ is computed for the middle of each time period.

r_{stsun} and r_{stsha} are functions of the PAR received by sunlit and shaded leaves respectively (PAR_{sun} and PAR_{shade}, in W m^{-2}), the minimum leaf stomatal resistance for the vegetation type (r_{stmin}), and a land cover specific empirical constant (b_{rs}, in W m^{-2}). The formulation is (Zhang et al. 2002):

$$r_{stsun} = r_{stmin}\left(1 + \frac{b_{rs}}{PAR_{sun}}\right) \tag{28}$$

$$r_{stshade} = r_{stmin}\left(1 + \frac{b_{rs}}{PAR_{shade}}\right) \tag{29}$$

$$PAR_{shade} = \begin{cases} PAR_{diff}\exp(-0.5\,LAI^{0.7}) + 0.07\,PAR_{dir}(1.1 - 0.1\,LAI)\exp(-\cos\theta), \\ \qquad \text{if } LAI < 2.5 \text{ or } SR < 200 \text{ W m}^{-2} \\ PAR_{diff}\exp(-0.5\,LAI^{0.8}) + 0.07\,PAR_{dir}(1.1 - 0.1\,LAI)\exp(-\cos\theta), \\ \qquad \text{if } LAI \geq 2.5 \text{ or } SR \geq 200 \text{ W m}^{-2} \end{cases} \tag{30}$$

$$PAR_{sun} = \begin{cases} PAR_{dir}\dfrac{\cos 1.047}{\cos\theta} + PAR_{shade}, & \text{if } LAI < 2.5 \text{ or } SR < 200 \text{ W m}^{-2} \\ PAR_{dir}^{0.8}\dfrac{\cos 1.047}{\cos\theta} + PAR_{shade}, & \text{if } LAI \geq 2.5 \text{ or } SR \geq 200 \text{ W m}^{-2} \end{cases} \tag{31}$$

Where PAR_{dir} and PAR_{diff} are the downward visible radiation fluxes above the canopy from direct beam and diffuse radiation respectively (W m^{-2}). χ represents the angle in radians between the leaf and the sun, taken as 1.047 (equal to 60°) assuming a spherical leaf angle distribution (Zhang et al. 2002). PAR_{dir} and PAR_{diff} are determined by first calculating the potential visible direct beam (PR_{DV}), visible diffuse (PR_{dV}), near infrared direct beam (PR_{DN}) and near infrared diffuse (PR_{dN}) radiation (W m^{-2}), then scaling the fraction of each type of radiation by the total observed solar radiation (Weiss and Norman 1985).

$$PR_{DV} = 600\exp\left(-0.185\frac{P}{101.325}m\right)\cos\theta \tag{32}$$

$$PR_{dV} = 0.4\,(600 - PR_{DV})\cos\theta \tag{33}$$

$$PR_{DN} = \left[720\exp\left(0.06\frac{P}{101.325}w\right)\right]\cos\theta \tag{34}$$

$$PR_{dN} = 0.6\,(720 - PR_{DN} - w)\cos\theta \tag{35}$$

$$PAR_{dir} = \frac{PR_{DV}}{PR_{DV} + PR_{dV} + PR_{DN} + PR_{dN}} SR \tag{36}$$

$$PAR_{diff} = \frac{PR_{dV}}{PR_{DV} + PR_{dV} + PR_{DN} + PR_{dN}} SR \tag{37}$$

Where 600 and 700 represent the partitioning of the solar constant (1320 W m^{-2}) into visible and near infrared radiation, 0.4 and 0.6 are the fractions of intercepted visible and near infrared beam radiation converted to downward diffuse radiation at the surface, m is the optical air mass (dimensionless) and w is the water absorption in the near infrared for 10mm of precipitable water (W m^{-2}). These constants were determined by Weiss & Norman (1985) using empirical data.

These equations produce negative answers when the zenith angle is greater than the horizontal, and PR_{DN} and PR_{dN} also produce negative answers for zenith angles only very slightly above the horizontal. Therefore Equations (34) to (37) each have a lower limit of zero. Additionally, when $m < 0$ (which occurs when the zenith angle is greater than 90°), w is not a number; in this case PR_{DN} and PR_{dN} are also set to 0.

Assuming that precipitable water absorbs only near infrared radiation, and that 10mm is a reasonable minimum, m and w are calculated as follows (Weiss and Norman 1985):

$$m = \frac{1}{\cos\theta} \tag{38}$$

$$w = 1320\ \mathrm{antilog}_{10}\,[-1.195 + 0.4459\ \log_{10} m - 0.0345(\log_{10} m)^2] \tag{39}$$

The other functions in Equation (19) are calculated as follows (Zhang et al. 2003):

$$f_T = \frac{T - T_{min}}{T_{opt} - T_{min}} \left(\frac{T_{max} - T}{T_{max} - T_{opt}}\right)^{\frac{T_{max}-T_{opt}}{T_{opt}-T_{min}}} \tag{40}$$

Where T_{min} and T_{max} are the minimum and maximum temperature (measured in K) outside which complete stomatal closure occurs; and T_{opt} is the temperature (K) at which maximal stomatal opening occurs. These values depend on the type of vegetation present.

$$f_D = 1 - b_{vpd}(\nu^* - \nu) \tag{41}$$

$$v^* = 0.61121 \exp \frac{\frac{18.678 - T_C}{234.5} T_C}{257.14 + T_C} \tag{42}$$

$$v = v^* RH \tag{43}$$

Where b_{vpd} is a land cover dependent water vapour pressure deficit constant (kPa^{-1}); v^* is the saturation water vapour pressure at the ambient air temperature (kPa; calculated here according to the Arden Buck equation (Buck 1981, Buck Research Manual 1996)) and v is the ambient water vapour pressure (kPa), such that $v^* - v$ is the water vapour deficit; T_C is the temperature in Celsius, i.e. $T - 273.15$; and RH is the relative humidity as a proportion.

$$f_\Psi = \begin{cases} \dfrac{-0.72 - 0.0013\,SR - \Psi_{c2}}{\Psi_{c1} - \Psi_{c2}}, & \text{if} -72 - 0.0013SR \leq \Psi_{c1} \\ 1, & \text{if} -72 - 0.0013SR > \Psi_{c1} \end{cases} \tag{44}$$

Where Ψ_{c1} and Ψ_{c2} are land cover dependent parameters specifying leaf water potential dependency (MPa).

The other resistance components in Equation (17) are less complicated to determine. R_m for NO_2 is 0, regardless of whether vegetation is present (Zhang et al. 2002). R_{ac} is calculated as (Zhang et al. 2003):

$$R_{ac} = \frac{R_{ac0}\, LAI^{0.25}}{u_*^2} \tag{45}$$

Where R_{ac0} is a land cover dependent reference value that changes seasonally with the canopy structure. R_{ac0} at any time (t) is determined according to the seasonal (t), annual minimum (min) and annual maximum (max) LAI, and the difference between the annual maximum and minimum R_{ac0} (Zhang et al. 2003):

$$R_{ac0}(t) = R_{ac0}(min) + (R_{ac0}(max) - R_{ac0}(min)) \frac{LAI(t) - LAI(min)}{LAI(max) - LAI(min)} \tag{46}$$

R_g is dependent on land cover, the pollutant and whether conditions are wet or dry (i.e. whether there has been rainfall in the previous two hours; dew should again be considered separately but is omitted here). Because R_g for most pollutants is unknown, values for NO_2 are estimated from data for O_3, and scaled according to values based on comparative studies of O_3 and NO_2 deposition (Zhang et al. 2002). For NO_2, using known land cover specific values of $R_g(O_3)$, and with β as the scaling parameter, the scaling equation takes the form (Zhang et al. 2002):

$$R_g(\mathrm{NO_2}) = \frac{R_g(\mathrm{O_3})}{\beta} \tag{47}$$

R_{cut} also depends on whether conditions are wet or dry, and on pollutant-specific parameters known empirically for only O_3 and SO_2 (R_{cut} and R_{cutd0} for wet and dry conditions respectively) (Zhang et al. 2003). Therefore a scaling equation is again used:

$$R_{cut}(NO_2) = \begin{cases} 0, & \text{if vegetated} \\ \dfrac{R_{cutw0}(\mathrm{O_3})}{LAI^{0.5}\, u_*\, \beta}, & \text{if vegetated and wet} \\ \dfrac{R_{cutd}\ (\mathrm{O_3})}{\exp(0.03\, RH)\, LAI^{0.25}\, u_*\, \beta}, & \text{if vegetated and dry} \end{cases} \tag{48}$$

When temperatures are below -1°C and conditions are dry, R_g and R_{cut} are higher than estimated by Equations (47) and (48). Therefore in these conditions an adjustment is applied (Zhang et al. 2003):

$$R_{adj} = \begin{cases} R\exp\big(0.2(-1 - T_C)\big), & if\ T_C \geq -4.46 \\ 2R, & if\ T_C < -4.46 \end{cases} \tag{49}$$

Where R_{adj} is the resistance being adjusted. Zhang et al. (2003) report a further adjustment for when snow covers the ground, but as snowfall data was not available (and snowfall is relatively rare in Sheffield) this is omitted here.

Once all the components of Equation (19) have been estimated, Equation (13) can be determined. The deposition velocity can then be evaluated according to Equation (7).

A.1.1.3 PM_{10} deposition velocity

For particulate pollution such as PM_{10}, V_d is heavily dependent on the particle size distribution. When the particle size distribution is known it is therefore preferable to calculate V_d for each fraction separately. As this information was unavailable, a simulated particle size distribution was generated following Clarke and Hamilton (no date), who describe the particle size distribution initialisation procedure for an urban pollutant dispersion/transport model.

The initial particle size distribution is simulated as follows(Clarke and Hamilton, no date). One million random numbers are generated within each of three log-normal distributions with mean diameter (d_{p0}) of 0.075, 0.5 and 4µm (σ = 1.5, 1.8 and 1.8µm respectively). These distributions are summed and truncated at 0.01 and 10µm. This distribution is subdivided into 30 bands defined logarithmically (i.e. $\delta \log d_{p0}$ is constant). Assuming all particles are spherical, the volume and mass of each particle is computed, and the fractions of the total particle mass within each size band (f_p) determined. The total V_d is then the weighted sum of deposition within each size band (Zhang et al. 2001, Environmental Protection Agency 2004):

$$V_d = \sum_{i=1}^{N} f_{p,i} \left(V_{g,i} + \frac{1}{R_{a,i} + R_{s,i}} \right) \tag{50}$$

Where V_g is the gravitational settling velocity (m s^{-1}) and R_s is the surface resistance (s m^{-1}) R_a is calculated according to Equation (14). However, before these components can be determined, the hygroscopic growth of particles, which is especially important in high humidity conditions, must be taken into consideration (Zhang et al. 2001). Particle growth for a typical urban aerosol can be described as (Gerber 1985):

$$d_p = 2 \times 10^{-6} \left[\frac{0.3926 \left(\frac{d_{p0} \times 10^6}{2}\right)^{3.101}}{4.19 \times 10^{-11} \left(\frac{d_{p0} \times 10^6}{2}\right)^{-1.404} - \log RH} + \left(\frac{d_{p0} \times 10^6}{2}\right)^3 \right]^{\frac{1}{3}} \tag{51}$$

Where d_p is the particle diameter (m) considering hygroscopic growth following emission, assuming particle size is in equilibrium with the air. V_g is calculated as (Zhang et al. 2001):

$$V_g = \frac{p\, d_p^2\, g\, S_{CF}}{18\, \eta} \tag{52}$$

$$S_{CF} = 1 + \frac{2\lambda}{d_p} \left[1.257 + 0.4 \exp\left(-0.55 \frac{d_p}{\lambda} \right) \right] \tag{53}$$

Where p is the density of particulate matter (g m^{-3}), g is the acceleration due to gravity (m s^{-2}), S_{CF} is the slip correction factor for small particles (dimensionless), η is the absolute viscosity of air (g m^{-1} s^{-1}) and λ is the mean free path of air molecules (m). λ depends upon air temperature, pressure and kinematic viscosity (Zhang et al. 2001), but can be taken as 6.53×10^{-8}m (Zannetti 1990, Environmental Protection Agency 2004). The absolute viscosity is calculated by Sutherland's formula:

$$\eta = 1.827 \times 10^2 \frac{291.15 + 120}{T + 120} \left(\frac{T}{291.15} \right)^{\frac{3}{2}} \tag{54}$$

Where 1.827×10^{-2} g m^{-1} s^{-1} is a reference viscosity for air, 291.15K is a reference temperature for air and 120K is Sutherland's constant for air. The surface resistance is calculated as follows (Zhang et al. 2001):

$$R_s = \begin{cases} \dfrac{1}{3\,u_*(E_B + E_{IM} + E_{IN})R_1}, & \text{if vegetated} \\ \dfrac{1}{3\,u_*(E_B + E_{IM})R_1}, & \text{if unvegetated} \end{cases} \tag{55}$$

Where 3 is an empirical constant; E_B, E_{IM} and E_{IN} are the collection efficiencies of particles by Brownian diffusion, impaction and interception respectively; and R_1 is fraction of particles that stick to the surface, i.e. that do not rebound. The term E_{IN} is only relevant for vegetated surfaces, so is omitted otherwise. The collection efficiencies are computed as follows (Zhang et al. 2001):

$$E_B = \left(\frac{\nu}{D}\right)^{-\gamma} \tag{56}$$

$$E_{IM} = \left(\frac{St}{\alpha + St}\right)^2 \tag{57}$$

$$E_{IN} = 0.5\left(\frac{d_p}{A}\right)^2 \tag{58}$$

Where α and γ are land cover dependent coefficients, D is the Brownian diffusivity of the particulate matter (m^2 s^{-1}), St is the Stokes number, and A is the land cover and season dependent characteristic radius of collectors such as leaf hairs, grass blades, tree needles etc. D is computed as (Environmental Protection Agency 2004):

$$D = 8.09 \times 10^{-14} \frac{T\,S_{CF}}{d_p \times 10^6} \tag{59}$$

Particles only rebound if the diameter is larger than 5μm. Therefore R_1 is formulated as (Zhang et al. 2001):

$$R_1 = \begin{cases} 1, & \text{if } d_p \le 5\text{μm} \\ \exp(-St^{0.5}), & \text{if } d_p > 5\text{μm} \end{cases} \tag{60}$$

The formulation of St depends on whether the surface is vegetated (Zhang et al. 2001):

$$St = \begin{cases} \dfrac{V_g u_*}{g\,A}, & \text{if vegetated} \\ \dfrac{V_g u_*^2}{g\,\nu}, & \text{if unvegetated} \end{cases} \tag{61}$$

Finally the calculations of V_d for each size band are scaled in Equation (50), before calculation of the pollutant flux according to Equation (7).

A.1.2 Input parameters and data

A.1.2.1 Meteorological data

Six meteorological parameters are required for the functioning of the model: air pressure (P), relative humidity (RH), solar radiation (SR), air temperature (T), wind speed (U) and rainfall. As spatially resolved meteorological data was not available, a single value for each meteorological variable was applied to the entire study area at each time step.

The data input to the model was supplied by a combination of two weather stations: the weather station used by Sheffield City Council AIRVIRO modellers, situated near Worksop Road, NE Sheffield (data supplied by Andrew Elleker); and a station on Derbyshire Lane, SW Sheffield (data available from http://www.sheffieldweather.co.uk, accessed 27/05/2010). The stations are approximately 6km apart.

The AIRVIRO model output was generated using Worksop Road station data as input (Andrew Elleker, personal communication). Worksop Road station was therefore used preferentially as a data source for present model, and SR, T and U data originate from this station. However P, RH and rainfall were not available from this site and therefore Derbyshire Lane data was used instead. The Worksop Road data was supplied as hourly averages. Derbyshire Lane RH was supplied as five minute averages and converted to hourly averages; rainfall as five minute sums and converted to hourly sums; and P as daily readings, with the reading used for the whole day.

The raw data were checked for missing data or unexpected zeroes through manual searching, and where possible for obviously erroneous values by comparing data from the two sources. A number of cases of missing data were identified. In the Worksop Road data, there was one case of no data recorded for any variable for one hour, five cases of no temperature measurement for single hours, and one case of no temperature measurement for four consecutive hours. In the Derbyshire Lane dataset, there were two cases of no data recorded for any variable for one hour. These data gaps were filled by averaging values for the previous and subsequent hours (assuming a linear change over time where consecutive data were missing). Additionally in the Derbyshire Lane dataset there were nine cases of some missing five-minute measurements within a given hour. In these cases, the measurements that were present were used to generate the hourly averages.

Some conversion of measurement units was necessary. Raw temperature data were given in Celcius; this was converted to Kelvin by adding 273.15. Pressure was given in inHg, and converted to kPa using a multiplication factor of 3.38639. It should be noted that while the barometer, which was located indoors, was calibrated for altitude, temperature and relative humidity corrections could not be made (Jon Groocock, personal communication).

Pasquill stability classes (required for pre-processing; see Section A.1.1.1) were determined using the hourly averages for meteorological data, sunset and sunrise times obtained from the United States Naval Observatory Astronomical Applications Department website (http://aa.usno.navy.mil/data/docs/RS_OneYear.php; accessed 28/07/2009), and other parameters described in the following sections. The daytime stability classification rules were used if at least 30 minutes of any given hour were between sunrise and sunset times.

The height of the wind speed measurements (z) was 10m. The parameters d and t were also derived from the time stamps of the meteorological data.

A.1.2.2 Estimation of NO_2 from NO_x

As mentioned, AIRVIRO models NO_x, necessitating conversion to NO_2 before use in the deposition model. As part of their technical guidance for regulatory air quality modelling (DEFRA 2009), DEFRA have made available a Microsoft Excel calculator for the derivation of NO_2 estimates from NO_x (retrieved from http://www.airquality.co.uk/laqm/tools.php, accessed 06/11/2009). This calculator is suitable for use with combined point and traffic emissions, such as those used here (DEFRA 2009).

The calculator was used to estimate NO_2 from NO_x using local estimated regional concentrations of ozone, NO_x and NO_2, estimates of the fraction of NO_x emitted as NO_2 from non-London urban UK traffic, and AIRVIRO modelled NO_x concentrations. The background NO_x was taken to be the lowest concentration within the modelled field (which was outside the boundaries of the study area); this value was subtracted from the total modelled NO_x to represent the NO_x increment from pollution sources. The calculation was performed for each AIRVIRO modelled grid square within the study area boundaries, for each month.

At very low levels of NO_x (< approx. 7μg m^{-3}) the calculator exhibited unexpected behaviour, with estimated NO_2 jumping to higher levels than for slightly higher NO_x

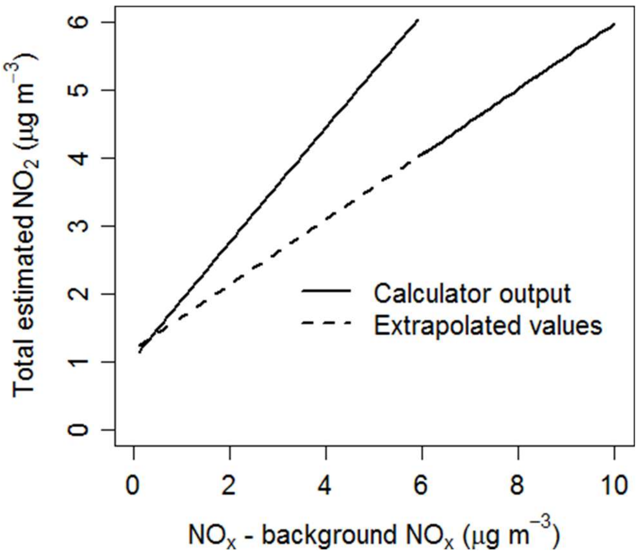


Figure A.1. Unexpected output from the DEFRA NO_x to NO_2 calculator at low concentrations. Solid line shows the original output form the calculator; dotted line shows the extrapolated values used as input to the present model.

concentrations. An example of this is shown for January values in Figure . This appears to be a bug in the calculator, and there is no indication in the documentation as to why it may occur. Therefore the gradient of the almost linear relationship between NO_x and estimated NO_2 in the region 8-10µg m^{-3} NO_x was extrapolated back to calculate the values of NO_2 for use in this model, using month-specific slope and intercept estimates (as shown in Figure).

A.1.2.3 Land cover specific parameters

A number of parameters used in calculation of V_d (deposition velocity) are specific to the land cover, and two (LAI and A - leaf area index and the radius of particle collectors) also vary over the course of a year. This section details the values given to these parameters.

Parameters were obtained from three sources: Zhang et al. (2001), Zhang et al. (2003) and Silva et al. (2007) (only z_0 i.e. roughness length). These sources use different land cover typologies to that in the present study; the matching of land cover typologies is detailed in Table 0.4 and, for Silva et al. (2007), explained further below. The “desert” category is applied to buildings and manmade surfaces because the specified parameter values for the “urban” category appear to apply to an urban matrix of streets, buildings and greenspace patches, whereas the “desert” category parameters are more appropriate for flat and unvegetated surfaces. Thus the “desert” category also applies to unvegetated land.

Table 0.4. Matching of the general land cover typology used in this project to typologies used in Zhang et al. (2001, 2002, 2003)

General typology	Typology of Zhang et al. (2001, 2002)	Typology of Zhang et al. (2003)
1 Arable	7 Crops, mixed farming	15 Crops
2 Building	8 Desert	24 Desert
4 Grassland (improved)	6 Grass	13 Short grass and forbs
5 Grassland (rough)	6 Grass	13 Short grass and forbs
6 Manmade surface	8 Desert	24 Desert
7 Moorland (bog)	6 Grass	13 Short grass and forbs
8 Moorland (heath)	10 Shrubs and interrupted woodlands	10 Evergreen broadleaf shrubs
9 Scrubland	10 Shrubs and interrupted woodlands	11 Deciduous shrubs
12 Unvegetated land	8 Desert	24 Desert
13 Water	13 Inland water	1 Water
14 Woodland (coniferous)	1 Evergreen needleleaf trees	4 Evergreen needleleaf trees
15 Woodland (non-coniferous and mixed)	5 Mixed broadleaf and needleleaf trees	25 Mixed wood forests

Land cover categories 3 (Garden), 10 (Unknown natural surface) and 11 (Unknown surface) are treated as mixed-type land covers with parameters generated from scaled combinations of land covers listed in Table 0.4. Manmade surfaces and buildings are treated as desert, rather than the urban category, because the urban category is parameterised assuming that there is significant vegetation cover in these areas.

During initial model runs the parameters given in Zhang et al. (2001) for urban land cover were found to give unreasonably high values of V_d for PM_{10}, in comparison to estimates by another group (Meteorological Synthesizing Centre - East, no date) and the suggestion of Zhang et al. (2001) that "V_d for the urban LUC is parameterized as having slightly higher values than smoother surfaces, but not as high as that for forests, grass and agricultural lands". Given this behaviour, buildings are given the same V_d PM_{10} values as manmade surfaces. This neglects possible effects of differences in height and z_0, and of deposition to walls. However deposition to walls is likely to be small: Haynie and Lemmons (1990) found PM_{10} V_d to be five times greater to roofs than walls, and Roed (1987) reported V_d of radioactive isotopes to be an order of magnitude higher to roofs than walls. Unfortunately there do not appear to be any empirical comparative studies of V_d to roofs versus ground-level surfaces to validate the approach taken here.

Table 0.5 lists the land cover specific parameter values. All parameters in Table 0.5 are taken from Zhang et al. (2003), except: α and γ (parameters used to determine particle collection efficiencies), which are from Zhang et al. (2001); and values of z_0, which are mostly taken from a parameterisation of the CORINE land cover classes (Silva et al. 2007), as these values were specifically determined for European land cover types and better matching of the typologies was possible.

The matching of CORINE land cover classes is as follows. Silva et al. (2007) give a single z_0 value to irrigated and non-irrigated arable land, which is used here as $z_0(max)$; in the present case arable is also given $z_0(min)$ of sparsely vegetated land to reflect seasonality. Silva et al. (2007) assign a single value to moors, heathland, natural grassland and pastures, which is given here to both grassland and both moorland classes and does not vary seasonally. Woodland is given the z_0 determined for forests by Silva et al. (2007), but non-coniferous and mixed woodland is here given a $z_0(min)$ of two thirds of that value. Unvegetated land and manmade surfaces are given the z_0 assigned to land covers such as airports, bare rock and sparsely vegetated areas. Silva et al al. (2007) do not parameterise scrubland, so the values here are those used by Zhang et al. (2003). Finally, Silva et al. (2007) assign a z_0 of 0 to water, which would produce

Table 0.5. Land cover specific parameters used in V_d (deposition velocity) modelling.

Parameter	Land cover type										
	1	2	4	5	6	7	8	9	12	13	14	15
b_{rs}	40	n/a	50	50	n/a	50	40	44	n/a	n/a	44	44
b_{vpd}	0	n/a	0	0	n/a	0	0.27	0.27	n/a	n/a	0.31	0.34
R_{ac0} (max)	40	n/a	20	20	n/a	20	60	20	n/a	n/a	100	100
R_{ac0} (min)	10	n/a	20	20	n/a	20	60	20	n/a	n/a	100	100
R_{cutd0} (O_3)	4000	n/a	4000	4000	n/a	4000	6000	5000	n/a	n/a	4000	4000
R_{cutw0} (O_3)	200	n/a	200	200	n/a	200	400	300	n/a	n/a	200	200
R_g (O_3)	200	500	200	200	500	200	200	200	500	2000	200	200
r_{stmin}	120	n/a	150	150	n/a	150	150	150	n/a	n/a	250	150
T_{max}	318.15	n/a	313.15	313.15	n/a	313.15	318.15	313.15	n/a	n/a	313.15	315.15
T_{min}	278.15	n/a	278.15	278.15	n/a	278.15	273.15	268.15	n/a	n/a	268.15	270.15
T_{opt}	300.15	n/a	303.15	303.15	n/a	303.15	303.15	288.15	n/a	n/a	288.15	294.15
z_0 (max)	0.05	0.5	0.03	0.03	0.005	0.03	0.03	0.2	0.005	0.0001	0.75	0.75
z_0 (min)	0.005	0.5	0.03	0.03	0.005	0.03	0.03	0.05	0.005	0.0001	0.75	0.5
α	1.2	1.5	1.2	1.2	1.5	1.2	1.3	1.3	50	100	1	1
γ	0.54	0.56	0.54	0.54	0.56	0.54	0.54	0.54	0.54	0.5	0.56	0.56
Ψ_{c1}	-1.5	n/a	-1.5	-1.5	n/a	-1.5	-2	-2	n/a	n/a	-2	-2
Ψ_{c2}	-2.5	n/a	-2.5	-2.5	n/a	-2.5	-4	-4	n/a	n/a	-2.5	-2.5

errors in the present routines; therefore the European Wind Atlas value is used (Silva et al. 2007).

LAI and A are dependent on month and season respectively, as well as land cover type. Parameter values are shown in Table 0.6 and Table 0.7. Monthly values for LAI are taken from Zhang et al. (2003), except that the urban values are set to zero. Values of A are all taken from Zhang et al. (2001). Each month was classified into one of five seasons listed in Zhang et al. (2001) and described in Brook et al. (1999). This was done using month- and latitude-dependent classification rules given for the US by Brook et al. (1999) and identifying the most appropriate latitudinal band for each month, given differences between the US and UK climate patterns, using global temperature maps produced by the Food and Agriculture Organization's Sustainable Development Department (http://www.fao.org/WAICENT/FAOINFO/sustdev/EIdirect/climate/EIsp0002.htm, accessed 28/07/2009). Using this method, December to February were classified as season 4 ("Winter, snow on ground and subfreezing"); March to May as season 5 ("Transitional spring with partially green short annuals"); June to August as season 1 ("Midsummer with lush vegetation"); and September to November as season 2 ("Autumn with cropland that has not been harvested").

A.1.2.4 Simulated parameters

Section A.1.1.3 describes the simulation used to derive values for the particulate matter size distribution: f_p and d_{p0}. f_p is computed as the fraction of total mass of particles within each simulated size band, while d_{p0} is the mean particle size within a band (before hygroscopic growth). The values are shown in Table 0.8.

A.1.2.5 Particulate matter density

p, the density of particular matter, is a parameter in the estimation of V_g, the gravitational settling velocity (Equation (52)). Empirically determined values of p do not appear to be common, and no local estimates were available. Zannetti (1990) suggests that the range of particle densities is at least from 1–11.5x10^6 g m^{-3}. Due to the uncertainty of the value of this parameter, a constant value of 2 x10^6 g m^{-3} is used irrespective of the degree of hydroscopic growth, following the example simulations of Zhang et al. (2001).

Table 0.6. Values of the month and land cover dependent parameter *LAI* (leaf area index).

LC	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
1	0.1	0.1	0.1	0.5	1	2	3	3.5	4	0.1	0.1	0.1
2	0	0	0	0	0	0	0	0	0	0	0	0
4	1	1	1	1	1	1	1	1	1	1	1	1
5	1	1	1	1	1	1	1	1	1	1	1	1
6	0	0	0	0	0	0	0	0	0	0	0	0
7	1	1	1	1	1	1	1	1	1	1	1	1
8	3	3	3	3	3	3	3	3	3	3	3	3
9	0.5	0.5	1	1	1.5	2	3	3	2	1.5	1	0.5
12	0	0	0	0	0	0	0	0	0	0	0	0
13	0	0	0	0	0	0	0	0	0	0	0	0
14	5	5	5	5	5	5	5	5	5	5	5	5
15	3	3	3	4	4.5	5	5	5	4	3	3	3

Table 0.7. Values of the month and land cover dependent parameter *A*.

LC	Season				
	1	2	3	4	5
1	0.02	0.02	0.05	0.05	0.02
2	0.01	0.01	0.01	0.01	0.01
4	0.02	0.02	0.05	0.05	0.02
5	0.02	0.02	0.05	0.05	0.02
6	0.01	0.01	0.01	0.01	0.01
7	0.02	0.02	0.05	0.05	0.02
8	0.01	0.01	0.01	0.01	0.01
9	0.01	0.01	0.01	0.01	0.01
12	0	0	0	0	0
13	0	0	0	0	0
14	0.002	0.002	0.002	0.002	0.002
15	0.005	0.005	0.005	0.005	0.005

A.1.3 Model setup

Meteorological data were obtained for the full year from 01/10/2007 to 30/09/2008, as the most recent full year without significant data gaps. The meteorological data were used to produce an hourly estimate of V_d (deposition velocity) for each land cover class.

As mentioned above, AIRVIRO modelled pollutant concentration was supplied as typical monthly values. Therefore the V_d estimates were aggregated by month and land

Table 0.8. Simulated parameters, f_p (proportion of particular matter in a size band) and d_{p0} (particle diameter before hydroscopic growth). NB 1μm = 1×10^{-6}m.

Band	Size range (m)	f_p	d_{p0} (m)
1	1.00×10^{-8} - 1.26×10^{-8}	5.20×10^{-14}	1.18×10^{-8}
2	1.26×10^{-8} - 1.58×10^{-8}	1.41×10^{-12}	1.49×10^{-8}
3	1.58×10^{-8} - 2.00×10^{-8}	2.46×10^{-11}	1.85×10^{-8}
4	2.00×10^{-8} - 2.51×10^{-8}	2.89×10^{-10}	2.31×10^{-8}
5	2.51×10^{-8} - 3.16×10^{-8}	2.49×10^{-9}	2.89×10^{-8}
6	3.16×10^{-8} - 3.98×10^{-8}	1.60×10^{-8}	3.63×10^{-8}
7	3.98×10^{-8} - 5.01×10^{-8}	7.47×10^{-8}	4.54×10^{-8}
8	5.01×10^{-8} - 6.31×10^{-8}	2.54×10^{-7}	5.68×10^{-8}
9	6.31×10^{-8} - 7.94×10^{-8}	6.32×10^{-7}	7.11×10^{-8}
10	7.94×10^{-8} - 1.00×10^{-7}	1.15×10^{-6}	8.89×10^{-8}
11	1.00×10^{-7} - 1.26×10^{-7}	1.59×10^{-6}	1.11×10^{-7}
12	1.26×10^{-7} - 1.58×10^{-7}	1.84×10^{-6}	1.40×10^{-7}
13	1.58×10^{-7} - 2.00×10^{-7}	2.59×10^{-6}	1.78×10^{-7}
14	2.00×10^{-7} - 2.51×10^{-7}	6.22×10^{-6}	2.26×10^{-7}
15	2.51×10^{-7} - 3.16×10^{-7}	1.81×10^{-5}	2.84×10^{-7}
16	3.16×10^{-7} - 3.98×10^{-7}	4.73×10^{-5}	3.57×10^{-7}
17	3.98×10^{-7} - 5.01×10^{-7}	1.09×10^{-4}	4.48×10^{-7}
18	5.01×10^{-7} - 6.31×10^{-7}	2.15×10^{-4}	5.63×10^{-7}
19	6.31×10^{-7} - 7.94×10^{-7}	3.72×10^{-4}	7.07×10^{-7}
20	7.94×10^{-7} - 1.00×10^{-6}	5.67×10^{-4}	8.88×10^{-7}
21	1.00×10^{-6} - 1.26×10^{-6}	8.54×10^{-4}	1.12×10^{-6}
22	1.26×10^{-6} - 1.58×10^{-6}	1.50×10^{-3}	1.42×10^{-6}
23	1.58×10^{-6} - 2.00×10^{-6}	3.47×10^{-3}	1.79×10^{-6}
24	2.00×10^{-6} - 2.51×10^{-6}	9.33×10^{-3}	2.26×10^{-6}
25	2.51×10^{-6} - 3.16×10^{-6}	2.39×10^{-2}	2.84×10^{-6}
26	3.16×10^{-6} - 3.98×10^{-6}	5.46×10^{-2}	3.56×10^{-6}
27	3.98×10^{-6} - 5.01×10^{-6}	1.08×10^{-1}	4.47×10^{-6}
28	5.01×10^{-6} - 6.31×10^{-6}	1.85×10^{-1}	5.61×10^{-6}
29	6.31×10^{-6} - 7.94×10^{-6}	2.71×10^{-1}	7.05×10^{-6}
30	7.94×10^{-6} - 1.00×10^{-5}	3.41×10^{-1}	8.84×10^{-6}

cover in order to produce typical monthly values of F (pollutant flux to surfaces) for each land cover.

For each 500m grid square in the study area, the proportion of each land cover and land cover specific V_d and F estimates were used to compute the grid square's monthly

V_d and F. The values for HEC and OA polygons were obtained using the method described above to calculate V_d for each polygon from the land cover map, and obtaining area weighted means of F from the original 500m^2 grid.

A.1.3.1 Modelling the ecosystem service

Section 3.1.4 discusses stages in the generation of the ecosystem service of pollutant deposition. The values of V_d (deposition velocity) and F (pollutant flux to surfaces) are considered to be measures of potential and actual ecosystem service production respectively. However, since these values are non-zero to manmade surfaces, the actual values of ecosystem service production are calculated by re-running the model at the same spatiotemporal scale but with a constant land cover composition of 28% buildings and 72% manmade surfaces (as calculated from the actual total ratio of building to manmade surface area in the whole study area), and comparing these values to V_d and F given the actual land cover composition.

Finally, to produce a single output variable quantifying the level of ecosystem service provided by a spatial unit of analysis, the mean was taken of the values for the two pollutants. These calculations are demonstrated for F in Eqn. (62):

$$\text{Ecosystem service} = \frac{\left(\frac{\text{actual } NO_2F}{\text{all manmade } NO_2F} + \frac{\text{actual } PM_{10}F}{\text{all manmade } PM_{10}F}\right)}{2} \tag{62}$$

Since F is computed as V_d multiplied by the pollutant concentration (Eqn. (7)), and the same pollutant concentration is used in these two computations of F, the effect of calculating the ecosystem service in this manner is to cancel out the pollutant concentration. The result of Eqn. (62) is simply the average of the ratios between the two values of V_d for each pollutant. This can be interpreted as the ratio between the amount of pollution that is removed from the air with greenspace present in the urban environment, compared to the amount that would be removed if it were not.

A.1.4 Sensitivity tests

Sensitivity tests of the V_d model were performed in order to highlight where poorly known data or parameters may be causing disproportionately large error in the model, as well as to improve understanding of how the model works. Sensitivity tests involved running the entire model, for all land covers for the whole year, changing a single input variable while leaving all the others unchanged (where a single variable is considered to be every value of a meteorological input variable or land cover specific parameter, or a non-varying constant). Each variable was permuted by addition, then subtraction, of 10% of the original value. The absolute percentage change of V_d with the permuted parameter compared with the non-permuted value was calculated for each model time step and land cover combination.

As the model output is used as month and land cover specific averages, the results of each permutation were first aggregated by month and land cover. These aggregated values were then compared to the non permuted average values by calculating the absolute percentage change. To limit data analysis, sensitivities were first examined as total averages, i.e. the average over all land cover and month specific aggregated values. Due to the number of parameters that vary between land covers, percentage changes were also aggregated by land cover (i.e. average over all months and time steps, for each land cover separately) and examined in this form.

In order to keep the permuted variables within reasonable value ranges, the following steps were taken. Temperatures (T, T_{max}, T_{min}, T_{opt}) were permuted by 10% of their value in Celcius rather than Kelvin, so that, for example, a temperature of 10°C (283.15K) became 9°C /11°C rather than -18°C/38°C, as the latter are unreasonable temperatures for Sheffield's climate (whereas permuting the other meteorological values by 10% does not take them outside a reasonable range). RH was given a maximum value of 1 (i.e. 100%). R_{ac0} (min) and R_{ac0} (max) were permuted simultaneously, as were R_{cutd0} and R_{cutw0} and z_0 (min) and z_0 (max).

Sensitivity to frequency of canopy wetness was also tested, by adding to the actual distribution of wet canopy periods a number of hours equivalent to 10% of the total hours; and then assigning 10% of the wet canopy hour as dry. This process was repeated five times, and the average proportional change taken as the sensitivity.

A.1.4.1 NO_2 sensitivity

Table 0.9 and Table 0.10 show the results of sensitivity testing for V_d NO_2 as total averages (proportional change and rank order of change) and land cover specific averages (rank order only) respectively. The order of sensitivity changes slightly depending on whether values are increased or decreased.

The model is highly sensitive to the values of κ in both directions (increase and decrease), with a 10% change in κ causing an 8-9% change in V_d. This importance is relatively constant across land covers (Table 0.10), and is unsurprising because κ is important in determination of R_a, R_b and u_*, the latter of which is itself also a

Table 0.9. Average sensitivity of V_d NO_2 to permutation of input parameters over all times and land covers (proportional changes; ranks in italics).

	Plus 10%		**Minus 10%**	
Meteorological inputs				
P	0.0291	*5*	0.0303	*6*
RH	0.0264	*7*	0.0325	*5*
SR	0.0089	*14*	0.0099	*13*
T	0.0036	*20*	0.0055	*18*
U	0.0604	*2*	0.0625	*2*
Wet canopy freq.	0.0012	*24*	0.0018	*24*
Land-cover dependent parameters				
b_{rs}	0.0078	*16*	0.0086	*15*
b_{vpd}	0.0021	*23*	0.0021	*23*
LAI	0.0056	*17*	0.0066	*17*
R_{ac0}	0.0117	*12*	0.0131	*12*
R_{cut0}	0.0031	*21*	0.0037	*21*
R_g	0.0373	*4*	0.0427	*4*
r_{stmin}	0.0150	*9*	0.0175	*9*
T_{max}	0.0023	*22*	0.0033	*22*
T_{min}	0.0004	*25*	0.0004	*26*
T_{opt}	0.0102	*13*	0.0094	*14*
z_0	0.0137	*11*	0.0146	*11*
Ψ_{c1}	0.0003	*26*	0.0005	*25*
Ψ_{c1}	0.0001	*27*	0.0001	*27*
Constants				
a	0.0051	*18*	0.0054	*19*
b	0.0049	*19*	0.0045	*20*
D_{H20}	0.0150	*9*	0.0175	*9*
D_{NO2}	0.0277	*6*	0.0291	*7*
z	0.0173	*8*	0.0196	*8*
β	0.0419	*3*	0.0445	*3*
κ	0.0819	*1*	0.0879	*1*
χ	0.0088	*15*	0.0069	*16*

Table 0.10. Rank average sensitivity of V_d NO_2 to permutation of input parameters, by land cover.

	Plus 10%										Minus 10%									
Land cover	**1**	**2**	**4/5**	**6**	**8**	**9**	**12**	**13**	**14**	**15**	**1**	**2**	**4/5**	**6**	**8**	**9**	**12**	**13**	**14**	**15**
Meteorological inputs																				
P	3	4	4	5	6	5	5	6	10	7	3	5	4	5	8	5	5	5	10	7
RH	26	14	26	14	3	3	14	14	3	1	26	14	26	14	3	3	14	14	2	1
SR	15	12	17	12	12	15	12	11	11	10	14	11	17	11	11	14	11	11	11	10
T	13	10	10	11	10	17	11	12	18	19	11	10	10	12	10	20	12	12	20	18
U	2	5	2	4	2	2	4	4	2	3	2	4	2	4	2	2	4	4	3	3
Wet canopy freq.	23	13	22	13	24	20	13	13	20	24	23	13	20	13	24	18	13	13	18	24
Land-cover dependent parameters																				
b_{rs}	17	14	18	14	13	16	14	14	14	13	15	14	18	14	12	15	14	14	13	11
b_{vpd}	27	14	27	14	20	21	14	14	19	18	27	14	27	14	20	21	14	14	19	19
LAI	19	14	14	14	16	13	14	14	17	17	17	14	15	14	17	13	14	14	17	17
R_{ac0}	8	14	7	14	9	11	14	14	12	14	8	14	7	14	9	11	14	14	12	12
R_{cut0}	20	14	16	14	23	18	14	14	16	21	20	14	16	14	22	17	14	14	16	21
R_g	6	2	6	3	19	10	3	2	15	16	6	1	6	2	19	10	2	1	14	16
r_{stmin}	11	14	12	14	7	7	14	14	5	5	9	14	12	14	6	7	14	14	5	5
T_{max}	22	14	19	14	18	24	14	14	24	20	21	14	14	14	16	23	14	14	23	20
T_{min}	21	14	21	14	25	25	14	14	23	22	22	14	22	14	26	24	14	14	24	23
T_{opt}	9	14	8	14	5	23	14	14	21	12	13	14	9	14	5	25	14	14	22	14
z_0	10	8	11	10	15	12	10	10	9	9	12	8	11	9	14	12	9	10	9	9
Ψ_{c1}	24	14	24	14	25	26	14	14	26	26	24	14	24	14	25	26	14	14	26	26
Ψ_{c1}	25	14	25	14	25	26	14	14	26	26	25	14	25	14	26	27	14	14	27	27
Constants																				
a	18	9	20	9	21	19	9	8	22	23	16	9	21	8	21	19	8	8	21	22
b	16	11	23	8	22	22	8	5	25	25	18	12	23	10	23	22	10	6	25	25
D_{H20}	11	14	12	14	7	7	14	14	5	5	9	14	12	14	6	7	14	14	5	5
D_{NO2}	4	7	5	7	4	4	7	9	4	4	4	7	5	7	4	4	7	9	4	4
z	7	6	9	6	14	9	6	7	8	8	7	6	8	6	13	9	6	7	7	8
β	5	1	3	2	17	6	2	1	7	15	5	2	3	3	18	6	3	2	8	13
κ	1	3	1	1	1	1	1	3	1	2	1	3	1	1	1	1	1	3	1	2
χ	14	14	15	14	11	14	14	14	13	11	19	14	19	14	15	16	14	14	15	15

component of R_a, R_b and R_{cut}. κ is now thought with some certainty to have a universal value close to 0.41, despite earlier measurements suggesting a lower value

(Kantha and Clayson 2000); therefore there should be low error associated with this constant.

V_d is also highly sensitive to U for all land covers, which is again unsurprising because of the importance of wind speed to meteorological processes (Zannetti 1990). Other meteorological variables to which V_d is sensitive are P and RH, which are components of the determination of the kinematic viscosity of air and the fraction of stomatal closure due to water vapour pressure deficit respectively. The importance of P and RH is, however, somewhat more variable between land covers. The data for these parameters are likely to be the greatest source of inaccuracy in the model, because the measurement error is unknown and more importantly because a single measurement is applied to the whole study area.

The model is relatively insensitive to most of the land cover dependent parameters, except for R_g. R_{ac0}, r_{stmin}, T_{opt} and z_0 cause a 1-2% change in V_d (although the rank importance varies considerably between land covers), while the others all cause < 1%. Therefore the fact that the values of many of these parameters are only estimated, rather than measured directly, should not be too significant. Table 0.10 indicates that the importance of R_g arises from land covers with no vegetation, probably because R_c for these land covers is determined from R_g alone.

The scaling parameter β also has a large effect on V_d for most land covers. The accuracy of β is difficult to determine, as the studies discussed in Zhang et al. (2002), according to which β was chosen, found considerable variation in comparative deposition rates.

Finally, the molecular diffusivities D_{NO_2} and D_{H_2O} cause a moderate change of 2-3% on average, although this again varies between land covers. This may also have some influence on the accuracy of results because constant values are used here as elsewhere (Hicks et al. 1987, Environmental Protection Agency 2004) despite some weather dependency .

A.1.4.2 PM$_{10}$ sensitivity

The PM$_{10}$ V_d model has very high sensitivity to RH and d_{p0}, with changes of around 50% and 20% resulting from plus and minus 10% changes to these parameters respectively (Table 0.11). This is unsurprising given that the size of a particle has strong effects on its gravitation and diffusion effects. The influence of RH operates via its effect on d_p: for example, the mean value of RH during the study period is 0.81, which for the largest d_{p0} class increases d_p by a factor of 1.47. For $RH = 0.99$ (the highest RH recorded here,), the factor is 3.59. At $RH = 1$, which is found within the plus 10% sensitivity test, the factor jumps to >4000 (Gerber (1985) reports that Equation (51) is valid for $0 < RH \leq 1$).

The dependence of V_d on particle size is problematic as the particle size distribution is simulated. The problem is exacerbated by the high sensitivity of V_d to p, which is also not empirically determined. During interpretation of the model output it must therefore be borne in mind that the values of V_d refer only to one possible type of PM_{10}, and that variations in particle size distribution or density will result in very different V_d.

The value of g also has strong effects on V_d, but this is a comparatively accurately known constant. As with V_d for NO_2, V_d PM_{10} is quite sensitive to changes in the value of κ and variations in U; probably for similar reasons as above.

Finally, V_d is slightly sensitive to three land cover dependent parameters (A, z_0, α), with changes of 1-2% resulting from 10% changes in the values. As might be expected,

Table 0.11. Average sensitivity of V_d PM_{10} to permutation of input parameters over all times and land covers (proportional changes; ranks in italics).

	Plus 10%		**Minus 10%**	
Meteorological inputs				
P	0.0015	*13*	0.0018	*13*
RH	0.5003	*1*	0.5185	*1*
SR	0.0010	*15*	0.0011	*15*
T	0.0029	*12*	0.0029	*12*
U	0.0393	*6*	0.0356	*6*
Land cover dependent parameters				
A	0.0158	*8*	0.0187	*8*
z_0	0.0128	*9*	0.0127	*10*
α	0.0188	*7*	0.0226	*7*
γ	0.0044	*11*	0.0100	*11*
Constants				
a	0.0005	*16*	0.0007	*16*
b	0.0003	*17*	0.0003	*17*
d_{p0}	0.2108	*2*	0.1921	*2*
g	0.0819	*4*	0.0819	*4*

p	0.0990	*3*	0.0997	*3*
z	0.0120	*10*	0.0148	*9*
κ	0.0433	*5*	0.0384	*5*
λ	0.0014	*14*	0.0014	*14*

Table 0.12. Rank average sensitivity of V_d PM_{10} to permutations of input parameters, by land cover.

Land cover	**Plus 10%**									**Minus 10%**								
	1	**2/6**	**4/5/7**	**8**	**9**	**12**	**13**	**14**	**15**	**1**	**2/6**	**4/5/7**	**8**	**9**	**12**	**13**	**14**	**15**
Meteorological inputs																		
P	14	8	14	12	13	8	10	16	16	14	9	14	11	13	9	10	16	16
RH	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
SR	16	16	17	17	16	16	16	11	13	15	16	16	17	16	16	16	12	13
T	12	9	12	10	10	9	6	12	12	12	10	12	10	11	10	6	13	12
U	6	5	6	5	5	5	8	4	4	6	5	6	6	6	5	8	4	5
Land cover dependent parameters																		
A	8	17	8	9	9	17	17	9	8	8	17	8	9	9	17	17	8	7
z_0	10	13	10	13	11	13	14	7	9	11	13	11	13	12	13	14	9	10
α	7	7	7	8	8	7	11	6	6	7	8	7	8	8	8	11	6	6
γ	11	10	11	7	7	10	5	13	11	9	6	9	5	5	6	5	11	11
Constants																		
a	17	15	15	15	15	15	15	14	15	17	15	15	15	15	15	15	14	15
b	15	14	16	16	17	14	13	17	17	16	14	17	16	17	14	13	17	17
d_{p0}	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2	2
g	4	4	4	4	4	4	4	10	7	4	4	4	4	4	4	4	10	8
p	3	3	3	3	3	3	3	5	5	3	3	3	3	3	3	3	5	3
z	9	12	9	11	12	12	12	8	10	10	12	10	12	10	12	12	7	9
κ	5	6	5	6	6	6	9	3	3	5	7	5	7	7	7	9	3	4
λ	13	11	13	14	14	11	7	15	14	13	11	13	14	14	11	7	15	14

there is some variation of rank importance between land covers (Table 0.12). The other variables have only very small effects on V_d.

A.1.4.3 Nighttime stability classes

The sensitivity of V_d for both NO_2 and PM_{10} to violation of the assumption that all nights have cloud cover > 4/10 was also tested. Sensitivity was tested for the case that all nights had cloud cover < 4/10, and also the case that 10% of nights had cloud cover < 4/10. In the latter case, sensitivity was determined from five repetitions of assignment of cloud cover < 4/10 to a randomly selected 10% of nighttime time periods, and calculating the average percentage change of V_d.

As Table 0.13 shows, V_d for both pollutants is comparatively low. Even if the assumption were wrong in all cases, V_d for NO_2 and PM_{10} would differ by only 4% and

Table 0.13. Sensitivity of V_d to violation of the assumption of a constant nighttime cloud cover > 4/10 for 10% of the nighttime of all the nighttime across the simulated period. Shown as proportional changes.

Pollutant	10%	All
NO_2	0.0039	0.0400
PM_{10}	0.0015	0.0153

1.5% respectively. These values do not differ greatly between land covers (2-6% for NO_2; 0.02-6% for PM_{10}, with the higher values for woodland). Therefore the potential consequences of this assumption are considered to be small.

A.2 Heat island mitigation model

The urban energy exchange model developed by Tso et al. (1991), and used by Whitford et al. (2001) and Gill (2006), is further customised and implemented here for modelling the heat island effect in Sheffield. The model is based on traditional energy exchange equations, with the addition of a term representing heat storage in buildings (Tso et al. 1991):

$$M = R - H - LE - G \tag{63}$$

Where M is heat storage in buildings, R is the net radiation flux, H is the sensible heat flux to the air, L is the latent heat of water, E is the evaporation rate and G is the heat flux to the soil substrate. Figure 4.4 represents these fluxes, and illustrates the urban layers important to the model.

The model proceeds by pairs of simultaneous, time dependent, linear, first-order differential equations representing the temperature at levels 0 and s (see Figure 4.4). Separate equations are used for level 0 for daytime and nighttime, while the equation for level s remains the same (Gill 2006).

A.2.1 Model formulation

A.2.1.1 Energy exchange components

The following description of the model formulation uses the account given by Gill (2006). R is equal to the insolation flux minus long wave radiation emitted from the surface to the atmosphere, and varies according to the solar generation equation:

$$R = (1 - \alpha)\, \tau\, R_0\, |\sin\phi \sin\delta + \cos\phi \cos\delta \cos\gamma| - R_1 \tag{64}$$

Where α is the surface albedo, τ is the transmission coefficient, R_0 is the solar constant, ϕ is the latitude, δ is the solar declination, γ is the solar hour angle and R_1 is the net infrared flux at the surface of the earth. Neglecting emissivity and absorptivity, R_1 can be approximated as:

$$R_1 = \sigma(T_0^4 - T_{sky}^4) \tag{65}$$

Where T_0 and T_{sky} are the surface and sky temperatures, and σ is the Stefan-Boltzman constant, taken as 5.67×10^{-8} W m^{-2} K^{-4}. However, the unknown sought by the model (T_0) appears in Eqn. (65) in the fourth power, so in the model R is estimated for daytime from an empirically fitted sine wave for the particular geographic location during the daytime, and for nighttime by assuming a uniform loss of radiative energy:

$$R = \begin{cases} a_3 \sin(\omega\, t), & \text{for daytime} \\ a_3', & \text{for nighttime} \end{cases} \tag{66}$$

$$\omega = \frac{\pi}{3600\, hoD} \tag{67}$$

Where a_3 is the peak insolation, t is the number of seconds since sunrise and hoD is the number of hours of daylight at the location.

H and LE are described in a bulk adiabatic transfer form by the following equations:

$$H = -K_h C_a (T_2 - T_0) \tag{68}$$

$$LE = -K_v L (q_2 - q_0) \tag{69}$$

$$K_h = K_v = \frac{\rho_a k^2 U_2}{\left[\ln \frac{Z_2}{Z_0}\right]^2} \tag{70}$$

Where K_h and K_v are the eddy diffusivities of heat and water vapour (described by Eqn. (70) for neutral or near-neutral atmospheric conditions), C_a is the specific heat of air at a constant pressure, T_2 and T_0 are the temperatures at levels 2 and 0 respectively (see Figure 4.4), q_2 and q_0 are the specific humidities at levels 2 and 0 respectively, ρ_a

is the air density, k is the von Kármán constant (taken as 0.41), U_2 is the wind velocity at level 2, Z_2 is the height of the surface boundary layer (SBL) and Z_0 is the surface roughness. Eqn. (70) can be substituted into Eqns. (68) and (69) to yield two finite difference equations.

q_0, the specific humidity of the atmosphere, takes the form of a quadratic approximation:

$$q_0 = E_f\left[3.74 + 2.64\left(\frac{T_0}{10}\right)^2\right] \times 10^{-3} \tag{71}$$

Where E_f is the evaporating fraction, i.e. the fraction of the area under study with an evapotranspiring land cover. However, Eqn. (71) contains T_0, the sought unknown, so $q_0(T_0)$ is linearised by Taylor expansion around a reference temperature (T_f), and higher powers of $T_0 - T_f$ are neglected. The expansion yields an equation with two determinable constants, a_1 and a_2:

$$q_0(T_0) = q_0(T_f) + \left.\frac{\delta q_0}{\delta T_0}\right|_{T_0=T_f} \times (T_0 - T_f) \tag{72}$$

$$q_0(T_0) = E_f(a_1 + a_2 T_0) \tag{73}$$

G, which represents the transfer of heat through the intermediate soil level s to the lower level b (Figure 4.4), where the temperature is assumed to be constant (T_b). G is expressed by the Fourier law:

$$G = -\frac{k_s}{d}(T_s - T_0) \tag{74}$$

Where k_s is the soil thermal conductivity, T_s is the soil temperature at level s and d is the soil depth at level s. The soil temperatures at levels 0, s and b are linked by the finite difference form of the differential equation given by Eqn. (74):

$$\frac{dT_s}{dt} = \frac{k_s}{\rho_s C_s d^2}(T_b - 2T_s + T_0) \tag{75}$$

Where ρ_s is the soil density and C_s is the specific heat capacity of the soil.

The model reduces building mass to a homogeneous plane without volume but with the ability to store thermal energy. M varies with the rate of temperature change as follows:

$$M = m_c C_c \frac{dT_0}{dt} \tag{76}$$

Where m_c is the equivalent homogeneous building mass per unit of surface area, C_c is the specific heat of the building material, and dT_0/dt is the rate of change of the surface temperature, which is assumed to be in thermal equilibrium with the uniform building mass temperature.

A.2.1.2 Simultaneous equations

Equations (63) and (75) represent the temperature at levels 0 and s respectively. Equations (66), (68), (69), (73), (74) and (76) are substituted into the differential form of Eqn. (63), and Eqn. (75) is reformulated, to produce the simultaneous equations (Gill 2006):

$$\frac{dT_0}{dt} = \begin{cases} b_1 \sin(\omega t) + b_2 T_0 + b_3 T_s + b_4, & \text{for daytime} \\ b_2 T_0 + b_3 T_s + b_5, & \text{for nighttime} \end{cases} \tag{77}$$

$$\frac{dT_s}{dt} = b_6 T_0 + b_7 T_s + b_8 \tag{78}$$

Where the b coefficients are as follows (Gill 2006):

$$b_1 = a_3 (m_c C_c)^{-1} \tag{79}$$

$$b_2 = -\left[\rho_a k^2 U_2 \left(C_a + L E_f a_2\right) \left(\ln \frac{Z_2}{Z_0}\right)^{-2} + \frac{k_s}{d}\right] (m_c C_c)^{-1} \tag{80}$$

$$b_3 = k_s (m_c C_c d)^{-1} \tag{81}$$

$$b_4 = \rho_a k^2 U_2 \left[C_a T_2 + L\left(q_2 - E_f a_1\right)\right] \left[\ln \frac{Z_2}{Z_0}\right]^{-2} (m_c C_c)^{-1} \tag{82}$$

$$b_5 = \left\{\rho_a k^2 U_2 \left[C_a T_2 + L\left(q_2 - E_f a_1\right)\right] \left[\ln \frac{Z_2}{Z_0}\right]^{-2} + a_3'\right\} (m_c C_c)^{-1} \tag{83}$$

$$b_6 = k_s (\rho_s C_s d^2)^{-1} \tag{84}$$

$$b_7 = -2 k_2 (\rho_s C_s d^2)^{-1} \tag{85}$$

$$b_8 = k_s T_b (\rho_s C_s d^2)^{-1} \tag{86}$$

A.2.1.3 Mathematica model

Whitford et al. (2001) developed a version of the urban energy exchange model in the computer program Mathematica. The model was obtained from S. Gill and customised for use here. The model finds the solutions to Eqns. (77) and (78) using the following process:

1. The symbolic relationships between the parameters are defined and the parameters are given their numerical values.
2. The specific humidity is linearised around the reference temperature, as described in Eqns. (71) to (73) above.
3. The values of the a and b coefficients are determined.
4. The day is divided into three periods: sunrise to sunset; sunset to midnight; and midnight to sunrise. The ordinary differential equations are solved using an iterative process, starting from a reasonable guess for the sunrise temperatures.
5. The daytime differential equations are solved for sunrise to sunset using the sunrise temperatures as the initial conditions.
6. The nighttime differential equations are solved for the sunset to midnight period, using the numerical solutions from stage 5 as the initial conditions.
7. The nighttime differential equations are solved for the midnight to sunrise period, using the numerical solution from stage 6 as the initial conditions.
8. A convergence test is performed to test whether the new sunrise temperatures obtained from stage 7 are acceptably close to those used as the initial conditions for stage 5. If the difference in Celcius between the new and old temperature at level 0, plus the new and old temperature at level s, is greater than 0.001, stages 5-7 are repeated using the new sunrise temperature as the initial conditions. If the difference is less than 0.001 then the simultaneous equations are considered to be solved.
9. The model returns the maximum and minimum surface temperatures.

The original Mathematica script also plots the temperatures at levels 0 and s over the course of the day, but this function is not used in the present study.

A.2.2 Input data and parameters

Input parameters were customised from the values used by Gill (2006), who in turn adapted the model from Whitford et al. (2001) and Tso et al. (1991). Table 0.14 provides a summary of the parameter definitions and their sources, which are explained further below.

A.2.2.1 Meteorological parameters

The model estimates the maximum surface temperature reached for a given reference temperature T_f. Whitford et al. (2001) used the value for a specific day with typical hot British summer conditions to estimate the surface temperature under these

Table 0.14. Summary of parameters used in the urban energy exchange model. See text for further details of derivation and values of variable parameters.

Parameter	Definition	Units	Value	Original source
a_3	Peak insolation	W m^{-2}	802.5	CIBSE (1999)
a'_3	Nighttime radiation	W m^{-2}	-93	CIBSE (1982)
C_a	Specific heat of air	J kg^{-1} °C^{-1}	1005.9	Holman (1997)
C_c	Specific heat of building material	J kg^{-1} °C^{-1}	880	Holman (1997)
C_s	Specific heat of soil	J kg^{-1} °C^{-1}	Vary with soil type	Oke (1987)
$2d$	Soil depth	m	0.2	Assumption
E_f	Evaporative fraction	-	Vary with land cover	Land cover map
hoD	Hours of daylight	hrs	16	Gill (2006)
k	Von Kármán constant	-	0.41	Kantha and Clayson (2000)
k_s	Thermal conductivity of soil	W m^{-1} °C^{-1}	Vary with soil type	Oke (1987)
L	Latent heat of evaporation	J kg^{-1}	$2.434*10^6$	Oke (1987)
M_c	Equivalent homogeneous mass of building material	kg m^{-2}	Vary with land cover	Gill (2006)
q_2	Specific humidity at SBL	-	0.002	UK Met Office
T_2	Air temperature at SBL	° C	24.03	See text
T_b	Soil temperature at depth 2d	° C	17.47	UK Met Office
T_f	Reference temperature	° C	29.23	UK Met Office
U_2	Wind velocity at SBL	m s^{-1}	5	UK Met Office
Z_0	Roughness length	m	2	UK Met Office
Z_2	Height of the SBL	m	800	UK Met Office
ρ_a	Air density	kg m^{-3}	1.152	Oke (1987)

ρ_s	Soil density	kg m^{-3}	Vary with soil type	NATMAP data
ω	Parameter in equation	s^{-1}	pi/(HoD*3600)	N/A

typical conditions. Gill (2006) used the 98th percentile air temperature for the period 1961-1990, and the output of a climate change daily weather simulator, to estimate the change in surface temperature under extreme hot summer conditions given different climate change trajectories.

Extreme hot summer conditions were also of interest in the present study. Therefore T_f takes the value of the mean maximum temperature of the hottest day of each year from 1999-2008, which is 29.23°C. Air temperature data was obtained from the online MIDAS Land Surface Observation Stations dataset, for the weather station located in Weston Park in central Sheffield (UK Meteorological Office 2006). A few observations were missing from the dataset, but none during hot summer periods.

The density and specific heat of air (ρ_a and C_a respectively) and the latent heat of evaporation (L) are temperature-dependent, and therefore vary with T_f. The values of ρ_a and L used here were obtained using linear interpolation from known values as tabulated by Oke (1987). Tabulated values of C_a could not be found except as cited by Gill (2006), from Holman (1997); linear interpolation was again used. Interpolation of values is shown in Table 0.15.

The air temperature at the SBL, T_2, can be estimated by applying an environmental lapse rate (the decrease in temperature with altitude) to T_f (Gill 2006). Whitford et al. (2001), using Met Office data for the same typical hot British summer day, set the SBL to 800m; this value is also used here. The environmental lapse rate used by Gill (2006) is the long-term global average, which is 0.65°C per 100m through the troposphere. Therefore the value of T_2 used in the present study is:

$$T_2 = 29.23 - (8 * 0.65) = 24.03 \tag{87}$$

Other meteorological parameters at the SBL are the wind speed U_2 and the specific humidity q_2. The values used here (5 m s^{-1} and 0.002 respectively) are the same as

Table 0.15. Interpolation of model parameters dependent on the reference temperature.

Parameter	Tabulated value			Interpolation	Source
	20°C	25°C	30°C	29.23°C	
ρ_a (kg m^{-3})	-	1.168	1.149	1.168	Oke (1987)
L (J kg^{-1})	-	2.443	2.432	2.434	Oke (1987)
C_a (J kg^{-1} °C^{-1})	1005.6	-	1005.9	1005.9	Holman (1997)

ρ_s (water, kg m^{-3})	-	992.3	990.2	990.5	Oke (1987)

obtained by Whitford et al. (2001) for the typical hot British summer day, and also used by Gill (2006).

Peak insolation a_3 is set at the same value used by Gill (2006): 802.5 W m^{-2}. This value is cited from the CIBSE Environmental Design Guide (CIBSE 1999), and is the 97.5th percentile maximum normal to beam irradiance averaged over June 21st and July 4th annually from 1981-1992 for Manchester. The other parameters required to determine the daytime radiation are ω and hoD. Again, the same values are used as by Gill (2006). Manchester is relatively near to Sheffield, and at very similar latitude; therefore in the absence of other data sources these parameters should suffice.

Nighttime radiative loss a'_3 is set at -93 W m^{-2}, following Gill (2006). This value was calculated using an equation from the CIBSE Weather and Solar Data Guide for determining the net long-wave radiation loss from a black body at air temperature to the external environment for horizontal surfaces (CIBSE 1982). In evaluating this equation Gill (2006) assumed the sky to be cloudless.

In contrast to the air pollution reduction model (Section A.1.2.3), the urban energy exchange model assumes a constant roughness length Z_0 (Gill 2006). This approach was used in favour of assigned land cover specific values because of the necessity of implementing the model for squares within a 500m^2 grid, rather than for combinations of land cover/soil types (Section A.2.3). No guidance on assigning roughness lengths to areas of mixed land cover composition could be found, so a constant value was applied across the study area. The value used here is 2m, obtained by Whitford et al. (2001) from the UK Meteorological Office Engineering Science Data Unit item 72026 as the value for a city centre with some tall buildings, and also used by Gill (2006).

Finally, the value of the von Kármán constant k is 0.41, as obtained from Kantha and Clayson (2000).

A.2.2.2 Soil parameters

T_b is the soil temperature at soil depth $2d$. Although soil depth data is available in the NATMAP dataset, these values are not directly measured. In an urban situation, where soil may be removed during building and construction, soil depth may not reflect the NATMAP values. Furthermore, using the NATMAP soil depth estimates in model test runs was found to give unreasonable results. Therefore the value of 20cm was used, following the assumption made in previous studies implementing the model (Tso et al. 1991, Whitford et al. 2001, Gill 2006).

Soil temperature data was available from the MIDAS dataset for 30cm but not for 20cm (UK Meteorological Office 2006); in the absence of alternative data sources, these data were used. The average summer soil temperature for the period 1999-2008 was 17.47°C. Again, a small number of observations were missing from the dataset but there is no reason to assume a temperature bias in the missing data.

The soil density ρ_S varies with soil type. Oke (1987) lists values for dry and saturated sandy, clay and peat soils; Gill (2006) used the average of dry and saturated values of the appropriate soil types. For this parameter however, NATMAP estimates did not give unreasonable model results and were therefore used instead. Therefore the bulk density of soils to a depth of 20cm was obtained from NATMAP and used as the dry ρ_S. Saturated ρ_S was estimated by calculating the mass of water that would fit into the pore space (using NATMAP total porosity estimates) and adding this to the dry ρ_S. The average of dry and wet ρ_S was then determined.

NATMAP gives multiple bulk density and porosity estimates for up to four different land uses: arable, permanent grass, ley grassland and other. This categorisation is simplified for the present study by averaging arable, permanent grass and ley grassland into a single value, which is used where the majority of the soil type is under farmland or built land covers (as construction and farmland both alter the soil profile); and the 'other' category value is used for all other land covers. Finally the values for soil types were combined using the given proportions to produce a single ρ_S value for each NATMAP polygon, as listed in Table 0.16.

Estimates of the specific heat capacity and thermal conductivity of soils (C_S and k_S) are not given in NATMAP, therefore the average of the wet and dry values given by Oke (1987) for the appropriate soil texture is used. The appropriate soil texture is determined from the majority soil texture of the soil types making up individual polygons: peat soils are classified as such by NATMAP, and other soils were classified according to USDA rules (Soil Survey Division Staff 1993).

For the water land cover type, ρ_S is the temperature-dependent density of water; the interpolation of ρ_S from known values given by Oke (1987) is shown in Table 0.15. C_S and k_S of water as listed in Table 0.16 are also from Oke (1987).

Table 0.16. Soil type specific parameters used in the urban energy exchange model.

NATMAP soil ID	ρ_s (kg m^{-3})	C_s (J kg^{-1} °C^{-1})	k_s (W m^{-1} °C^{-1})
6 (water)	990	4.18	0.025
54106	1690	1.44	1.25
54107	1590	1.44	1.25
54125	1650	1.44	1.25
63101	970	1.44	1.25
65101	970	2.785	0.28
71201	1530	1.22	0.915
71301	1600	1.44	1.25
72103	570	2.785	0.28
81102	1500	1.22	0.915
101102	220	2.785	0.28

A.2.2.3 Built environment parameters

The area of built environment is multiplied by the building mass parameter M_c, which states the average building mass per unit area of building, in order to determine the total built mass. The land cover classes of buildings and manmade surfaces were considered separately in calculating M_c. Gill (2006), through various data sources and personal communication with another researcher, estimated the mass of a typical two-floor brick house in Manchester to be 842 kg m^{-2}, and the mass of residential roads to be 255.5 kg m^{-2}. The mass of other types of road was also estimated, but due to the inability to easily differentiate different types of roads in the present study, this single common road type is used. No estimates are given for other types of buildings; however two storey buildings are common and high-rise buildings also vary more in construction methods and therefore mass (Gill 2006), so the single value is used for all building types.

M_c is thus calculated as:

$$M_c = [\text{fractional road area}] \times 255.5 + [\text{fractional building area}] \times 842 \text{ kg m}^2 \quad (88)$$

However, the model produces an error if $M_c = 0$, so this parameter is given a minimum value of 0.001.

The specific heat of building material C_c is set to the value for concrete (880 J kg^{-1} °C^{-1}), and varies little with temperature (Oke 1987, Gill 2006). Following Gill (2006), this single value is used to represent all building materials as it is intermediate between the values for other common materials.

Finally, the evaporating fraction E_f is calculated from the land cover map as the fractional area that is not covered by unvegetated ground, building or manmade surfaces.

A.2.3 Model setup

The first attempt to implement the model was to produce a single maximum temperature value for each combination of soils and land cover types and to superimpose these values on the maps. However, this approach gave unreasonable results due to excessively high temperatures reached with low evaporating fractions for buildings, manmade surfaces and unvegetated land. Therefore the 500m^2 grid was imposed and the land cover and soil properties aggregated to this spatial scale, to produce a single maximum daytime temperature estimate for the extremely hot summer day scenario. A finer resolution grid was not feasible due to the computational effort involved. Area weighted means from the 500m^2 grid were used to calculate values for the other spatial units of analysis, i.e. Output Areas and Historic Environment Character polygons, again due to computational requirements.

A.2.3.1 Modelling the ecosystem service

Section 4.1.2 discusses the nature of the ecosystem service of heat island mitigation. In order to represent the effect from the ecosystem itself, which is the reduction in surface temperature compared to the temperature that would occur in the absence of natural land covers, the model was also run for each 500m^2 area for a land cover composition of 28% buildings and 72% manmade surfaces (as was found to be the ratio over the whole study area). The ecosystem service was then quantified as the difference between the temperature without natural land covers and the temperature given the actual land cover composition.

This value is the generation, or production of the ecosystem service. It can also represent the ecosystem service supply, because the benefits are provided to people at the site of production (Section 4.1.2).

A.2.3.2 Sensitivity tests

Sensitivity testing was carried out in order to explore the response of the urban energy exchange model to local changes in parameters, and therefore understand the possible consequences of low quality data. To produce a reference maximum surface temperature estimate, the average values for all varying parameters were computed over all grid squares and used as input to the model. Each input parameter was then in turn increased and decreased by 10% in order to determine the percentage change in the output maximum surface temperature.

The results of sensitivity testing are shown in Table 0.17. The rank order of changes is not vastly different whether parameter values are increased or decreased. The results also suggest that the model is relatively insensitive to changes in any one parameter, suggesting the output of the model is instead more sensitive to changes in groups of parameters. Such multi-parameter changes are likely to occur because meteorological parameters tend to form combinations characteristic to the climate and weather conditions of the area.

Table 0.17. Sensitivity test results for the urban energy exchange model. Values shown are the percentage change to maximum surface temperature estimate resulting from a plus or minus 10% change to the value of the input parameter. Parameters are shown in descending order of the averaged percentage change.

Parameter	+10%	-10%
k	5.08	6.73
E_f	3.19	3.66
a_3	3.04	3.04
L	2.75	3.18
U_2	2.65	3.21
T_2	2.50	2.50
ρ_a	2.65	1.01
T_f	2.01	1.37
Z_0	0.92	1.03
Z_2	0.93	1.01
q_2	0.50	0.50
T_b	0.27	0.27
k_s	0.09	0.10
d	0.06	0.06
C_a	0.05	0.05
ω	0.04	0.04
C_s	0.03	0.04
ρ_s	0.03	0.04
C_c	0.00	0.01

M_c	0.00	0.01
a'_3	0.00	0.00
hoD	0.00	0.00

The model is most sensitive to permutations in the value of k; the value of this constant is however well known so inaccuracies in the value used are unlikely (Kantha and Clayson 2000). The second most sensitive parameter is E_f, which is calculated directly from the land cover map, with a 3-4% change in the surface temperature estimate resulting from a 10% change in the parameter value.

a_3 is an average from multiple actual measurements over a period of time. However, it is highly variable from day to day, depending on the time of year and day and also on cloudiness. Nevertheless the percentage change from permutation is relatively small, and any changes in a_3 would be relatively uniform across the study area.

L and ρ_a are well known constants so sensitivity to these parameters is not important. The only remaining parameters with an average change in the output of greater than 1% are U_2, T_2 and T_f. Of these, T_f was deliberately chosen to represent a particular scenario. It is quite possible that the parameter estimates for U_2 and T_2 have low accuracy, as the former is a measurement for one particular day only, and the latter is calculated using a global environmental lapse rate.

A.3 Storm water runoff reduction model

The storm runoff model makes use of the USDA-NRCS Soil Conservancy Service's curve number (CN) method to estimate the proportion of rainfall in specified precipitation scenarios that is intercepted by soil and vegetation (USDA-NRCS 1986). This model is suited to urban hydrology (USDA-NRCS 1986), and has previously been used by Whitford et al. (2001) and Tratalos et al. (2007) as an indicator of the ecological or ecosystem "performance" of urban areas.

The inputs to the model are a land cover map, a soils map, two rainfall event scenarios, and empirically determined "curve numbers" (a description of how much precipitation is intercepted by different surfaces). The outputs are estimates of runoff for the two rainfall scenarios. The spatial resolution is the same as that of the land cover map (described in Section 2.2.1), and temporal aspects are not taken into consideration.

A.3.1 Model formulation

Stormwater runoff following a scenario precipitation event is calculated using the following two equations (USDA-NRCS 1986, Whitford et al. 2001):

$$Q = \begin{cases} \frac{(P - 0.2S)^2}{P + 0.8S}, & \text{if } P > 0.2S \\ 0, & \text{if } P \leq 0.2S \end{cases} \tag{89}$$

$$S = \frac{2540}{CN} - 25.4 \tag{90}$$

Where Q is the runoff depth (in cm), P is the precipitation (cm), S is the maximum potential retention once runoff begins, and CN is the curve number for a particular combination of land cover type, soil hydrological group (a categorisation of soil type based on infiltration rates, which is strongly related to soil texture) and antecedent soil moisture conditions (a categorisation of how dry or wet the soil is before the scenario precipitation event). The relationships between CN, P and Q are illustrated in Figure 5.3.

A.3.2 Input parameters and data

A.3.2.1 Scenarios

Two inputs to the curve number method - antecedent soil moisture condition and amount of rainfall - are specific to rainfall events(USDA-NRCS 1986). Therefore two scenarios were developed.

The ‘typical heavy rain’ scenario was designed to represent a relatively common event in the Sheffield area, with 1.2cm rainfall and normal (not dry but not saturated) antecedent moisture conditions (after Whitford et al. (2001)); these moisture conditions are designated as AMC II. In this scenario, bog moorland and water bodies were considered not to be saturated by these conditions, and therefore have zero runoff, i.e. absorb all precipitation.

The ‘extreme rain’ scenario is based on rare events such as the June 2007 rainfall that caused extensive fluvial and localised surface flooding. In this scenario, there is 6cm rainfall onto already-saturated soils (‘wet’ antecedent moisture conditions, or AMC III). Bog moorland and water bodies are assumed in this scenario to already be full to capacity, and so are treated as impervious surfaces.

A.3.2.2 Curve numbers

A USDA-NRCS technical report (USDA-NRCS 1986) lists curve numbers for various combinations of land cover and soil hydrologic groups (i.e. a category describing how freely water infiltrates a soil; designated by the letters A to D, with A being the most freely draining) for AMC II, from which the most appropriate were selected for the present analysis. Whitford et al. (2001), Tratalos et al. (2007) and examination of aerial photographs used for validation of the land cover classification were used as guidance in determining the most appropriate curve numbers. The land covers garden, unknown natural surface and unknown surface were calculated as mixed type land covers. Table 0.18 details the curve numbers used, and the corresponding land cover types in USDA-NRCS (1986).

USDA-NRCS (1986) lists curve numbers only for 'normal' antecedent soil moisture conditions (AMC II), i.e. neither dry nor saturated. Wet antecedent soil moisture condition (AMC III) curve numbers were calculated using conversion factors in Ward et al. (2003) (using linear interpolation between the listed factors).

Soil hydrologic group classification is based on infiltration rate, which is strongly related to soil texture. The groups range from group A, with "low runoff potential and high infiltration rates even when thoroughly wetted", to group D, with "very low infiltration rates when thoroughly wetted" (USDA-NRCS 1986). Table 0.19 shows the classification of soil textures into soil hydrologic groups.

Spatial soil texture information was obtained from the LANDis National Soil MAP (NATMAP) and the associated soil attribute data (SOILSERIES). The soils database is structured such that each NATMAP GIS polygon is composed of given percentages of one or more soil series, each of which has an associated attribute record. Where organic (O) soil horizons or peat were present as the surface layer, hydrologic soil group A was assigned. Where mineral soils formed the surface (A) horizon, the textural composition was used to assign a group (Table 0.19). One soil series had a H (humus) horizon over an A horizon, but this was so thin as to be considered insignificant (1.25cm). Where the soil series composing a polygon had different hydrologic groups, the compositionally dominant group was used. The curve numbers of buildings, manmade surfaces, water and bog moorland were not dependent on hydrologic group and were given a constant curve number regardless of NATMAP polygon soil type..

The NATMAP data adjacent to the edge of the Don Catchment (beyond which NATMAP data was unavailable) was consistently soil hydrologic group A, so this group was assigned for the area of missing soils data.

Table 0.18. Curve numbers used in the storm runoff model, and their sources in USDA-NRCS (1986). Numbers rounded to integers.

Land cover	AMC II				AMC III				USDA-NRCS land cover
	A	B	C	D	A	B	C	D	
1 Arable	67	78	85	89	83	90	94	96	Row crops – straight row treatment, management factors do not impair soil infiltration/increase runoff
2 Building, 6 Manmade surface	98	98	98	98	99	99	99	99	Impervious areas
3 Garden	60	75	82	86	79	88	93	95	Mixed land covers
4, 5 Grasslands	40	64	75	81	59	80	88	92	Average of pasture/grassland/range (50-75% ground cover, not heavily grazed) and meadow (continuous grass, ungrazed)
7 Moorland (bog)	0	0	0	0	99	99	99	99	Assumed to absorb all precipitation in AMC II, and to be saturated (treated as impervious) in AMC III
8 Moorland (heath)	30	48	65	73	50	68	82	87	Brush (brush-weed-grass mix with brush the major element) with >75% ground cover
9 Scrubland	35	56	70	77	55	75	85	89	Brush (brush-weed-grass mix with brush the major element) with 50-75% ground cover - as is often mixed with other covers
10 Unknown natural surface	56	72	81	85	76	86	92	94	Mixed land covers
11 Unknown surface	54	72	81	85	74	86	91	94	Mixed land covers
12 Unvegetated land	72	82	87	89	86	92	95	96	Urban dirt
13 Water	0	0	0	0	99	99	99	99	Assumed to absorb all precipitation with AMC II, and to be full (treated as impervious) in AMC III
14, 15 Woodlands	30	55	70	77	0	74	85	89	Woods – protected from grazing, litter and brush adequately cover soil

A.3.3 Model setup

For each scenario, Q and the total runoff volume per m^2 were computed for all relevant combinations of land cover, soil hydrologic group and antecedent soil moisture

Table 0.19. Classification of soil textures into soil hydrologic groups. Taken from USDA-NRCS (1986).

Group	Texture
A	Sand, loamy sand, sandy loam
B	Silt loam, loam
C	Sandy clay loam
D	Clay loam, silty clay loam, sandy clay, silty clay, clay

conditions. These values were then spatially assigned based on the land cover map and hydrologic soil group.

A.3.3.1 Modelling the ecosystem service

The model described above was used to estimate runoff for the two scenarios, using the land cover and soils maps. To investigate the reduction in runoff made by natural surfaces, a baseline was also calculated by running the model with a constant curve number of 98 or 99.37 (for normal and wet antecedent soil moisture conditions respectively), which is the curve number used for impervious surfaces. This baseline was subtracted from the estimated runoff to calculate the ecosystem service.

Area weighted means were used to find the average reduction of runoff due to natural surfaces for each unit of analysis (i.e. 500m grid squares, Output Areas and Historic Environment Character areas separately) for each scenario. The production of the ecosystem service was finally quantified as the mean of the values for the two scenarios for each spatial unit of analysis.

A.4 Carbon storage model

The carbon storage model uses land cover based estimates of carbon biomass in vegetation, and estimates of soil organic carbon contents from the NATMAP data, to estimate the total carbon stored at each location.

A.4.1 Model formulation

The approach to modelling carbon storage in vegetation is to assign land cover specific, area based estimates to the land cover map. In order to determine the total storage in any area of interest, an area weighted mean estimate can then be calculated (McGarigal et al. 2002):

$$\text{storage per area} = \sum_{j=1}^{n}\left[x_{ij}\left(\frac{a_{ij}}{\sum_{j=1}^{n} a_{ij}}\right)\right] \tag{91}$$

Where the subscript x_{ij} is the carbon storage value for all land cover map polygons i of a land cover category j, and a_{ij} is the area, such that $\frac{a_{ij}}{\sum_{j=1}^{n} a_{ij}}$ is the proportional abundance of a land cover category.

Carbon storage in soils, or specifically organic carbon (estimates of inorganic carbon were not available) is estimated using a similar approach, using soil and horizon specific estimates of organic carbon by weight and bulk density to calculate the carbon per unit area of a soil map polygon:

$$\text{carbon storage (g m}^{-2}\text{)} = \sum_{j=1}^{n}\left[\sum_{i=1}^{n}\left(\frac{C_{ij}}{100} B_{ij} H_{ij} \frac{S_j}{100} \times 10^4\right)\right] \tag{92}$$

Where C is the percent of carbon by weight in horizon i of a soil series j (there are multiple soil series within any one soil map polygon), B is the bulk density of the soil series in g cm^{-3}, H is the soil horizon depth in cm, S is the percentage of the soil map polygon composed of the soil series and 10^4 converts the estimate from 1 cm^2 to 1 m^2. An area weighted mean of the polygon estimates for a specific area of interest can then be made using Eqn. (91), where the subscripts are the soil map polygons i of a particular ID j.

A total estimate of carbon storage is made by adding the vegetation and soil estimates.

A.4.2 Input data and parameters

A.4.2.1 Vegetation

Vegetation carbon storage is calculated from land cover, using values specified by Cruickshank et al. (2000) for estimation of vegetation carbon storage in Ireland derived from satellite imagery land cover mapping. Table 0.20 shows the carbon storage values for each land cover category, and their equivalent categories in Cruickshank et al.

(2000). Ireland has a similar climate and patterns of land cover to England, meaning that these estimates should also be valid for Sheffield.

Table 0.20. Estimates of carbon storage in vegetation, and equivalent categories in the data source (Cruickshank et al. 2000)

	Land cover category	Carbon storage ($g\ m^{-2}$)	Cruickshank et al. (2000) category
1	Arable	220	Non-irrigated arable land
2	Building	0	N/A
3	Garden	352	(Mixed type land cover)
4	Grassland (improved)	90	Pastures
5	Grassland (rough)	150	Natural grassland
6	Manmade surface	0	N/A
7	Moorland (bog)	200	Peat bogs – unexploited
8	Moorland (heath)	200	Moors and heathland
9	Scrubland	1450	Transitional woodland-scrub
10	Unknown natural surfaces	670	(Mixed type land cover)
11	Unknown surfaces	246	(Mixed type land cover)
12	Unvegetated land	0	N/A
13	Water	0	N/A
14	Woodland (coniferous)	2990	Coniferous forest
15	Woodland (non-coniferous and mixed)	3540	Average of broad-leaved forest and mixed forest

A.4.2.2 Soil

Estimates of the carbon content of soils are derived from the NATMAP soils map and data. Each land cover category used in this study was matched to one of the four NATMAP land use types as shown in Table 0.21. In general, the arable land use type had lower carbon content than the other land uses.

Land under buildings and manmade surfaces is likely to be heavily disturbed, and consequently to have lost much of its carbon (Pouyat et al. 2006). Estimates for Boston, which according to the USDA-NRCS global soils map has the same broad soil type as Sheffield (USDA-NRCS 2005), suggest that soils under impervious surfaces have around half the organic carbon density of those under agricultural land use (Pouyat et al. 2006). In this model, buildings and manmade surfaces are therefore assigned one half of the carbon estimate for the arable land use. Gardens are given the higher value of permanent grass because studies suggest that relatively large amounts of soil carbon can accumulate in gardens (Golubiewski 2006, Pouyat et al. 2006).

Table 0.22 shows a summary of the soil carbon estimates for each soil mapping unit, along with a brief description of the soil. NATMAP polygons lying outside the Don Catchment were assigned the soil mapping unit of the closest NATMAP polygon.

Table 0.21. Assignment of land cover categories to NATMAP land use types for the purposes of soil carbon content estimation.

	Land cover category	**NATMAP land use type**
1	Arable	Arable
2	Building	Given ½ the value for arable
3	Garden	Permanent grass
4	Grassland (improved)	Ley grass (often found in farmland so likely field rotation)
5	Grassland (rough)	Permanent grass
6	Manmade surface	Given ½ the value for arable
7	Moorland (bog)	Other
8	Moorland (heath)	Other
9	Scrubland	Other
10	Unknown natural surfaces	Arable (conservative estimate)
11	Unknown surfaces	Arable (conservative estimate)
12	Unvegetated land	Arable (probably recently disturbed)
13	Water	N/A
14	Woodland (coniferous)	Other
15	Woodland (non-coniferous and mixed)	Other

Where estimates were not given for a required land use – mapping unit combination (indicated by a dash in Table 0.22) the lowest available estimate was used (there is no situation where, for example, an arable estimate is available where a land use with a higher estimate is not). Polygons identified by either the land cover map or by NATMAP as containing water were given a value of 0, because NATMAP does not describe properties for soils below water bodies.

A.4.3 Model setup

The areal carbon content of soils and vegetation were assigned to the soil type and land cover maps respectively. These values were then aggregated to each unit of analysis (500m grid squares, Output Areas and Historic Environment Character areas) using area-weighted means (Eqn. (91)), and the area-weighted means summed to calculate the total areal carbon content.

A.4.3.1 Modelling the ecosystem service

The ecosystem service is the storage of carbon over and above what would be stored in the absence of non-artificial land covers. Therefore the carbon storage assuming only artificial land covers was calculated (i.e. zero storage in vegetation, and half the lowest

Table 0.22. Carbon content estimate (g m^{-2}) of NATMAP soil mapping units. A dash indicates that data are not given for the land use type.

Mapping unit	Arable	Ley grass	Other	Permanent grass
54106	10435	14120	14884	15381
54107	-	13742	14910	15990
54125	12570	17140	22042	18057
63101	-	-	31783	25729
65101	-	-	24197	-
71201	15983	18674	17511	19897
71301	14061	17172	18077	19965
72103	-	-	44493	41436
81102	22163	25041	24900	26844
101102	-	-	170472	168390

available NATMAP estimate in soils) and subtracted from the actual carbon storage estimate in order to quantify the ecosystem service.

A.5 Habitat for flora and fauna index

The index of habitat for flora and fauna is calculated from three metrics that assess different aspects of landscape land cover composition that are relevant to biodiversity. This approach is similar to that used by Whitford et al. (2001), but with an additional metric to represent the shape of patches of natural land cover, as well as patch size and diversity of habitat types.

A.5.1 Index formulation

A.5.1.1 Proportion natural land cover

The species-area relationship is well known in ecology and also applies in urban areas: for example, Clergeau et al. (1998) found that the extent of urban greenspace correlates with the species richness of birds. Therefore the first metric is simply the proportion of the area of interest that is covered by natural land covers, i.e. including 83% and 89.67% of the unknown and unknown natural covers respectively (the proportions of natural land cover found to be in these land cover classes during validation of the land cover map; see Section 2.2.1.4) and 67% of gardens (Tratalos et al. 2007), and excluding manmade surfaces and buildings.

A.5.1.2 *Shannon diversity index*

As different species prefer different types of habitat, the total number of different types of land cover in an area can be broadly expected to correlate with species richness. The number of habitat types does not, however, contain any information about the distribution of area between habitat types. This is important because, all else being equal, habitats with a small total area will contribute less to biodiversity than those with a large total area due to the species-area relationship.

The Shannon diversity index, which is most commonly used as an index of species diversity, is determined by both the richness of types (e.g. species, habitats) and distribution of individuals (organisms, patches) amongst types. The Shannon diversity index H' is calculated as follows (Whitford et al. 2001):

$$H' = \sum_{i=1}^{k} (p_i \log_2 p_i) \tag{93}$$

Where p_i are the proportions of each habitat type and k is the number of habitat types present in the area of interest. Unknown surfaces and unknown natural surfaces were excluded from the calculation of H'; however, manmade surfaces and buildings were included as a single habitat type, because some species thrive in close association with humans (McKinney 2006). In order to have metrics with the same numerical range, H' was then scaled by the maximum possible value that would exist if all possible land covers were present in exactly equal proportions.

The value of the Shannon diversity index is standardised to have a maximum of 1, where a value of 1 would indicate exactly equal proportions of each of the twelve land cover types included in the calculation.

A.5.1.3 Natural land cover correlation lengths

The previous metrics describe habitat type composition at the levels of the landscape and habitat type. By contrast, the correlation length, also known as the area-weighted mean radius of gyration, is used to represent the connectivity of individual patches of natural land cover through their average size. Connectivity is another aspect of the species-area relationship, and is important because habitat fragmentation is thought to be an important cause of the loss of some species from urban areas (Bolger et al. 2000, Crooks 2002).

The radius of gyration is sensitive to shape as well as area: for two patches of equal area, the less circular will have a higher radius of gyration. In the present context, it can be interpreted as a measure of the average distance in which a mobile organism confined to natural land cover can travel in an area of interest, from a random starting point, before reaching intraversible land.

The radius of gyration G of each area of natural land cover (any except buildings, manmade surfaces, unknown surfaces – but not unknown natural surfaces – and water) is calculated on a cell-by-cell basis within each patch of natural land cover (see Section A.5.2) as follows:

$$G_i = \sum_{j=1}^{z} \frac{h_{ij}}{z} \tag{94}$$

Where h_{ij} is the distance (m) between cell j located within patch i and the centroid of patch i, and z is the number of cells within patch i. G is in units of metres. The area-weighted mean $\bar{G}$ of all natural land cover patches within the area of interest is then:

$$\bar{G} = \sum_{i=1}^{n} \left[G_i \left(\frac{a_i}{\sum_{i=1}^{n} a_i} \right) \right]$$

Where a_i is the area of patch i and n is the total number of patches. To account for differences in sizes of areas of interest, $\bar{G}$ is then scaled to the maximum possible value of $\bar{G}$ if that area contained only natural land cover. However, if the area of interest contains patches of highly irregular shape, or is itself irregularly shaped, the land cover rasterisation methods used (see Section A.5.2) mean it is occasionally possible for the scaled value to be greater than one; in these cases, a value of exactly one is assigned.

A.5.1.4 Combining the metrics

These three metrics each give different information about the amount and/or composition of the natural land cover in an area of interest. There is not sufficient empirical evidence about the relative importance of the three metrics to develop relative weightings for combining the metrics into an index, so they are assumed to exert equal effects on overall biodiversity potential. The most parsimonious case that effects are additive, rather than multiplicative, is also assumed. Therefore the overall metric of biodiversity potential is the sum of proportion of natural land cover, the standardised Shannon diversity index and the correlation length.

A.5.2 Metric calculation

The FRAGSTATS software (McGarigal et al. 2002) was used in the computation of the biodiversity potential metrics. FRAGSTATS analyses spatial patterns in categorical raster maps, such as land cover maps, and calculates metrics at three spatial scales: patch (for each individual group of raster cells of the same category), class (for all patches within one category) and landscape (for all patches in the whole map).

In the present study a *landscape* was considered to be a single area of interest (i.e. a single 500m grid square, Output Area or Historic Environment Character area). The landscape rasters were generated from the vector land cover map with a cell size of 1m. Therefore some very narrow areas on the vector map are not captured in the rasters.

The *classes* were the numerical codes for the land cover classes, although modified legends were applied or codes excluded from analysis as required for the particular metric. The percentage natural land cover and Shannon diversity index were both calculated using output from FRAGSTATS function number C3 (total areas of each class within a landscape), while the correlation length was directly calculated by FRAGSTATS using function C18.

For a very small number of landscapes (one 500m^2 grid square, two HEC polygons and four OAs) FRAGSTATS produces errors of unknown origin, although the cause may be the extremely small (in one case) or large (in six cases) size of the polygons. In order that these polygons could be included in analysis, alternative methods were used to estimate values for the metrics.

The 500m^2 grid square (ID 473) is entirely moorland, so was assigned a proportion natural cover of one, a Shannon diversity index of zero, and a standardised correlation length of one.

The four OA polygons (IDs 740, 1575, 1595 and 1599) are large farmland and moorland areas. The proportion natural land cover and Shannon diversity index were calculated by intersecting the OA and land cover vector layers and performing calculations on the areas of the resulting polygons. The standardised correlation length was estimated by comparison with polygons with a similar land cover pattern on a smaller scale.

The two HEC polygons (IDs 1364 and 1387), which are also large moorland areas, produced errors in FRAGSTATS for the correlation length only. This metric was therefore imputed using the same method as for the OAs.

Appendix B. Urban morphology metrics

B.1 Introduction

Chapter 10 is based upon analyses of the relationship between ecosystem service production and urban morphology. This appendix describes the metrics used to represent urban morphology. The metrics are first in Section 355, and Section B.3 decribes how the metrics were calculated.

This appendix also contains two analyses. In order to ensure that no pair of metrics were redundant, i.e. that all metrics describe fairly independent components of urban morphology, a correlation analysis was undertaken to determine the strength of the relationships between metrics. In order to determine the types of analyses necessary in Chapter 10 it was also essential to establish whether the chosen metrics vary with broad land use type, i.e. whether land use type is confounded with urban morphology. The methods for these two analyses are described in Section B.3. Section B.4 presents the maps of the metrics and also the results of the analyses.

A total of seven metrics of urban morphology were chosen to describe some of the facets of urban morphology. The metrics were chosen according to theoretical expectations as described in Section B.2 and/or have been used in previous studies.

B.2 Metric definition

B.2.1 Proportion of impervious cover

The proportion of impervious cover (buildings and manmade surfaces) in an area is the simplest measure of the amount of development. The higher the level of development, the less ecosystem service-providing greenspace will be present. This metric is used following Tratalos et al. (2007), who investigated relationships between measures of biodiversity/ecosystem services and urban morphology.

This metric was calculated as the sum of the areas of manmade surfaces, buildings, and the average proportional manmade surface/building cover for garden, unknown surface and unknown natural surface land cover types. This proportion is 33% for gardens, as determined by a previous study of land covers in Sheffield (Tratalos et al. 2007); and 17% for unknown surfaces and 10.33% for unknown natural surfaces, as determined in the land cover map validation exercise performed in Chapter 2 (see Table 2.4).

B.2.2 Building density

The building density, i.e. the number of buildings scaled by the size of the area of interest, provides information about the ‘amount’ of building in an area. A “building” is defined here as an area contained within contiguous external walls, rather than as an address, because the latter information was not available at the Historic Environment Character area scale (although at Output Area scale, the two measures were found to be highly correlated). This metric was also used following Tratalos et al. (2007). It should be noted that the method used to count the number of buildings had a spatial resolution of one metre, and thus small gaps between buildings may not have been ‘seen’ (see Section B.3).

B.2.3 Mean building size

As the proportion of impervious cover and the proportion of cover by buildings are not necessarily correlated, and building of different types differ considerably in size (for example, consider residential areas in comparison to industrial estates), the previous two metrics do not provide information about the ‘amount’ of building in terms of how much ground it actually covers. This is important for the ecosystem services that are affected differently by buildings and manmade cover, e.g. heat island mitigation.

The mean size of buildings, which is equivalent to the proportion of building cover divided by the building density, is used to give this information; this is used instead of the proportion of building cover because it is likely to be more meaningful in terms of urban planning. The method used to identify individual buildings for the building density metric was also used here.

B.2.4 Impervious surface normalised landscape shape index (nLSI)

The normalised landscape shape index (nLSI) is the sole metric describing a facet of the layout of the built environment. The spatial layout of the built environment is an important component of urban morphology as far as environmental issues are concerned because it determines the nature of the matrix of habitat patches available to organisms and, therefore, ecosystem processes (Garden et al. 2006, McKinney 2008). For example, a single large patch of habitat in an area might support viably large populations of species with poor migration and dispersal abilities that would become locally extinct if the same area of habitat was available as a number of small discrete patches (Garden et al. 2006, McKinney 2008). On the other hand, a large number of smaller patches may in other circumstances mean that ecosystem service production is more spread out, and, depending on the nature of the service, more accessible (Handley et al. 2003). It follows, then, that an obvious characteristic of the spatial layout to identify for analysis is variation in the area-perimeter relationship.

The normalised landscape shape index is used to quantify this relationship. The nLSI describes the extent to which the layout of an area diverges from the most compact shape, i.e. it is a measure of land cover fragmentation/landscape complexity (Patton 1975, McGarigal et al. 2002). The metric does not appear to have been used previously in a study such as this.

The index compares the total perimeter of impervious surface patches in an urban area to the minimum possible perimeter for the same area of land if the impervious surfaces were all aggregated into a single square. This number is then standardised by class area to range between zero and one, where zero is a maximally compact shape (which in this case is a square, rather than a circle, due to edge length being determined from discrete square raster cells) and one is maximal disaggregation.

The index is calculated from a rasterised version of the land cover map with a 1m^2 resolution using the FRAGSTATS software (see Section A.5.2). The index is calculated as (McGarigal et al. 2002):

$$nLSI = \frac{e - e_{min}}{e_{max} - e_{min}} \tag{95}$$

Where e is the length of edge of impervious surface in the area of interest, and e_{min} and e_{max} are the minimum and maximum possible edge lengths determined from maximimally aggregated and disaggregated shapes. e_{min} and e_{max} are determined as follows (McGarigal et al. 2002):

$$e_{min} = \begin{cases} 4n, & \text{if } a - n^2 = 0 \\ 4n + 2, & \text{if } n^2 < a \leq n(1 + n) \\ 4n + 4, & \text{if } a > n(1 + n) \end{cases} \tag{96}$$

Where a is the impervious area and n is the side of the largest integer square small than a, that is $\lfloor\sqrt{a}\rfloor$. All units are in terms of raster cells.

$$e_{max} = \begin{cases} 4a, & \text{if } P \leq 0.5 \\ 3A - 2a, & \text{if } A \text{ is odd and } 0.5 < P \leq \dfrac{0.5A + 0.5B}{A} \\ 3A - 2a + 3, & \text{if } A \text{ is even and } 0.5 < P \leq \dfrac{0.5A + 0.5}{A} \\ Z + 4(A - a), & \text{if } P > \dfrac{0.5A + 0.5B}{A} \end{cases} \tag{97}$$

Where A is the size of the area of interest, B is the number of raster cells in the boundary of the area of interest, Z is the length of the area boundary in terms of cell surfaces, and P is the proportion of the area comprised of impervious surfaces. Again, all units are in terms of raster cells.

B.2.5 Population density

Population density is an important consideration in urban morphology (Burton 2000, Tratalos et al. 2007). Although a densely populated area may contain less greenspace due to a greater amount of housing, it may also mean that in comparison to a low density area there are more large contiguous areas of greenspace elsewhere. Therefore population density is included as an urban morphology metric, following Tratalos et al. (2007).

B.2.6 Proportion of households in detached and semi-detached houses (proportion detached houses)

Population density may be related to the type of housing present (e.g. detached, terraces, flat blocks); however, the same type of house may be occupied by different numbers of people. In addition, housing type has a large influence over neighbourhood design; and there is a relationship between house type and footprint area, such that detached and semi-detached houses on average occupy a greater area of land than other house types (Environment Agency 2004). This aspect of space efficiency is also included here as a metric, again following Tratalos et al. (2007), represented in a simple numerical value by the proportion of households that occupy detached or semi-detached houses (shortened to “proportion detached houses”).

Both this metric and the population density are especially meaningful in the context of the residential-only analysis, as they both relate to the resident population. However, in the full dataset and industrial-only analyses they can be interpreted as indicating the importance of patterns of urban morphology typically found near particular types of residential areas (low/high population density, small/large proportion of households in detached/semi-detached houses).

B.2.7 Weighted road length density index (road LDI)

The links in an area to the transportation network are an important component of built environment design and, therefore, urban morphology; and indeed transportation is a key issue in research into urban morphology (Williams et al. 2000c). Although the thrust of much of this research is at the scale of whole cities and in terms of decreasing reliance on motorised transport (e.g. see Williams et al. 2000a), variation in the design of the transport network between different parts of a single city is also relevant in terms of accessibility of neighbourhoods and environmental disturbances caused by traffic (Volchenkov and Blanchard 2007, Bouchard et al. 2009).

A crude representation of the design of the transportation network is the total length of road. This type of metric does not appear to have been used previously in a study similar to this. (The number of road junctions was also considered as a metric of transportation network complexity, but was found to be strongly correlated with road length.) Initial testing of this metric suggested that weighting roads by type gave a better representation of how well connected areas are to the main transport system. Therefore local streets were given a value of 1; minor roads 2; B roads 3; A roads 4; and motorways 5. Alleys and private roads were given a value of 0, as these were considered not to contribute to the main road network and are likely to have low traffic flows. The weighted length of road was then scaled in order to account for the size of the area under consideration, yielding a metric in units of (weighted length) ha^{-1}. This metric is referred to in the thesis as the road LDI.

B.3 Methods

B.3.1 Metric calculation

The FRAGSTATS software introduced in Section A.5.2 was used to compute the proportion of impervious cover, mean building size, building size and impervious surface nLSI. The proportion of impervious cover and nLSI were computed using a simplified land cover legend, in which buildings and manmade surfaces were given the same identifying code. The proportion was derived from the output of FRAGSTATS function number C3 (total areas of each class within a landscape), and the nLSI calculated directly with function number C124 (nLSI of a class). FRAGSTATS encountered an error when calculating the nLSI for the two largest moorland polygons (FID numbers 2364 and 2387). These polygons were therefore manually assigned a value of 0.1, which is close to the value of polygons with a similar land cover layout.

Mean building size and building density were computed using the full, non-simplified land cover legend. Mean building size was the output of function number C11 (mean area of patches of a class); and building density was the number of patches of the building class (function C5) divided by the sum of the class areas (function C3). To reiterate, it should be noted that, due to the raster cell size of 1m, buildings separated by a gap of this distance or less will be seen by the raster as contiguous.

The population density and proportion of detached houses were taken from the UK 2001 Census statistics at the Output Area level; 2001 is the most recent year for which census data were available. Area-weighted means were used to spatially resolve the data to Historic Environment Character areas.

The road length density was determined from Ordnance Survey MasterMap Integrated Transport Network GIS data, which represents roads as lines. Weights were assigned to the lines using the road type data records in MasterMap. The total weighted length of lines in each Historic Environment Character area polygon was summed using Hawth's Analysis tool 'Sum Line Lengths in Polygons'; finally, this value was divided by the area of the polygon.

B.3.2 Metric correlation analysis

The degree to which urban morphology metrics correlate with each other is important to confirm that the metrics are not measuring the same thing. As non-normal frequency distributions are present, Spearman's rank correlation coefficient, rho, is used to quantify the association between each pair of metrics. The methods used to calculate Spearman's rho were described in Section 9.3.2, and the approach to statistical inference in Section 9.2. As a total of 21 correlations were performed, a Bonferroni-corrected significance threshold of $\frac{0.001}{21} = 4.67 * 10^{-5}$ is applied here and compared to the results if the correction is not applied (the uncorrected threshold is 0.001, rather than 0.05, to account for spatial autocorrelation; see Section 9.2.1).

B.3.3 Variation of metrics with land use

To investigate how urban morphology metrics vary with broad land use type, and therefore the extent to which land use is a confounding factor in ecosystem service–urban morphology analysis, Kruskal-Wallis rank analyses of variance were performed for each urban morphology metric. The methods used were the same are those described in Section 10.2.5. The results of all Kruskal-Wallis tests were significant at the Bonferroni-corrected threshold, so Mann-Whitney U tests were necessary in all cases to identify the land uses between which significant differences lay.

B.4 Results

B.4.1 Metric patterns and correlations across all land use types

Maps of the metrics are shown in Appendix D as the following maps:

- Proportion of impervious cover - Map 29
- Building density - Map 30
- Mean building size - Map 31
- Impervious surface nLSI - Map 32
- Population density - Map 33
- Proportion detached houses - Map 34
- Road LDI - Map 35

Table shows the results of the correlations between metrics.

The proportion of impervious cover (Map 29) obviously repeats the overview seen in the land cover map (Map 2), and also strongly reflects patterns of land use (Map 3). There are very high levels of impervious surface cover in the urban core, where industrial and commercial land uses are common, decreasing to moderate cover in the primarily residential areas and very low levels in the agricultural and moorland areas. This pattern is much as expected; the other metrics, however, reveal more complex and subtle relationships.

The map of building densities (Map 30) shows that, within the developed area, the building densities is in general lower in the city centre and increases towards the fringes and in the more remote developed patches. This pattern is largely explained by

Table B.1. Spearman's *rho* rank correlation coefficient matrix for urban morphology metrics. Darker greens and reds indicate stronger positive and negative correlations respectively. Grey background indicates results not significant at the Bonferroni-corrected significance threshold but significant at the non-corrected level; white background indicates results not significant at either level.

	Proportion impervious cover	Building density	Mean building size	Impervious surface nLSI	Population density	% detached houses	Road LDI
Proportion impervious cover	-						
Building density	0.54	-					
Mean building size	0.57	0.02	-				
Impervious surface nLSI	-0.02	0.10	-0.30	-			
Population density	0.30	0.54	0.00	0.11	-		
Proportion detached houses	-0.51	-0.05	-0.40	0.07	-0.17	-	
Road LDI	0.56	0.44	0.20	-0.04	0.25	-0.26	-

variation in the mean size of buildings (Map 31), which is large in the commercial and industrial areas but lower in the surrounding residential areas. These two metrics nevertheless are not correlated ($rho = 0.02$; Table) because in the agricultural and rural areas of Sheffield there is a fairly uniformly low building density and a small to medium building size.

The impervious surface nLSI shows very little consistent pattern across the study area (Map 32). Both very high and very low values are observed in the city centre, where industrial, commercial, communications and extractive land uses occur. This suggests diversity in the design of such estates, in terms of the distribution of impervious surfaces as multiple patches versus a single large patch, perhaps to the exclusion of pervious land covers entirely. Interestingly, there are few city centre polygons with medium values of this index. There is also much variation in values in the other parts of the study area, with perhaps a slight tendency to higher values in the residential urban area than rural areas, although there are fewer extreme values. The only other metric with which the index has a moderately strong correlation is the mean building size. This is a negative correlation ($rho = -0.3$), i.e. larger building size correlates with more aggregation in impervious surfaces. This relationship may be largely driven by residential areas, which have small buildings interspersed with gardens and small patches of natural land covers; and commercial/industrial estates, with typically larger buildings and less natural land cover (Map 29).

The population density (Map 33) is, as might be expected, quite strongly correlated with building density ($rho = 0.54$). The most obvious difference between these metrics is that the population density is low in industrial and commercial regions. Other variation can probably be explained in terms of building design: towards the city centre, apartment buildings are common and thus the population density is high for the building density. Further out, there are terraces, and then semi-detached and detached houses, which result in a higher building density but relatively lower population density. This pattern is confirmed by the map of the proportion detached houses (Map 34), which indicates an increasing proportion from the city centre outwards. (The patchy, clustered distribution of values that is apparent in Map 33 and Map 34 results from the re-aggregation of Output Area scale census data to Historic Environment Character areas.)

The road LDI (Map 35) also shows some fairly strong correlations, with the proportion of impervious cover and building density ($rho = 0.56$ and $rho = 0.44$ respectively). The map clearly delineates the major roads through the city. In the urbanised area, most polygons have quite high values, although there are also some with low values that correspond to land uses such as parks.

In summary, these seven metrics all appear to be measuring fairly independent components of urban morphology: no pair has a correlation coefficient greater than 0.57. The metrics show clearly different patterns across Sheffield, and seem to show different values both within and between land use types. Therefore all of these metrics will be included in further analyses.

B.4.2 Metric correlations within residential and industrial land uses

The correlations between urban morphology metrics for residential and industrial land use types are shown in Table. The patterns of correlations are quite different between the land use types, and different also to those across all land use types (Table). In the case of residential areas, most of the correlations are statistically significant regardless of the application of a Bonferroni correction; whereas many more correlations are weaker and non-significant within the industrial land use. Although this is likely to be partially due to the smaller sample size for industrial land use, it also suggests that the aspects of urban morphology described by these metrics constrain each other, or force trade-offs, to a lesser degree than in residential areas. Importantly, Table confirms that, despite a small number of stronger relationships than observed across all land uses, the metrics are still measuring quite different facets of urban morphology.

A notable difference between the overall and residential correlations is that, within the residential land use, there is a very strong negative correlation between building density and mean building size (Table). This correlation does not exist when all land use types are analysed together, presumably because, in contrast to highly developed land use types, in less developed land uses constructing more buildings does not necessarily mean making them smaller. Indeed, this correlation is also present within the industrial land use, although to a lesser degree.

Table B.2. Spearman's *rho* rank correlation coefficient matrix for urban morphology metrics in residential and industrial land uses. Darker greens and reds indicate stronger positive and negative correlations respectively. A grey background indicates results not significant at the Bonferroni-corrected significance threshold but significant at the non-corrected level; a white background indicates results not significant at either level.

		Proportion impervious cover	**Building density**	**Mean building size**	**Impervious cover nLSI**	**Population density**	**% detached houses**	**Road LDI**
Residential	Proportion impervious cover	-						
	Building density	0.18	-					
	Mean building size	0.35	-0.71	-				
	Impervious cover nLSI	-0.22	0.27	-0.35	-			
	Population density	0.43	0.11	0.18	0.05	-		
	Proportion detached houses	-0.46	0.36	-0.56	0.22	-0.37	-	
	Road LDI	0.45	-0.01	0.14	-0.22	0.07	-0.20	-
Industrial	Proportion impervious cover	-						
	Building density	0.31	-					
	Mean building size	0.47	-0.35	-				
	Impervious cover nLSI	0.65	0.36	0.00	-			
	Population density	0.02	0.23	-0.15	0.11	-		
	Proportion detached houses	-0.31	-0.16	-0.08	-0.13	-0.05	-	
	Road LDI	0.16	0.13	0.03	0.13	0.13	-0.15	-

For industrial areas there is also a strikingly strong positive correlation between the impervious surface nLSI and the proportion of impervious cover, which indicates that, where industrial sites have a high coverage of buildings and manmade surfaces, these land cover types are also comparatively disaggregated. In contrast, residential areas have a negative and weaker correlation between these metrics: a high impervious coverage correlates with a more compact shape of the impervious surfaces. It is possible that this reflects a tendency towards block designs in more intensively developed residential areas, but towards sprawl in intensively developed industrial areas.

In general, the population density, proportion of detached houses, and road LDI show few correlations with each other or with other metrics in industrial areas. The former two of these metrics are based on Output Area level statistics, so the poor correlations here simply suggest that industrial land use occurs in areas with a range of types of nearby residential areas. In the case of the road LDI, the weak correlations probably arise from industrial estates having a range of transport infrastructure designs that are dependent upon specific needs (e.g. access needs for large manufacturing companies compared to small craft industries).

B.4.3 Variation of metrics with land use

Figure shows that the average rank of urban morphology metric values varies considerably between land use types. It is also apparent that the metrics vary in different ways between land use types, providing further evidence that they are each measuring different aspects of urban morphology; although some pairs of metrics covary more than others.

Pairwise comparison tests, the results of which are shown in Table , confirm that in many cases the differences between land uses reach a high degree of statistical significance. The proportion of impervious cover is far higher for land uses that involve extensive building (commercial; industrial; communications; residential; institutional) than for those that do not (woodland; enclosed and unenclosed land; extractive; water bodies; ornamental, parkland and recreational; horticultural is also included in this category according to its mean rank, although the sample size is too small to test statistical significance) (Figure , Table a). Nevertheless, there are also many significant differences between land uses within these groupings, indicating that land use strongly influences the proportion of impervious cover between different types of

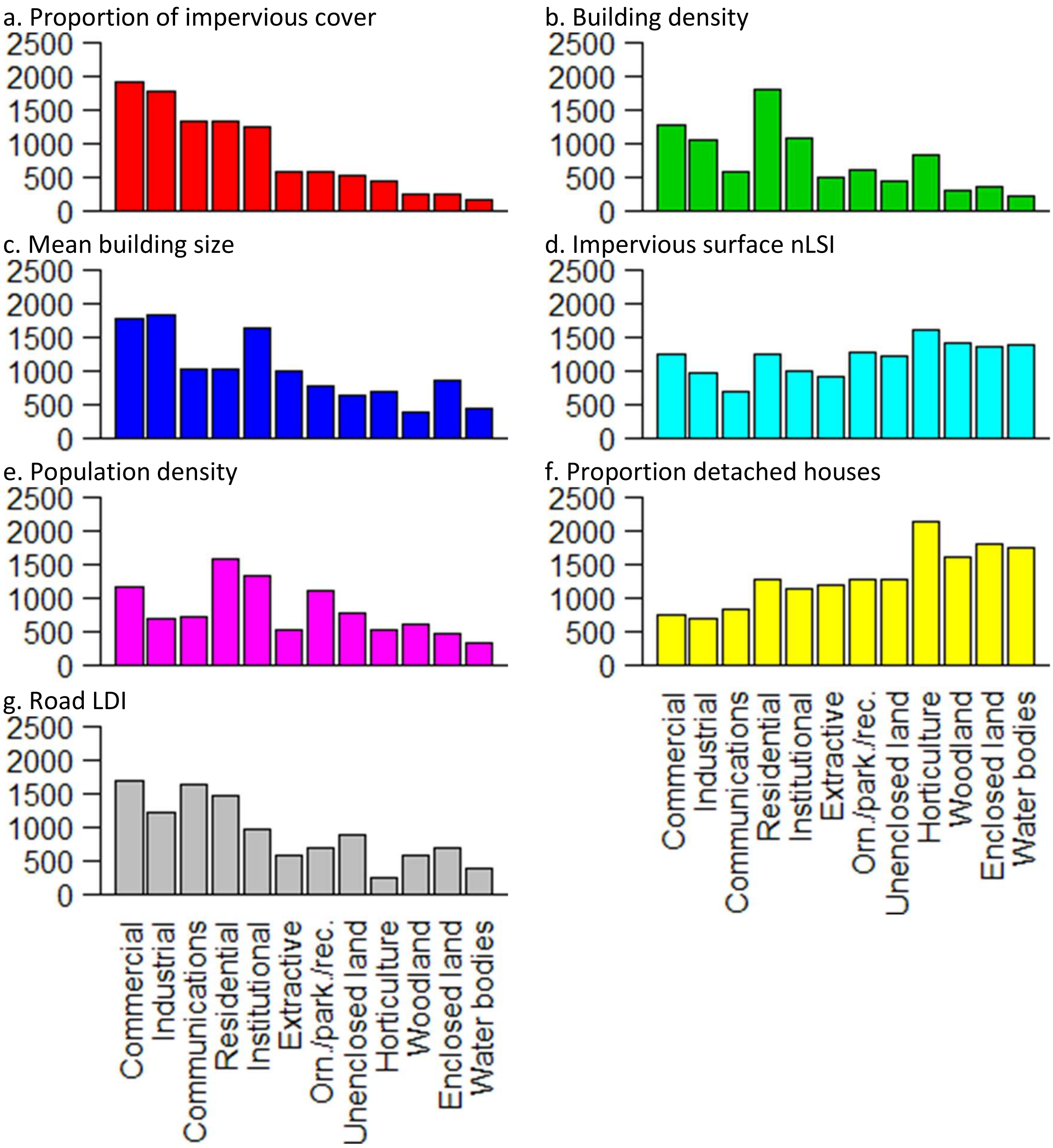


Figure B.1. Average rank of urban morphology metric values across different land use types. High ranks indicate high metric values. Land use types ordered by decreasing proportion impervious cover.

both intensive and non-intensive land use types. For example, commercial and industrial areas tend to have more impervious cover than residential and institutional areas, probably because the latter have more gardens and common areas.

This grouping of land uses into more and less intensive also holds for several other metrics, namely mean building size (larger in intensive land uses; Table c) and road LDI (higher in intensive land uses; Table g), although in both cases the boundary between the groups is less sharply defined and the differences between less intensive land uses tend not to be significant. The average rank order of land uses

Table B.3. Mean ranks and pairwise multiple comparisons of urban morphology metric values between different land use types. Numbers in second column show the average rank of metric values for the land use type in the first column (where high ranks are high production). In the subsequent columns in the cells under the dashes, a black fill indicates where the land use type in that column has significantly greater metric values than the land use type in that row at the Bonferroni-corrected significance threshold; grey indicates a difference significant at the non-corrected threshold; and white indicates no significant difference. (Multiple comparison test: Mann-Whitney U-test.) In (e), N/As arise due to tied average ranks.

a. Proportion of impervious cover

		Commercial	Industrial	Communications	Residential	Institutional	Extractive	Orn./park./rec.	Unenclosed land	Horticulture	Woodland	Enclosed land	Water bodies
Commercial	*1912*	-											
Industrial	*1779*		-										
Communications	*1330*			-									
Residential	*1328*				-								
Institutional	*1250*					-							
Extractive	*567*						-						
Orn./park./rec.	*565*							-					
Unenclosed land	*521*								-				
Horticulture	*434*									-			
Woodland	*245*										-		
Enclosed land	*244*											-	
Water bodies	*157*												-

within the groupings also differs between all three of these metrics, indicating that the constraints on urban morphology imposed by development differ between land uses along more than one gradient.

The building density (Table b) is highest by far in the residential land use, with commercial, institutional and industrial also having relatively high density (Figure). With the exception of horticulture, which has a very small sample size, the other land uses all on average have far lower building densities. Communications land use also tends to have a low building density, suggesting that the impervious cover here mostly comprises manmade surfaces, rather than buildings (the mean building size is also smallest of the intensive land uses).

The population density is obviously highest in residential areas and the land use types that tend to be closely associated with residential land use, i.e. institutional,

Table B.3 continued.

b. Building density

		Residential	Commercial	Institutional	Industrial	Horticulture	Orn./park./rec.	Communications	Extractive	Unenclosed land	Enclosed land	Woodland	Water bodies
Residential	*1796*	-											
Commercial	*1257*		-										
Institutional	*1072*			-									
Industrial	*1035*				-								
Horticulture	*808*					-							
Orn./park./rec.	*587*						-						
Communications	*563*							-					
Extractive	*479*								-				
Unenclosed land	*430*									-			
Enclosed land	*336*										-		
Woodland	*302*											-	
Water bodies	*212*												-

c. Mean building size

		Industrial	Commercial	Institutional	Residential	Communications	Extractive	Enclosed land	Orn./park./rec.	Horticulture	Unenclosed land	Water bodies	Woodland
Industrial	*1824*	-											
Commercial	*1775*		-										
Institutional	*1624*			-									
Residential	*1019*				-								
Communications	*1011*					-							
Extractive	*987*						-						
Enclosed land	*852*							-					
Orn./park./rec.	*777*								-				
Horticulture	*691*									-			
Unenclosed land	*616*										-		
Water bodies	*434*											-	
Woodland	*384*												-

commercial, and ornamental/parkland/recreational (Table e, Figure). Population density is lower for land uses that tend to be found further away from residential areas, explaining why industrial and communications land uses have similar levels of this metric to woodland, unenclosed land etc. In contrast, the proportion of

Table B.3 continued.

d. Impervious surface nLSI

		Horticulture	Woodland	Water bodies	Enclosed land	Orn./park./rec.	Commercial	Residential	Unenclosed land	Institutional	Industrial	Extractive	Communications
Horticulture	*1608*	-											
Woodland	*1399*		-										
Water bodies	*1379*			-									
Enclosed land	*1351*				-								
Orn./park./rec.	*1277*					-							
Commercial	*1236*						-						
Residential	*1232*							-					
Unenclosed land	*1205*								-				
Institutional	*981*									-			
Industrial	*958*										-		
Extractive	*910*											-	
Communications	*693*												-

e. Population density

		Residential	Institutional	Commercial	Orn./park./rec.	Unenclosed land	Communications	Industrial	Woodland	Extractive	Horticulture	Enclosed land	Water bodies
Residential	*1571*	-											
Institutional	*1323*		-										
Commercial	*1169*			-									
Orn./park./rec.	*1105*				-								
Unenclosed land	*778*					-							
Communications	*722*						-						
Industrial	*678*							-					
Woodland	*588*								-				
Extractive	*517*									-			
Horticulture	*517*										-		
Enclosed land	*456*											-	
Water bodies	*322*												-

detached houses is in general significantly lower in industrial, commercial and communications land uses (Table f), indicating that these land uses tend to be found in association with higher density residential land use subtypes. On the other

Table B.3 continued.

f. Proportion detached houses

		Horticulture	Enclosed land	Water bodies	Woodland	Orn./park./rec.	Residential	Unenclosed land	Extractive	Institutional	Communications	Commercial	Industrial
Horticulture	*2135*	-											
Enclosed land	*1783*		-										
Water bodies	*1739*			-									
Woodland	*1608*				-								
Orn./park./rec.	*1265*					-							
Residential	*1261*						-						
Unenclosed land	*1254*							-					
Extractive	*1187*								-				
Institutional	*1129*									-			
Communications	*812*										-		
Commercial	*727*											-	
Industrial	*690*												-

g. Road LDI

		Commercial	Communications	Residential	Industrial	Institutional	Unenclosed land	Orn./park./rec.	Enclosed land	Extractive	Woodland	Water bodies	Horticulture
Commercial	*1684*	-											
Communications	*1642*		-										
Residential	*1471*			-									
Industrial	*1204*				-								
Institutional	*953*					-							
Unenclosed land	*868*						-						
Orn./park./rec.	*692*							-					
Enclosed land	*677*								-				
Extractive	*581*									-			
Woodland	*572*										-		
Water bodies	*387*											-	
Horticulture	*249*												-

hand, enclosed land, water bodies and woodland are associated with low density residential land use subtypes, i.e. detached and semi-detached houses.

In comparison with the other metrics, there are few significant differences between land uses for the impervious surface nLSI (Table d). Communications land uses have the lowest metric value, with extractive, industrial and institutional uses also having quite low values, indicating the impervious surfaces in these areas tend to be compact relative to other land use types.

The comparison of urban morphology metric values for different land uses performed in this section has shown that, to varying extents, all of the metrics vary with land use. This needs to be considered in the design and interpretation the analyses of ecosystem service production-urban morphology relationships in Chapter 10.

Appendix C. Socioeconomic datasets

C.1 Introduction

Chapter 11 focuses on analysis of the relationships between socioeconomic conditions and ecosystem service production in Sheffield. The datasets used to quantify and classify socioeconomic conditions are introduced in Section 11.2.1.

This appendix contains more information about the production of the two datasets used to represent socioeconomic conditions. These census statistics are produced at Output Area scale, and no manipulation was necessary in order to spatially assign the data.

C.2 Social grade approximation algorithm

The algorithm used to approximate social grade from census data operates as follows (Market Research Society, no date):

1. Full-time and part-time workers are assigned an approximated grade or combination of grades based on the census occupational classification, employment status and size of work establishment.
2. For workers assigned a combination of grades in stage 1, apply coding rules derived from decision tree analysis. Coding rules are based on employment status, qualifications, size of establishment, sex, working status and tenure.
3. For non-workers, and workers who could not be assigned a grade/grade combination in stage 1, apply coding rules derived from another decision tree analysis. Coding rules are based on employment status, qualifications, size of establishment, sex, working status and tenure.
4. For persons aged 75+, apply a grade based solely on tenure instead of the above rules.

Comparison of classification by this algorithm with classification by interview suggests that the algorithm provides good approximation for adults aged 16-64 (Market

Research Society, no date). However, accuracy is far lower for adults aged 65-74 because the majority of households with people of this age do not have a chief income earner and are automatically classified by the algorithm as grade E, regardless of pensions (Market Research Society, no date). Accuracy is also quite low for adults aged 75+ (Market Research Society, no date).

An alternative dataset is available which avoids this problem, namely Output Area-level approximated social grade based only on the workplace population aged 16-74. However, this dataset was not used here because it excludes all unemployed and younger state welfare dependent individuals, i.e. all individuals coded as grade E. As grade E indicates the poorest socioeconomic status of these grades, it is important that this group is included in the present analysis.

C.3 Area classification variables and method

The area classification of Output Areas was generated by selecting a subset of census variables and using a clustering algorithm to define groups of Output Areas with similar characteristics. Full technical details of the methods used to generate the area classification of Output Areas can be found in Vickers and Rees (2007); what follows is a description of the input variables and a summary of the clustering process.

The 41 variables selected for use in the area classification of Output Areas, shown in Table , represent all five domains of data collected in the census: demographic structure, households composition, housing, socio-economic group and employment (Vickers and Rees 2007). Variables were selected by disregarding variables that were highly correlated, had very skewed distributions, were not collected consistently across the UK, are known not to be precise, show uninteresting geographic distributions, or which were likely to lose their meaning before the 2011 Census; and by producing new composite variables from multiple original variables where appropriate (Vickers and Rees 2007).

A *k*-means clustering algorithm was used recursively to produce the hierarchical classification from the variables shown in Table , of which full details are given by Vickers and Rees (2007). *k*-means clustering is a method that partitions multivariate datasets into a pre-set number of groups, in such a way that the total error sum of

Table C.1. Variables used in the cluster analysis that generated the area classification of Output Areas, by variable domain. From Vickers and Rees (2007).

Variable	**Definition**
Demographic (9 variables)	
Age 0–4	% of resident population aged 0–4 years
Age 5–14	% of resident population aged 5–14 years
Age 25–44	% of resident population aged 25–44 years
Age 45–64	% of resident population aged 45–64 years
Age 65+	% of resident population aged 65 or more years
Indian, Pakistani or Bangladeshi	% of people identifying as Indian, Pakistani or Bangladeshi
Black African, Black Caribbean or Other Black	% of people identifying as black African, black Caribbean or other black
Born outside the UK	% of people not born in the UK
Population density	population density (the number of people per hectare)
Household composition (6 variables)	
Separated/divorced	% of residents 16 years old or older who are not living in a couple and are separated or divorced
Single person household (not pensioner)	% of households with one person who is not a pensioner
Single pensioner household	% of households which are single-pensioner households
Lone parent household	% of households which are lone parent households with dependent children
Two adults no children	% of households which are cohabiting or married couple households with no children
Households with non-dependant children	% of households comprising one family and no others with non-dependent children living with their parents
Housing (8 variables)	
Rent (public)	% of households that are public sector rented accommodation
Rent (private)	% of households that are private or other rented accommodation
Terraced housing	% of all household spaces which are terraced
Detached housing	% of all household spaces which are detached
All flats	% of households which are flats
No central heating	% of occupied household spaces without central heating
Average house size	average house size (rooms per household)
People per room	average number of people per room

squares, given by the divergence of data points from the group means, is minimised. This method was used to provide a three-level categorical scheme with seven supergroups, 21 groups and 52 subgroups. The subgroup level was not used in the present analysis due to many small sample sizes; the group level was determined to have a more suitable balance between Output Area differentiation and sample size for the purpose of the analysis in Chapter 11.

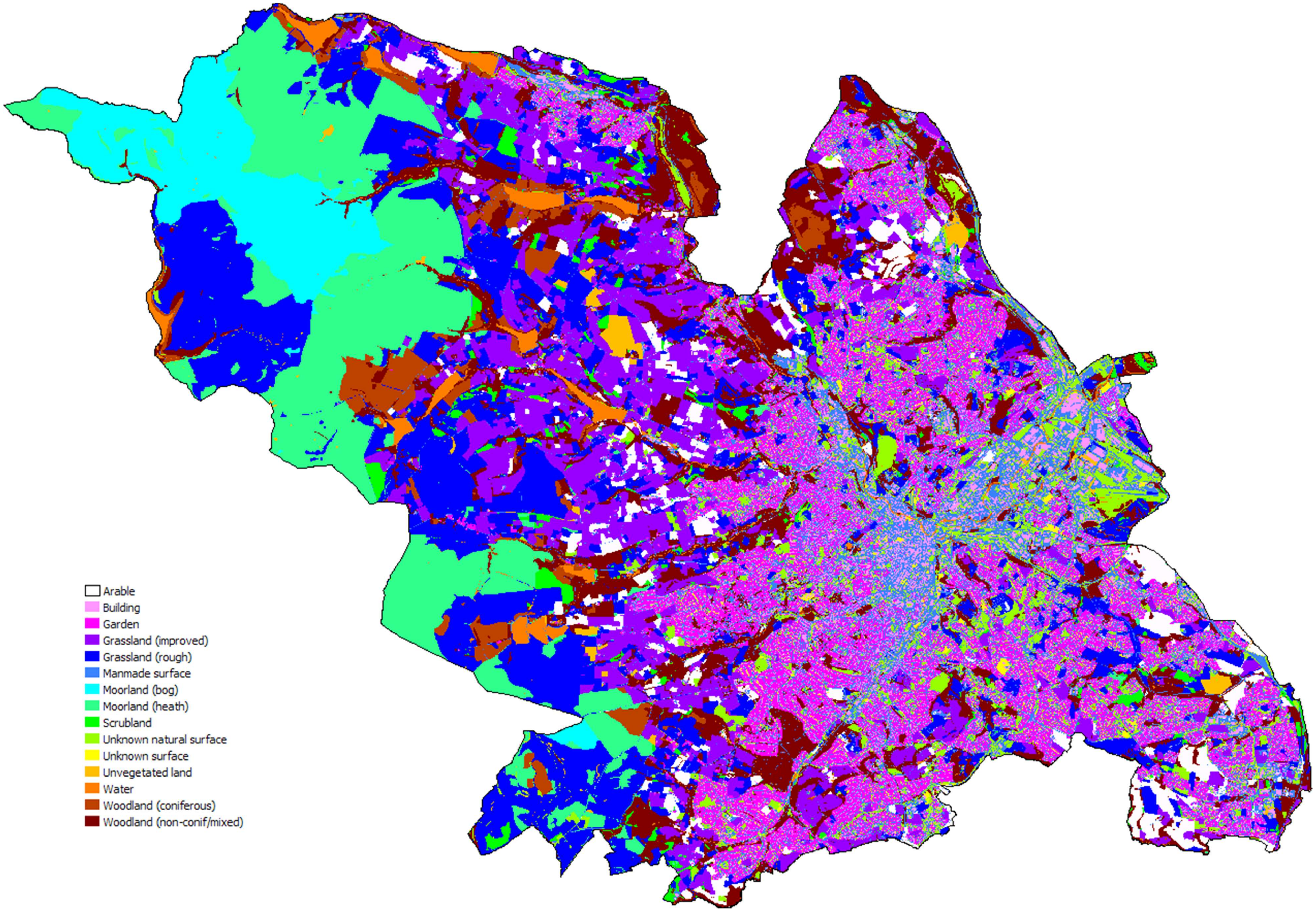
Table C.1 continued.

Variable	Definition
Socio-economic (7 variables)	
HE qualification	% of people aged between 16 and 74 years with a higher education qualification
Routine/semi-routine occupation	% of people aged 16–74 years in employment working in routine or semiroutine occupations
2+ car household	% of households with 2 or more cars
Public transport to work	% of people aged 16–74 years in employment who usually travel to work by public transport
Work from home	% of people aged 16–74 years in employment who work mainly from home
LLTI (SIR)	% of people who reported suffering from a limiting long-term illness (standardized illness ratio, standardized by age)
Provide unpaid care	% of people who provide unpaid care
Employment (11 variables)	
Students (full-time)	% of people aged 16–74 years who are students
Unemployed	% of economically active people aged 16–74 years who are unemployed
Working part-time	% of economically active people aged 16–74 years who work part time
Economically inactive looking after family	% of economically inactive people aged 16–74 years who are looking after the home
Agriculture/fishing employment	% of all people aged 16–74 years in employment working in agriculture and fishing
Mining/quarrying/ construction employment	% of all people aged 16–74 years in employment working in mining, quarrying and construction
Manufacturing employment	% of all people aged 16–74 years in employment working in manufacturing
Hotel and catering employment	% of all people aged 16–74 years in employment working in hotel and catering
Health and social work employment	% of all people aged 16–74 years in employment working in health and social work
Financial intermediation employment	% of all people aged 16–74 years in employment working in financial intermediation
Wholesale/retail trade employment	% of all people aged 16–74 years in employment working in the wholesale or retail trade

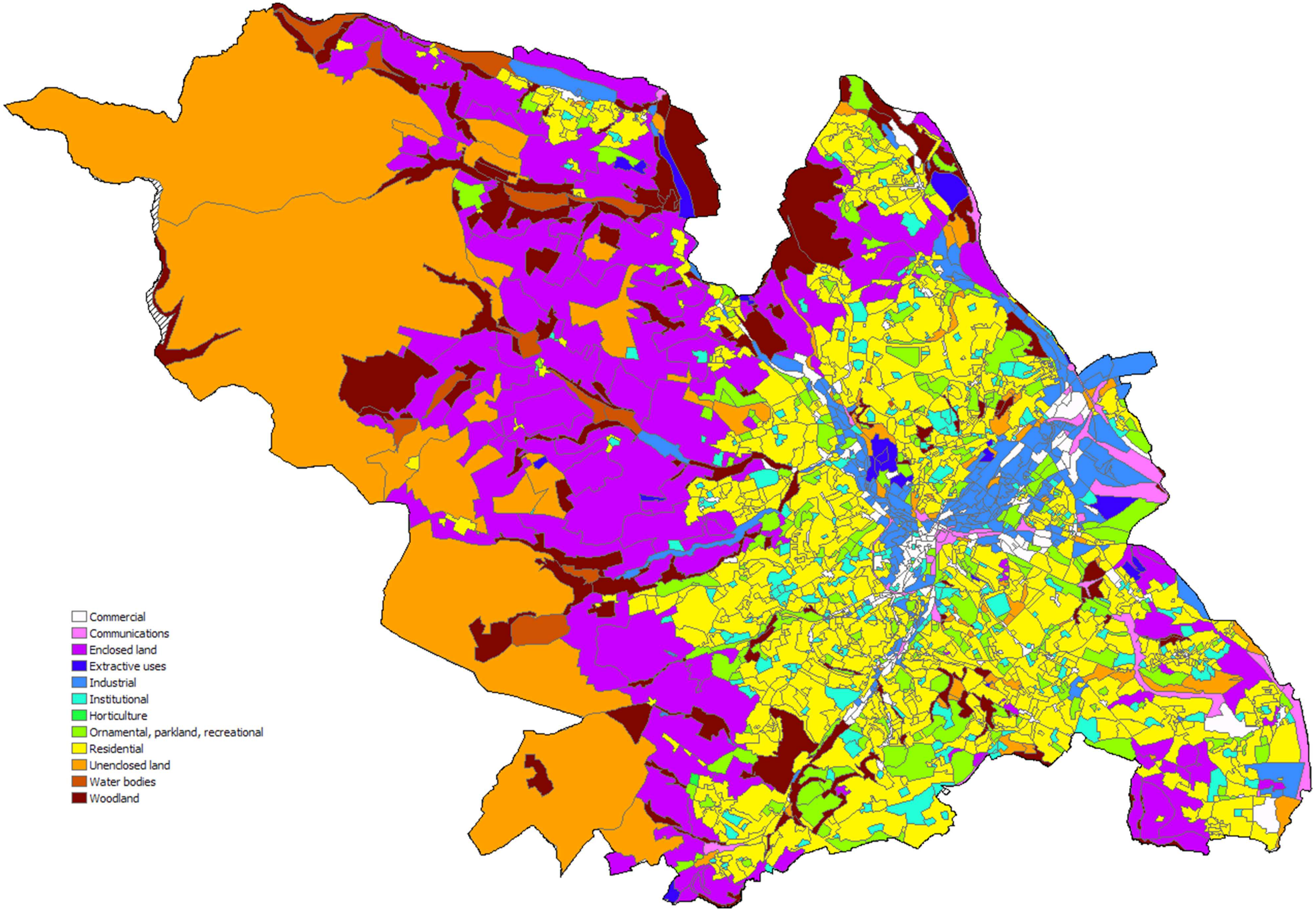
Each category was analysed in order to determine which patterns of combinations of variable values make that category unique – in order words, to establish the category's demographic profile (Vickers and Rees 2007). This profiling was achieved by identifying the variables for which the category's mean value is significantly different from the national mean.

Appendix D. Map appendix

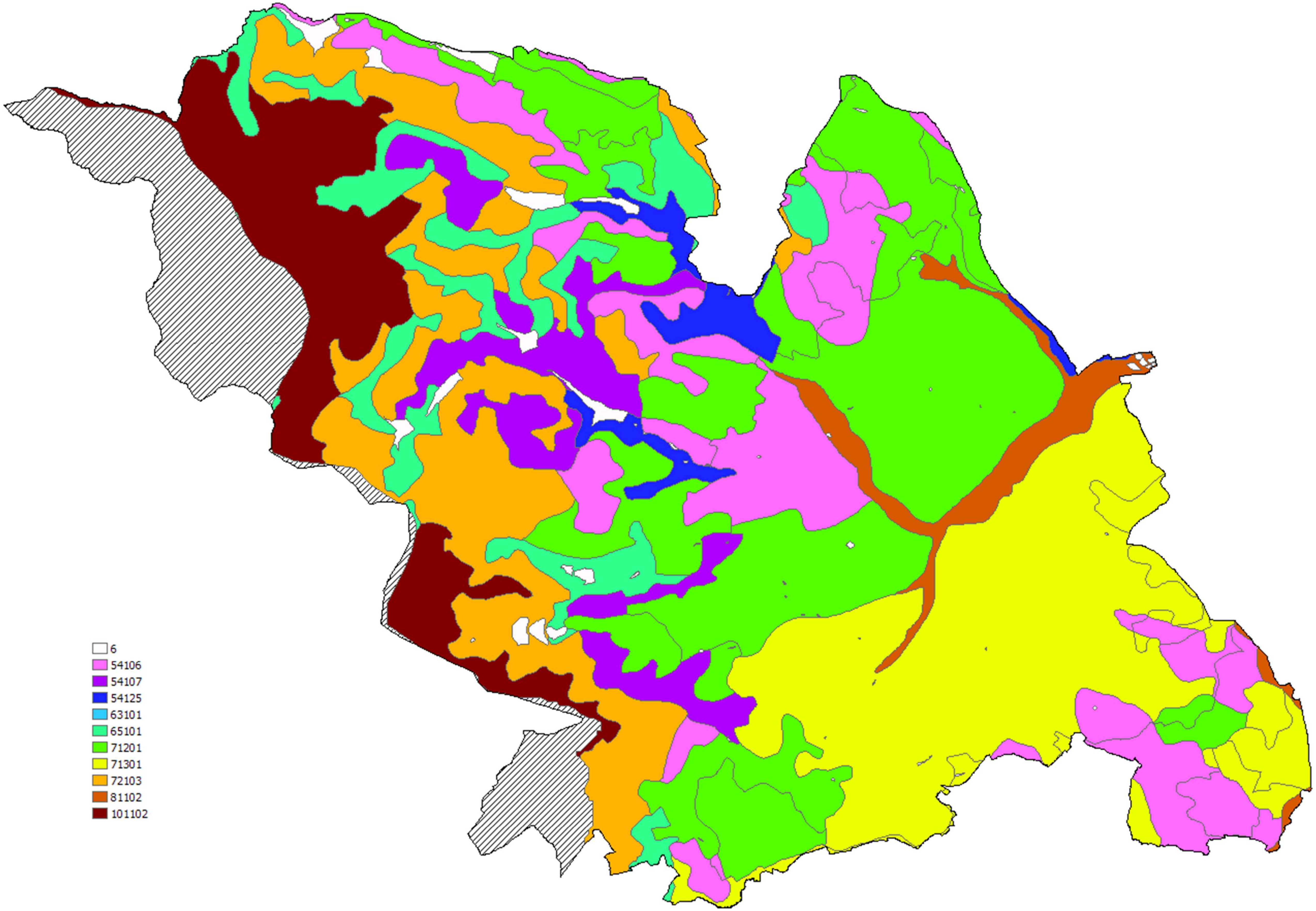
Map 1. Administrative wards of Sheffield.



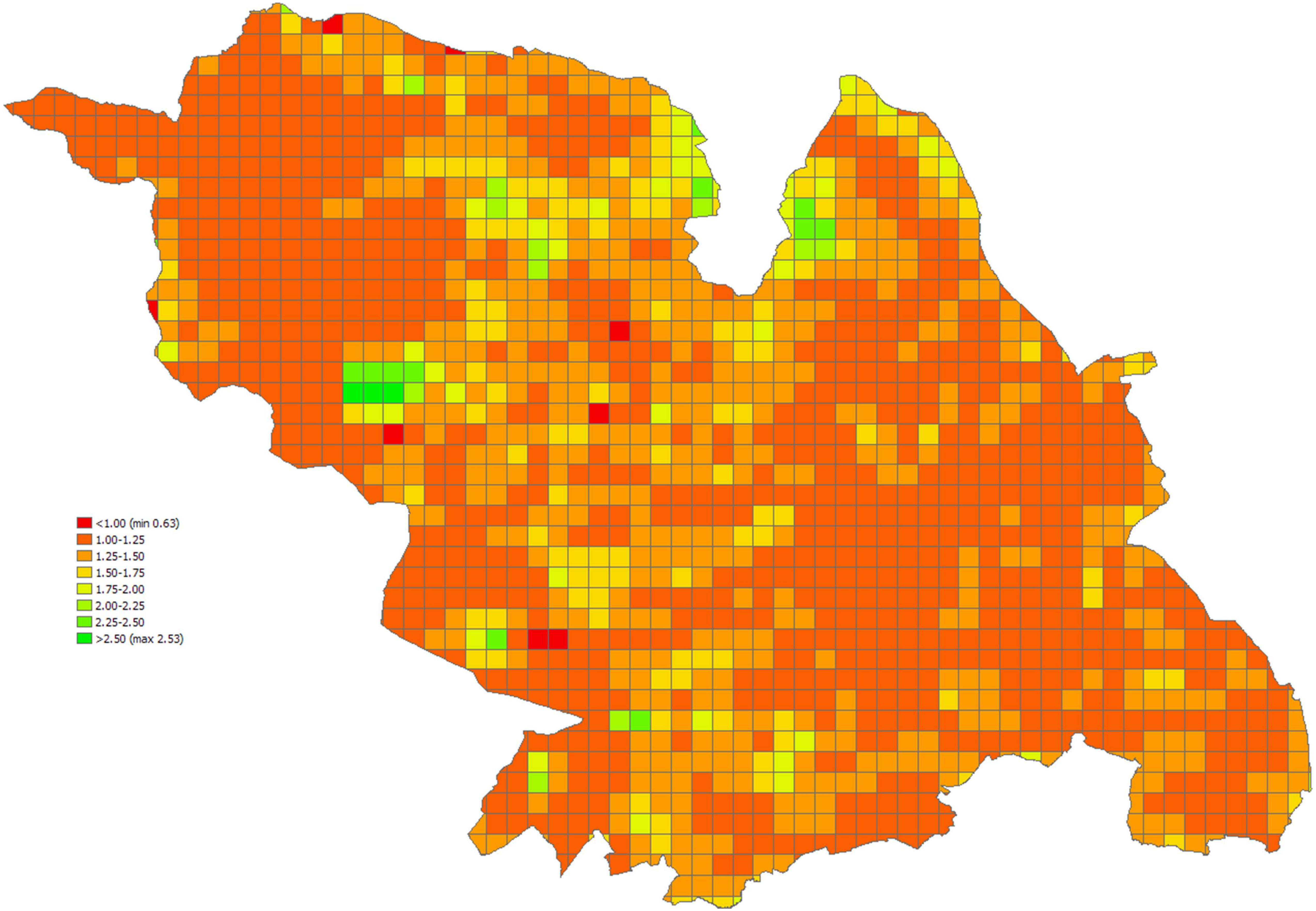
Map 2. Land cover map.



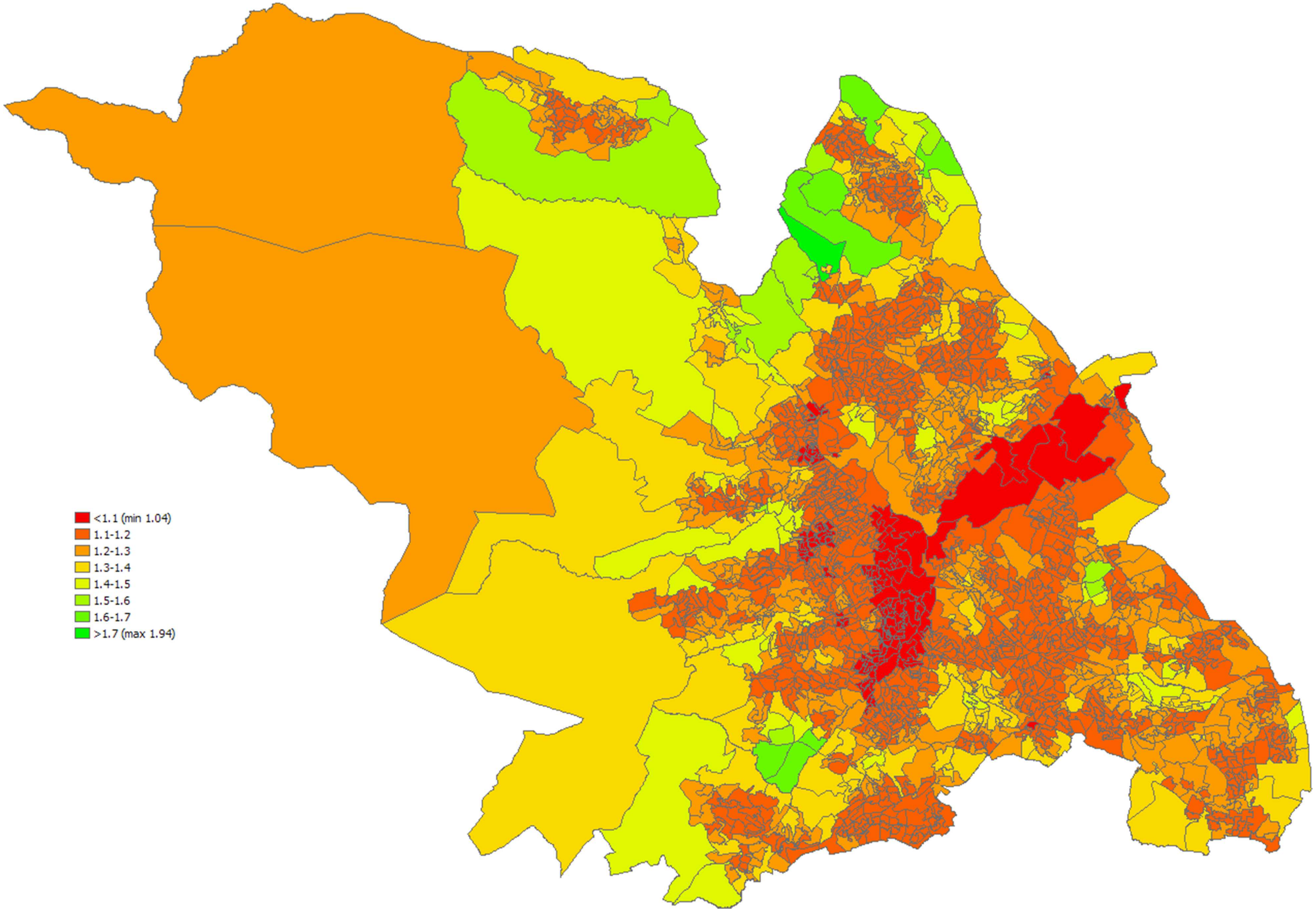
Map 3. Sheffield Historic Environment Character GIS dataset, showing broad character types. Hatched areas are not covered by the dataset, and are both part of a reservoir.



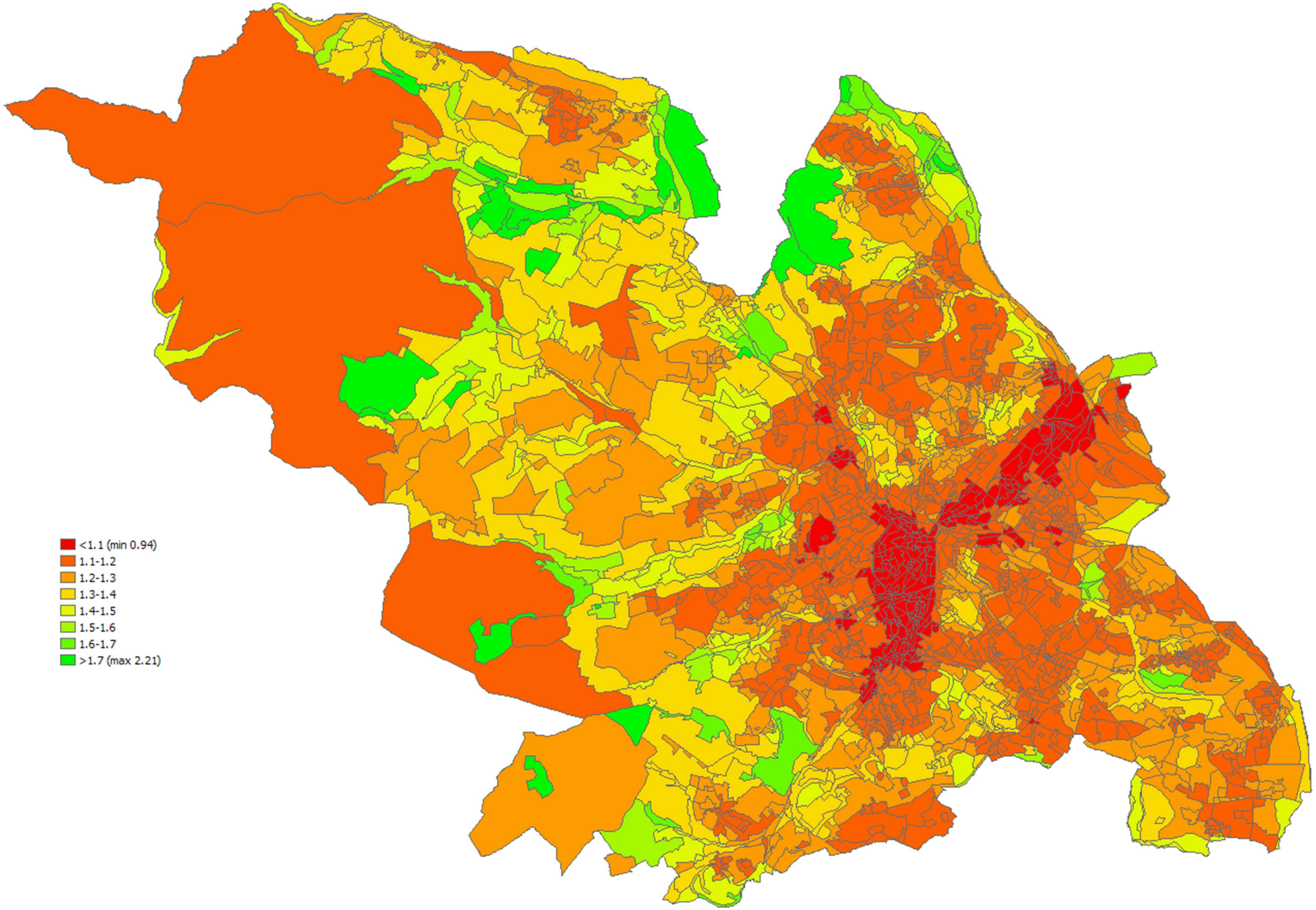
Map 4. NATMAP soils map of the study area; hatching indicates the part of the study area not covered by this dataset. Legend shows the NATMAP soil association ID (ID 6 is water).



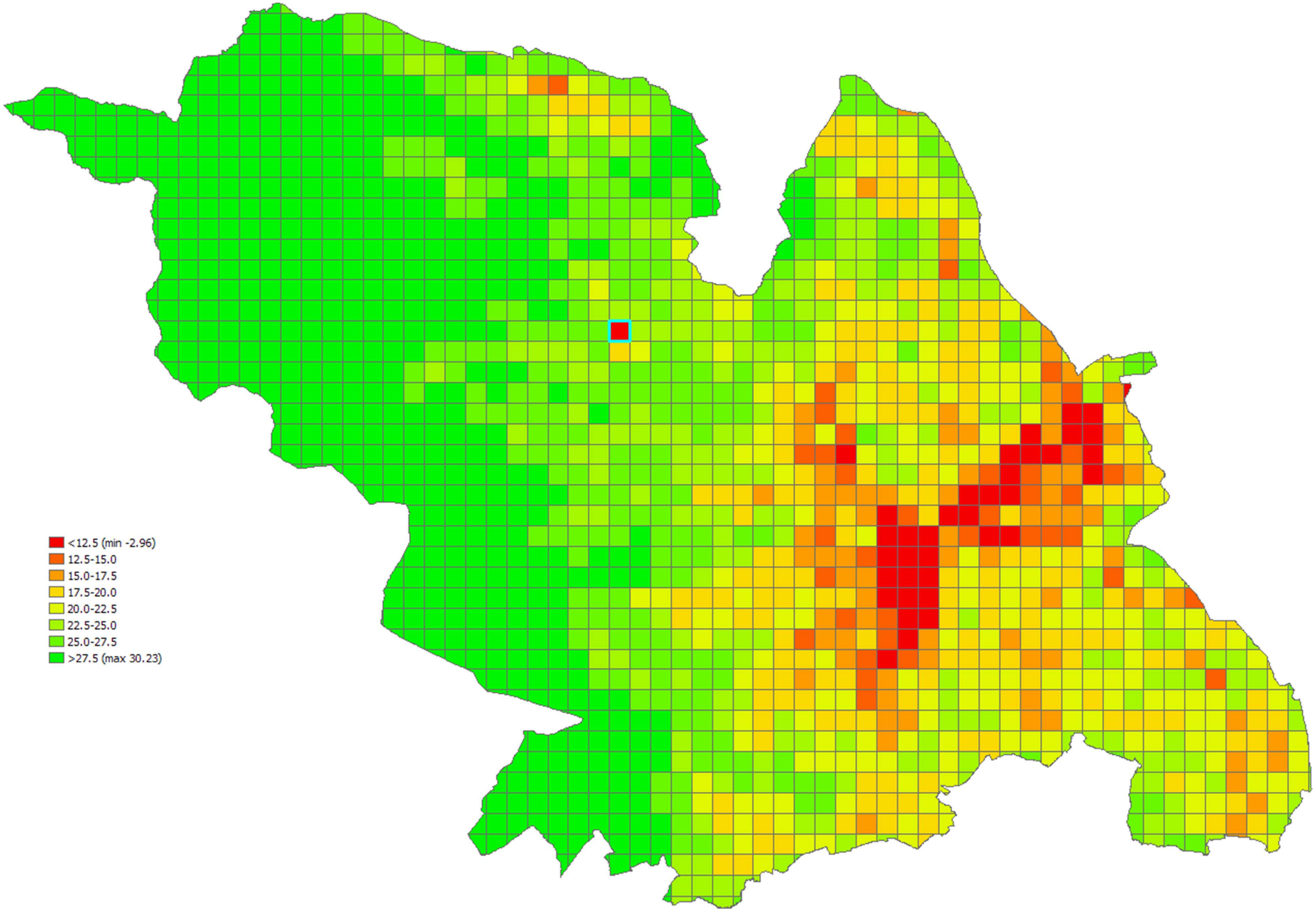
Map 5. Index of the ecosystem service of air pollution reduction, aggregated to 500m grid squares.



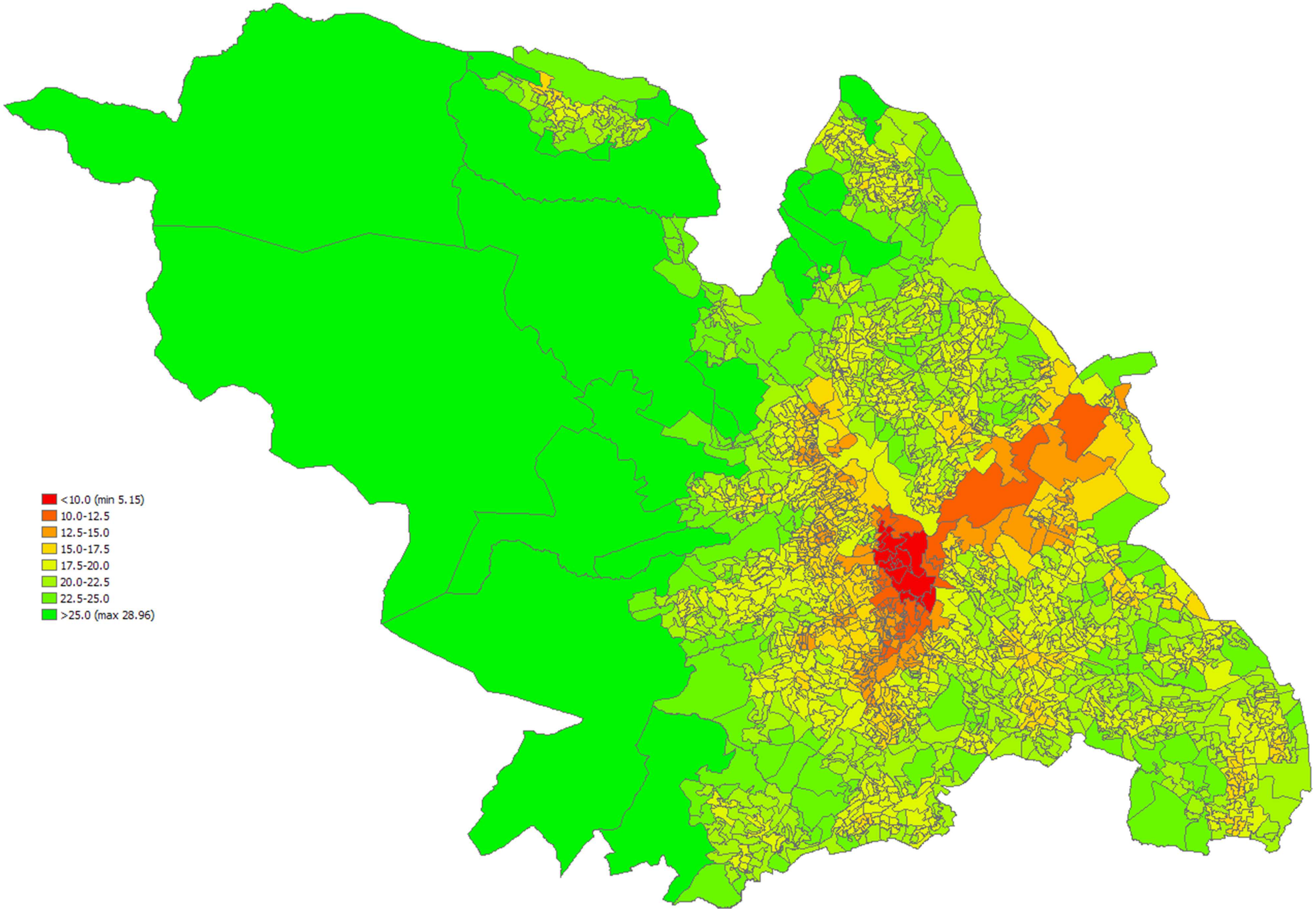
Map 6. Index of the ecosystem service of air pollution reduction, aggregated to Output Area boundaries.



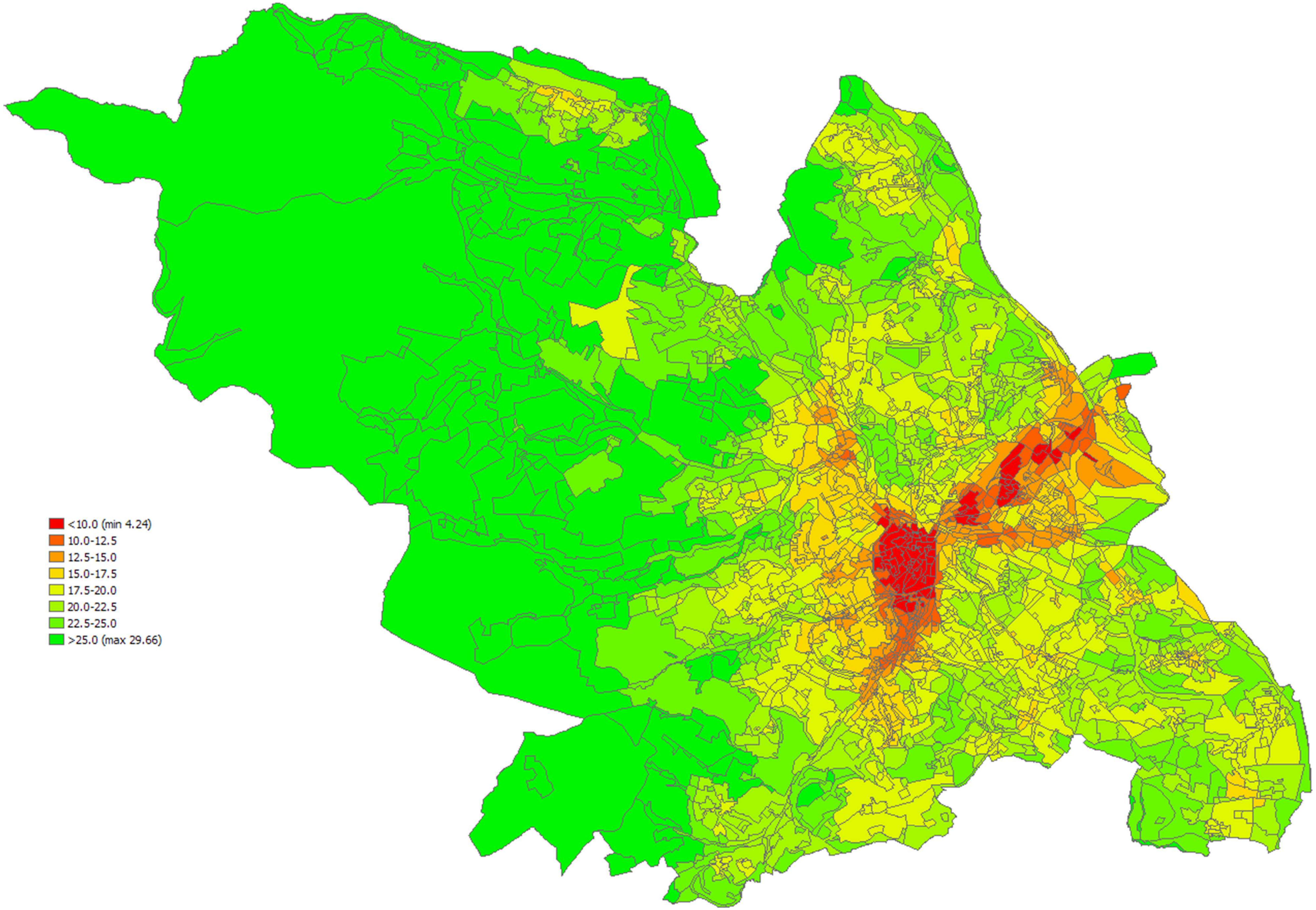
Map 7. Index of the ecosystem service of air pollution reduction, aggregated to Historic Environment Character area boundaries.



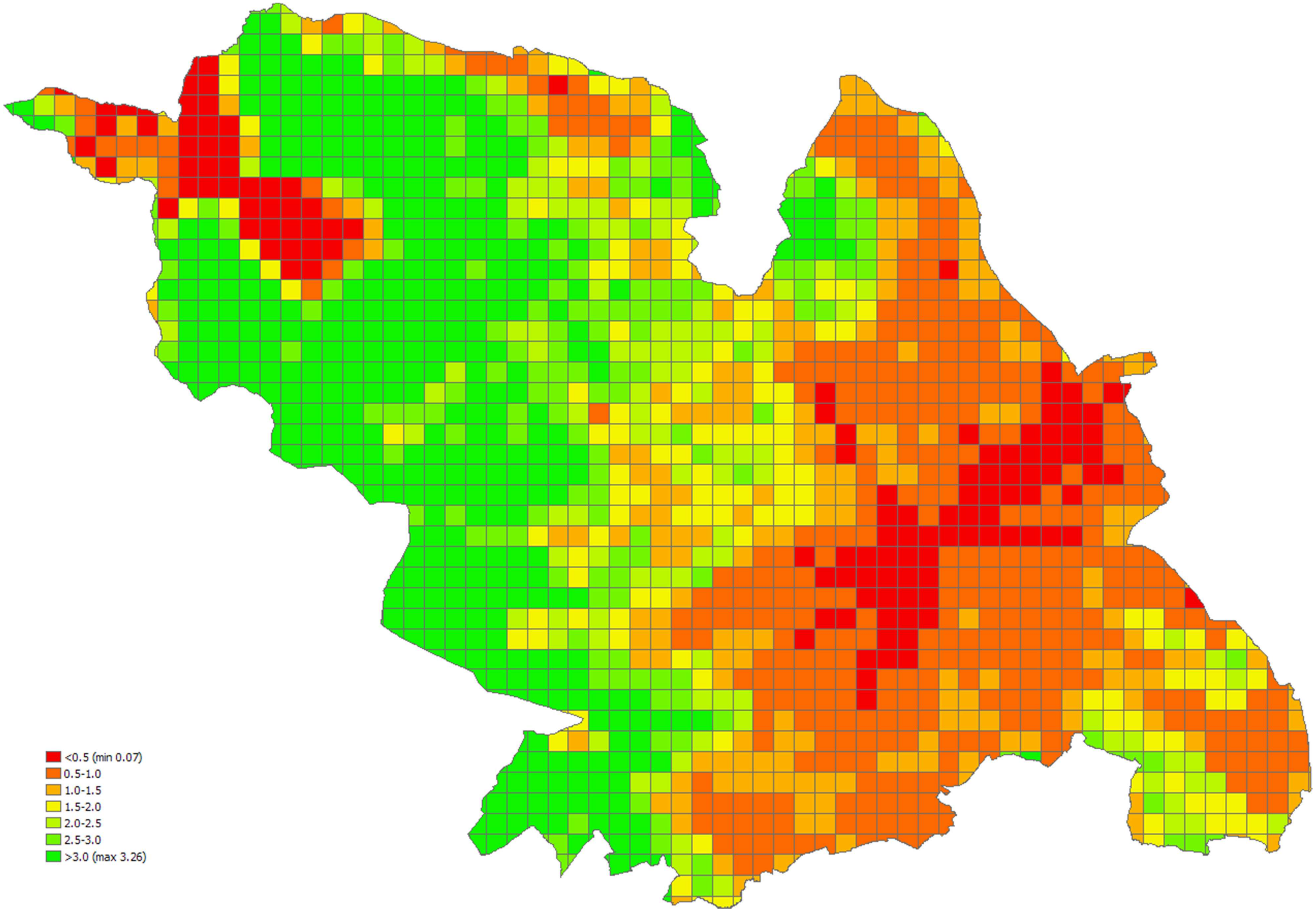
Map 8. The ecosystem service of heat island mitigation, aggregated to 500m grid squares. Units: °C. Square outlined in blue is the only negative value.



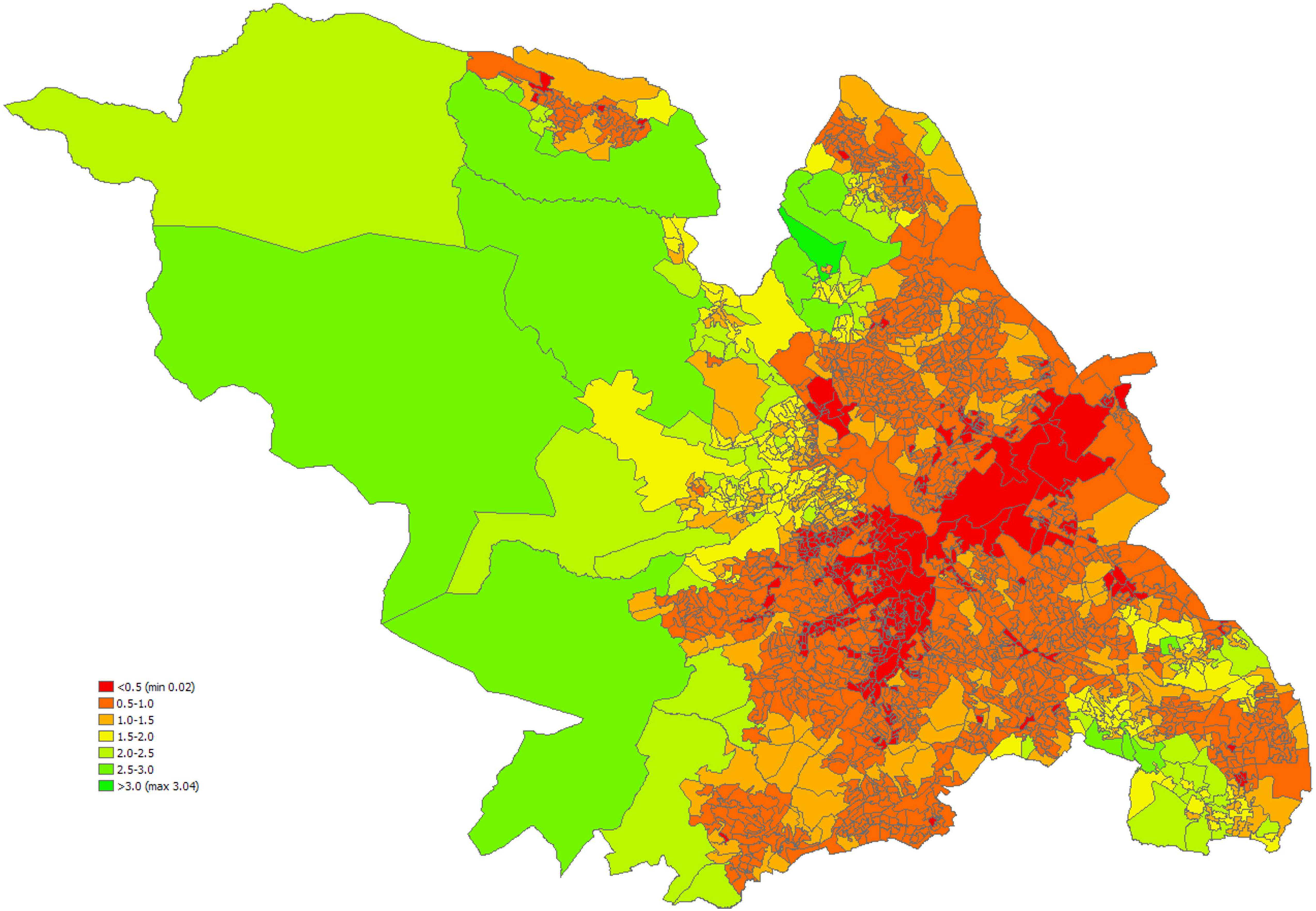
Map 9. The ecosystem service of heat island mitigation, aggregated to Output Area boundaries. Units: °C.



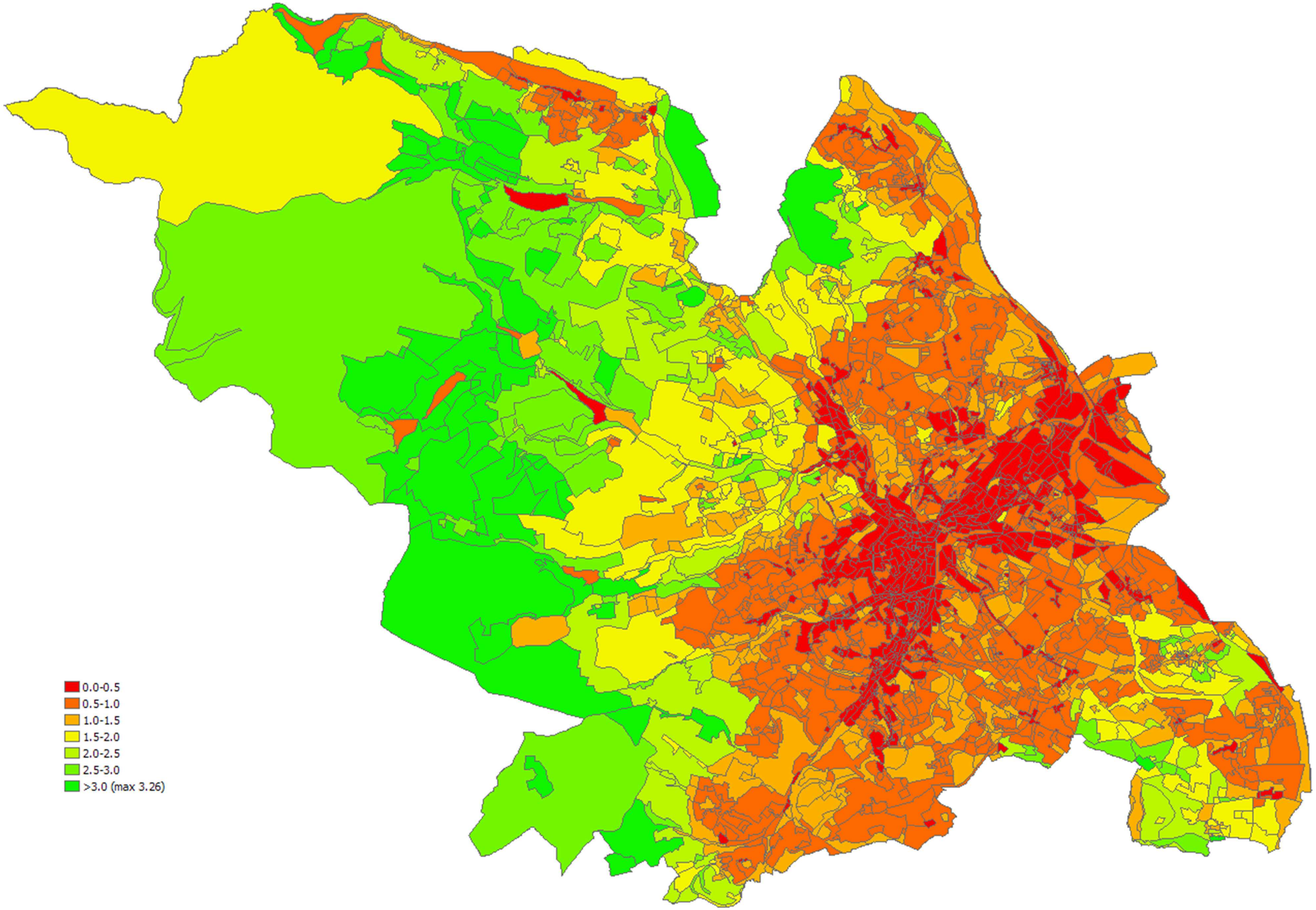
Map 10. The ecosystem service of heat island mitigation, aggregated to Historic Environment Character area boundaries. Units: °C.



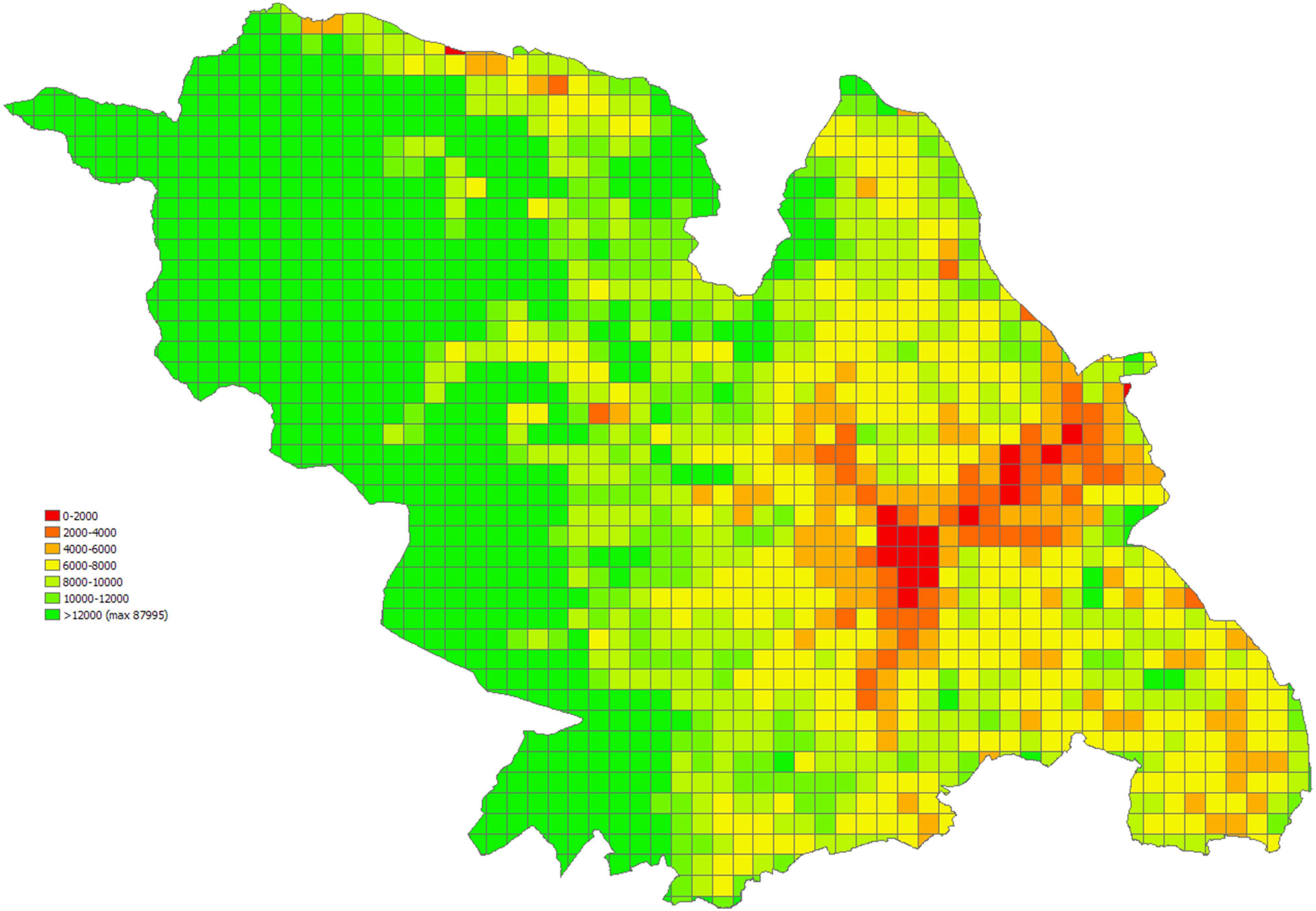
Map 11. The ecosystem service of reduction of storm water runoff, aggregated to 500m grid squares. Units: cm depth.



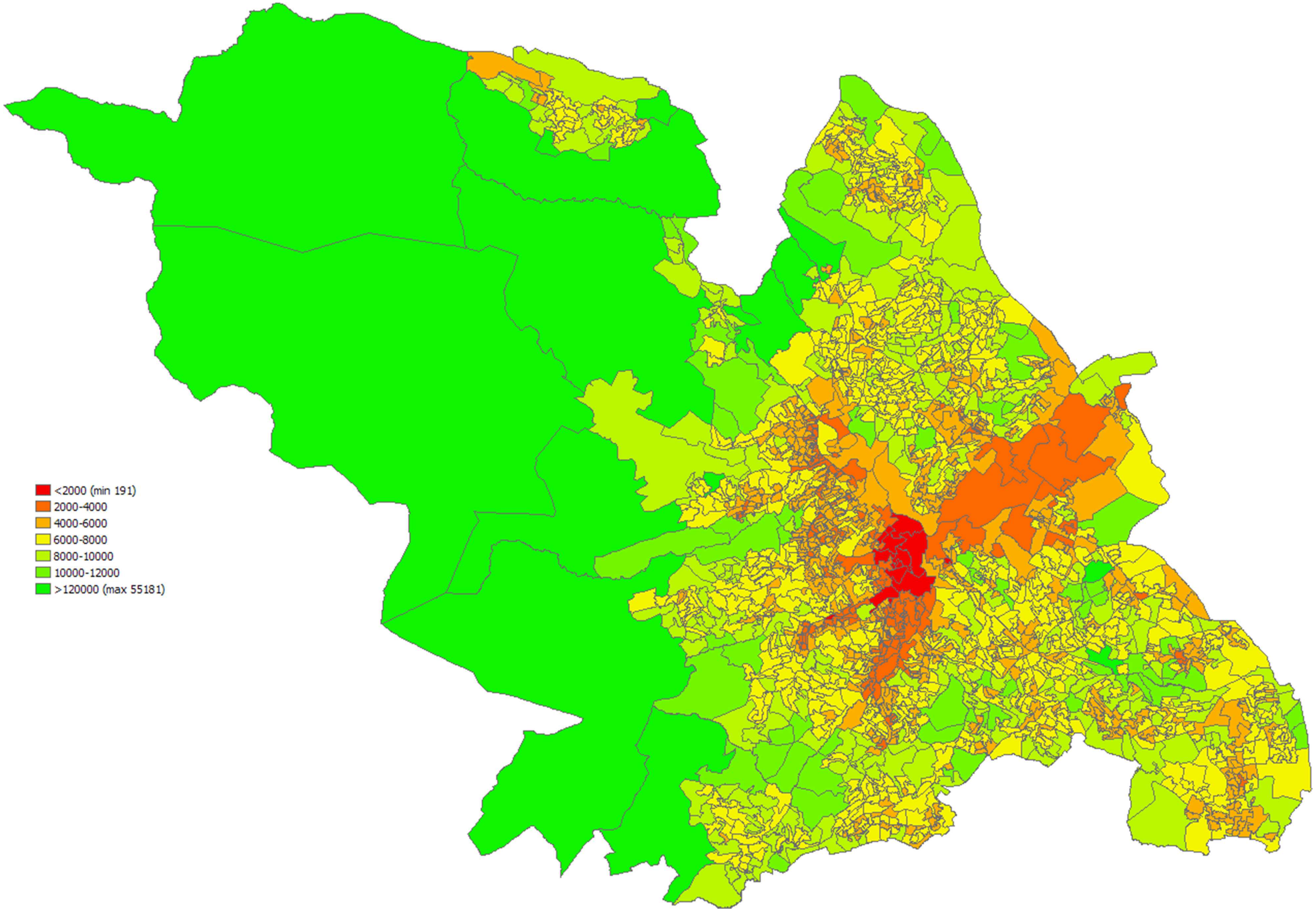
Map 12. The ecosystem service of reduction of storm water runoff, aggregated to Output Area boundaries. Units: cm depth.



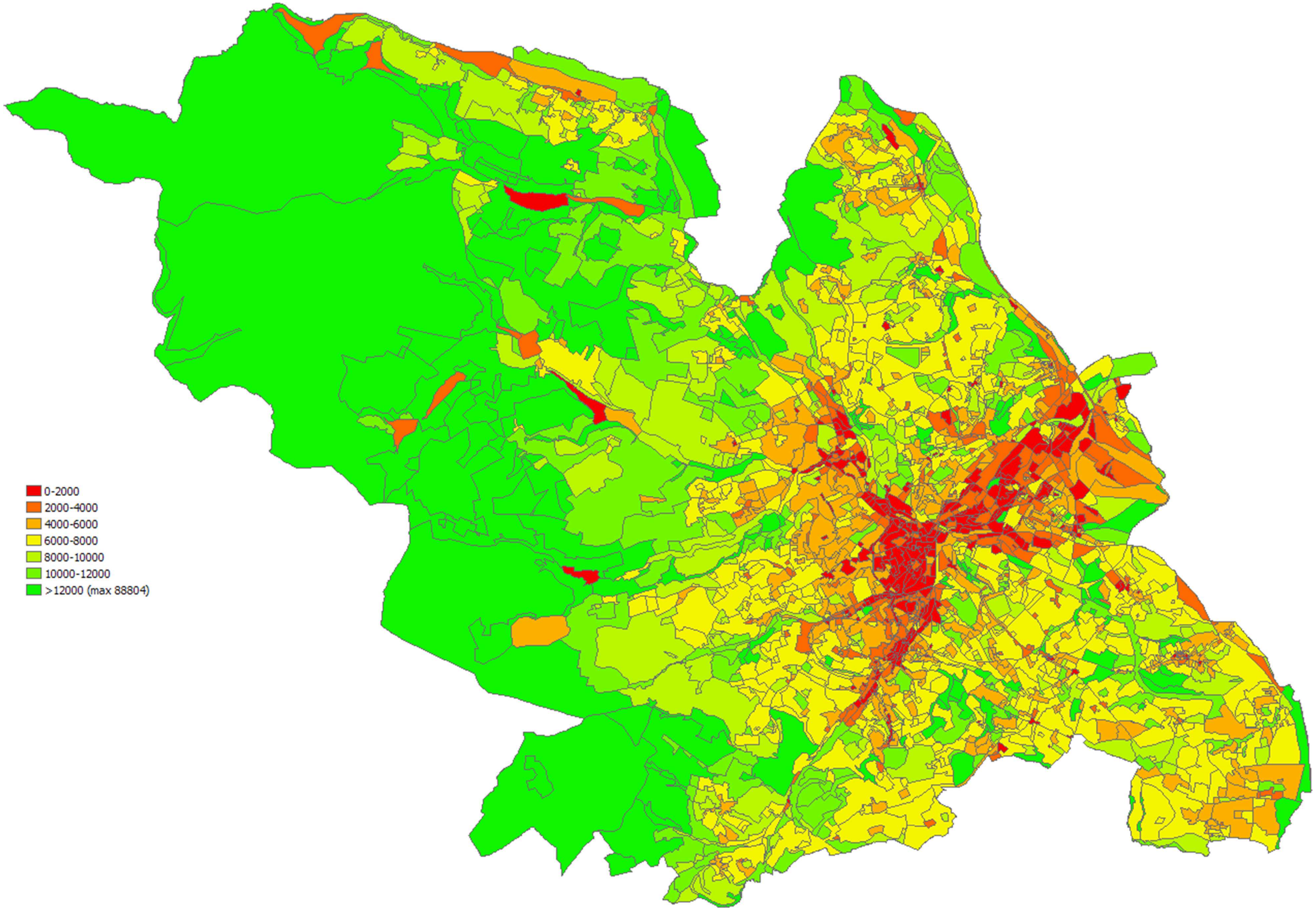
Map 13. The ecosystem service of reduction of storm water runoff, aggregated to Historic Environment Character area boundaries. Units: cm depth.



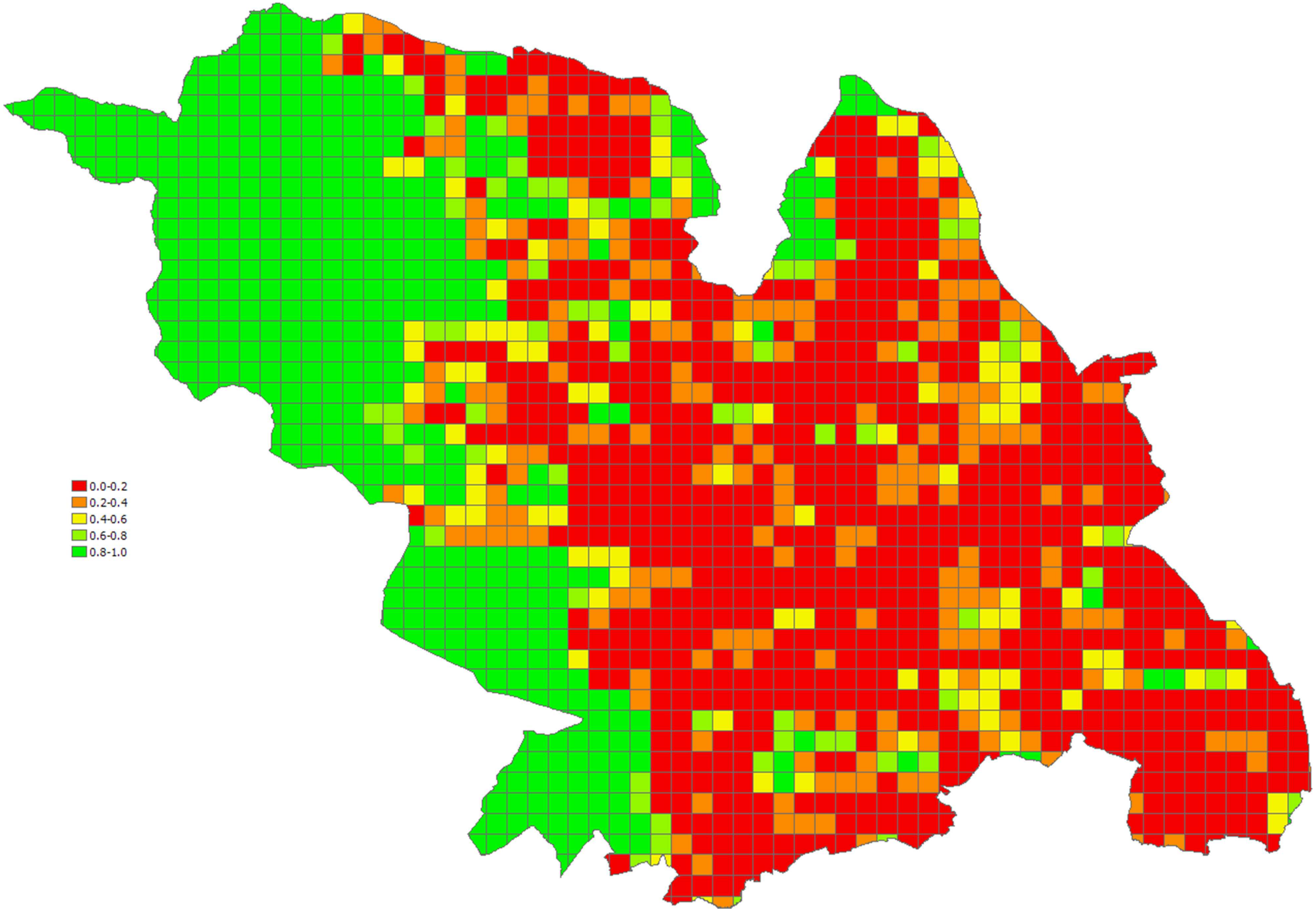
Map 14. The ecosystem service of carbon storage, aggregated to 500m grid squares. Units: g m^{-2}.



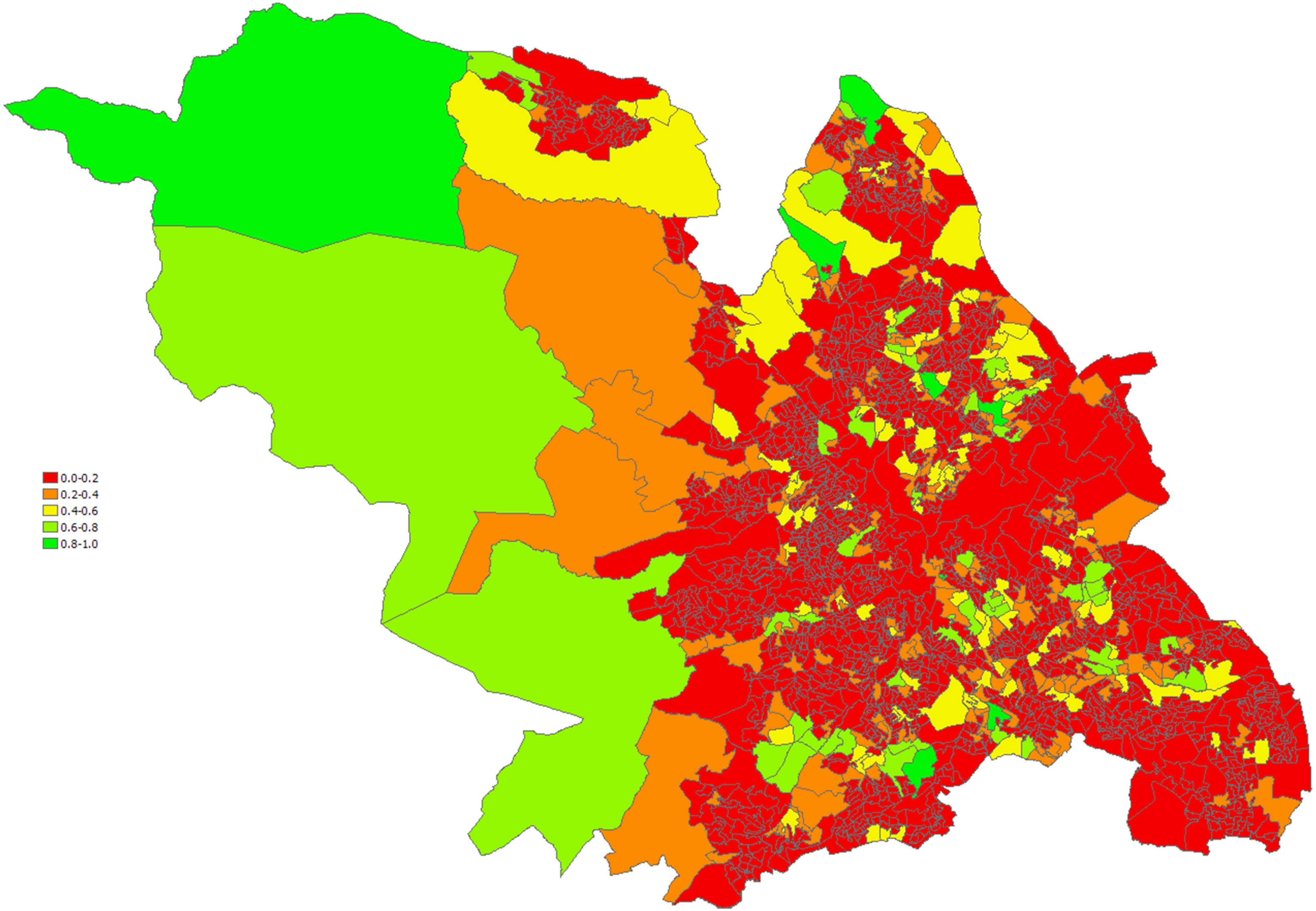
Map 15. The ecosystem service of carbon storage, aggregated to Output Area boundaries. Units: g m^{-2}.



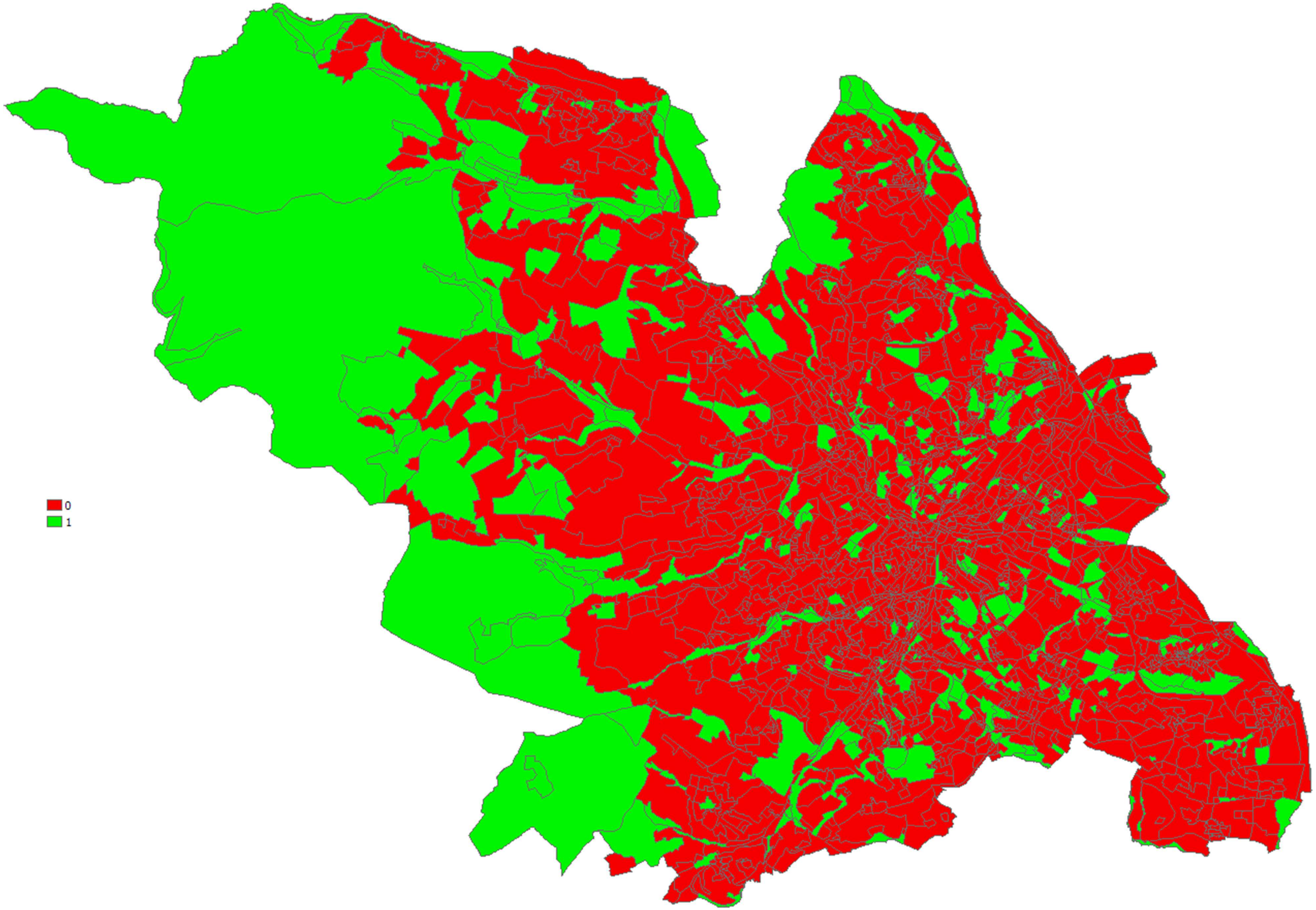
Map 16. The ecosystem service of carbon storage, aggregated to Historic Environment Character area boundaries. Units: g m^{-2}.



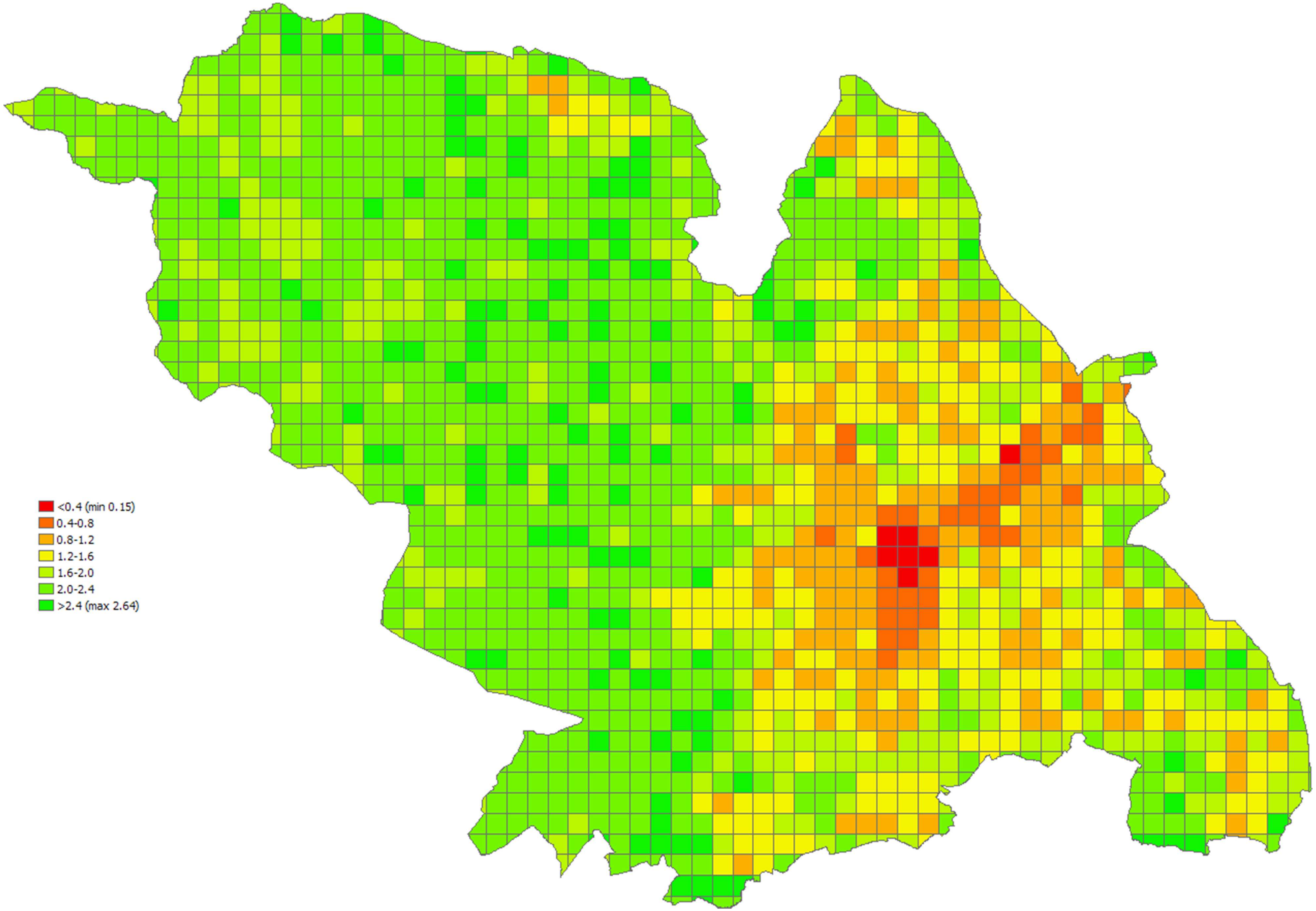
Map 17. Index of production of opportunities for cultural ecosystem services from greenspace, aggregated to 500m grid squares.



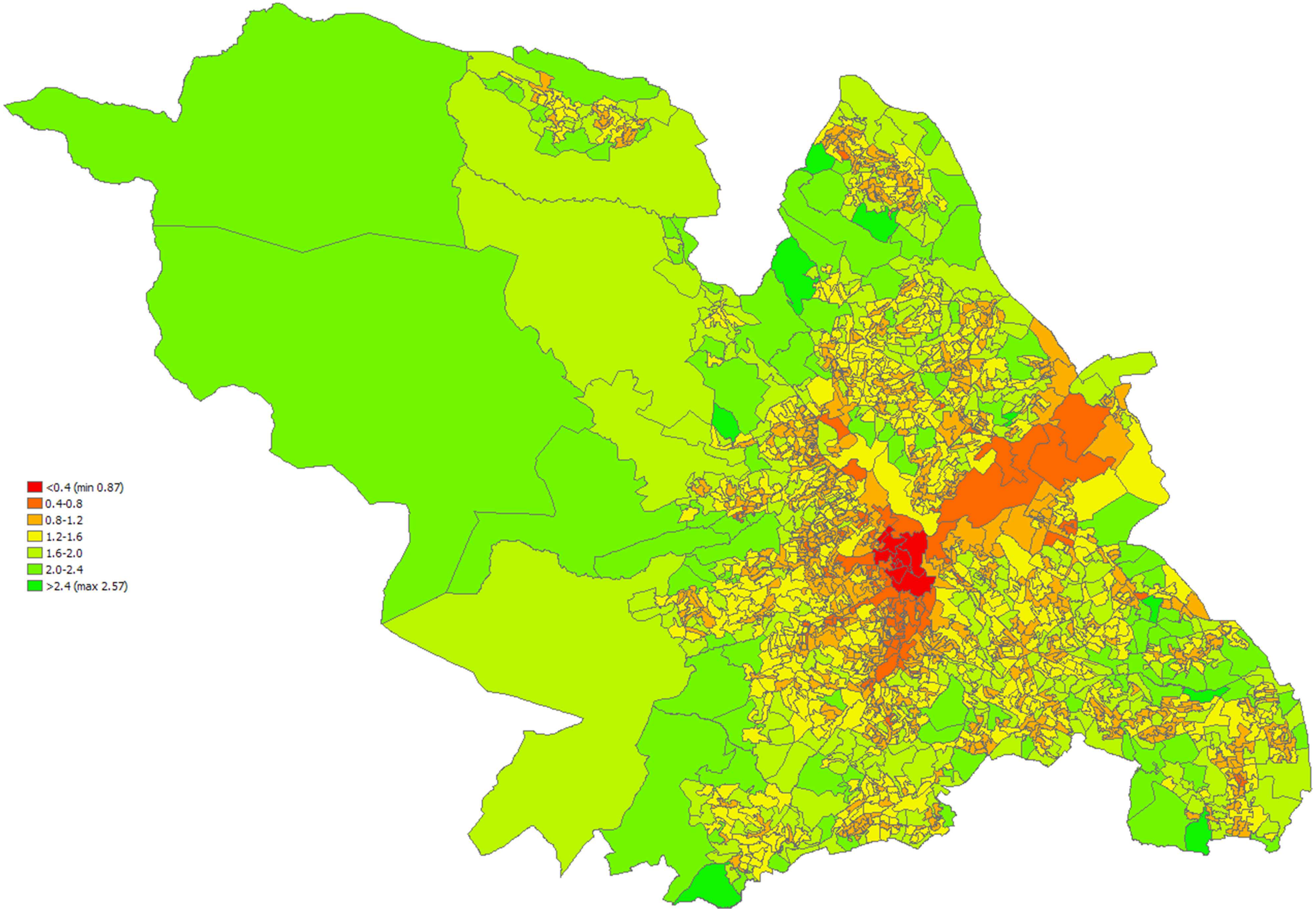
Map 18. Index of production of opportunities for cultural ecosystem services from greenspace, aggregated to Output Areas.



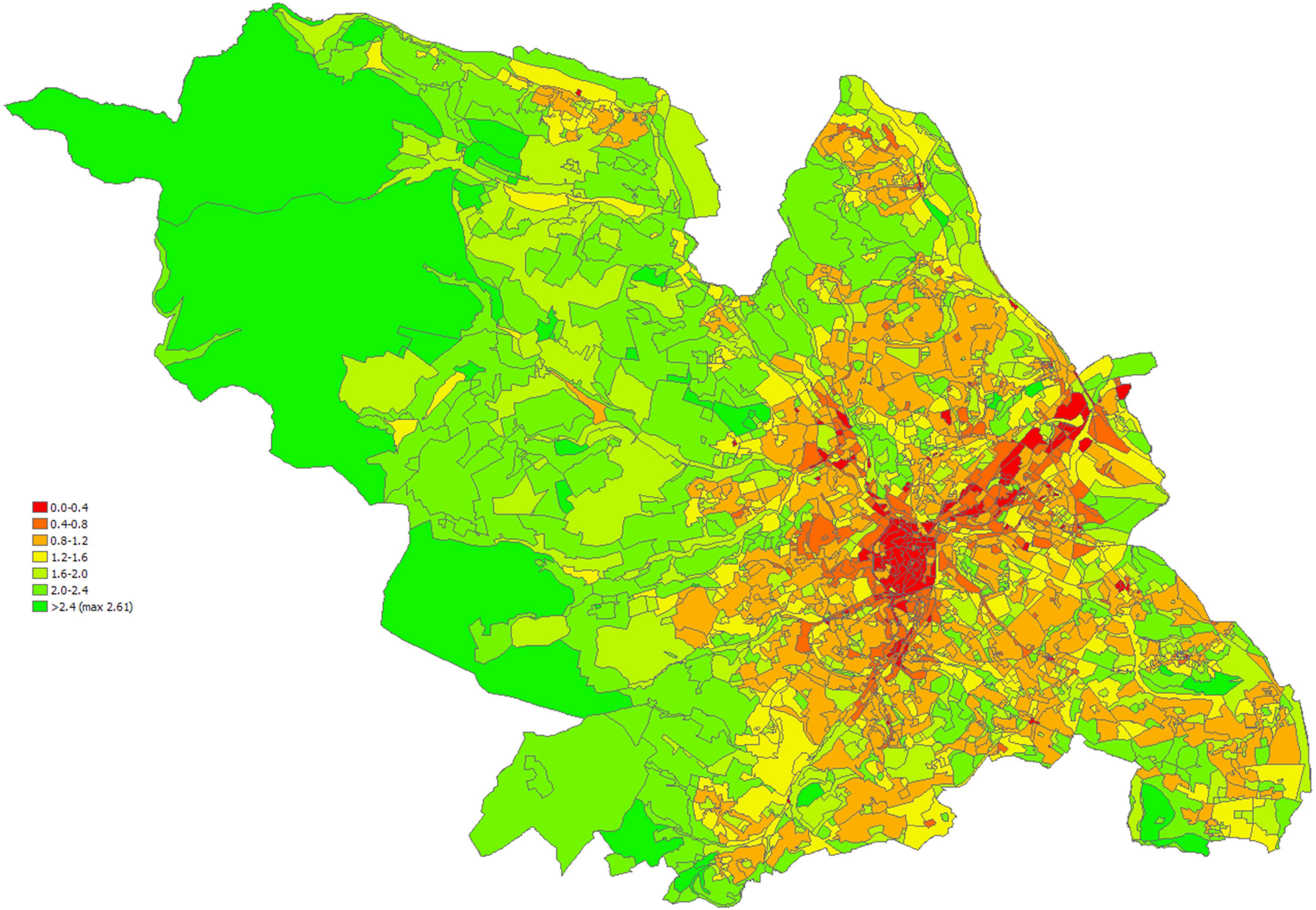
Map 19. Index of production of opportunities for cultural ecosystem services from greenspace, aggregated to Historic Environment Character areas.



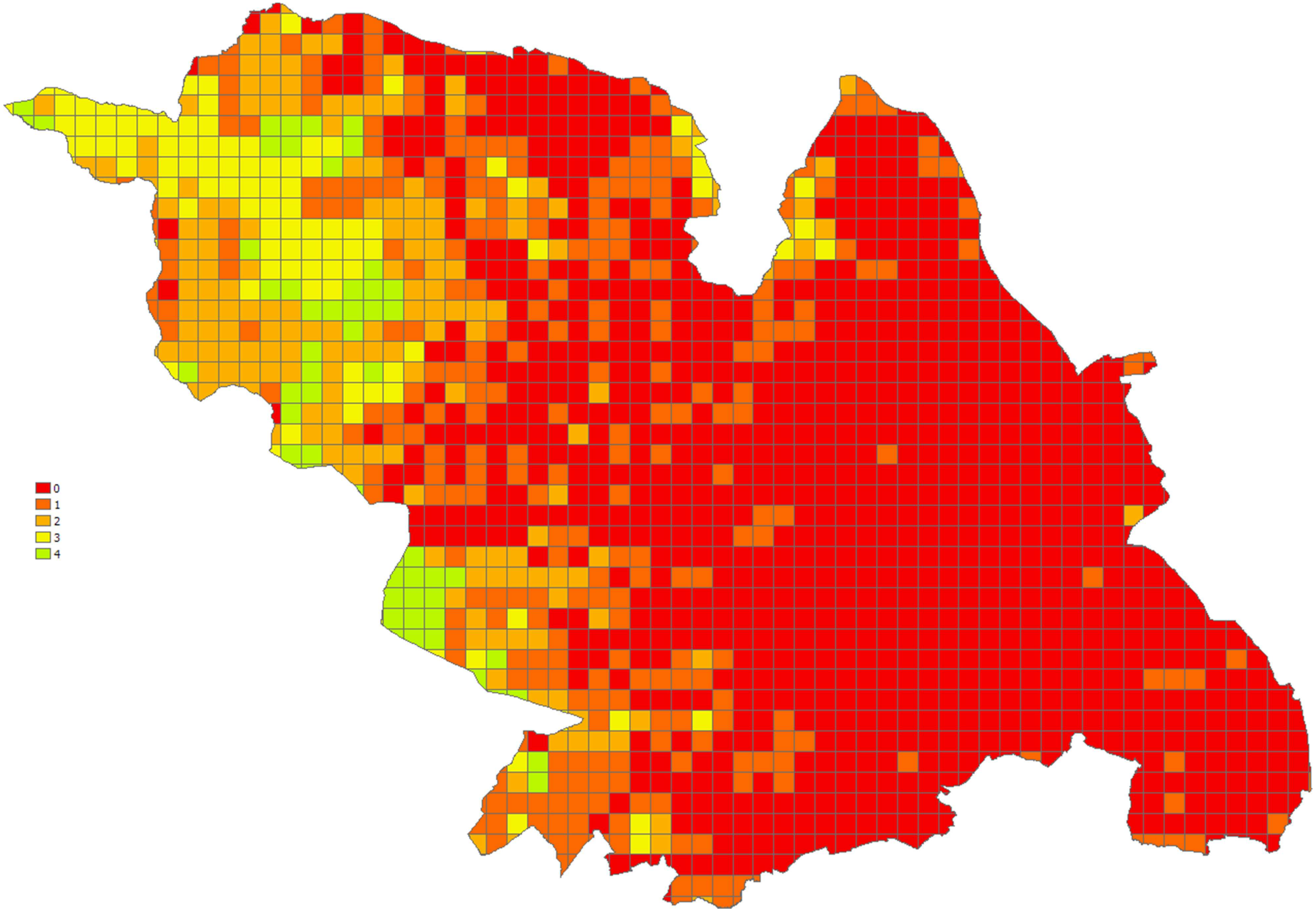
Map 20. Index of habitat for flora and fauna, aggregated to 500m grid squares.



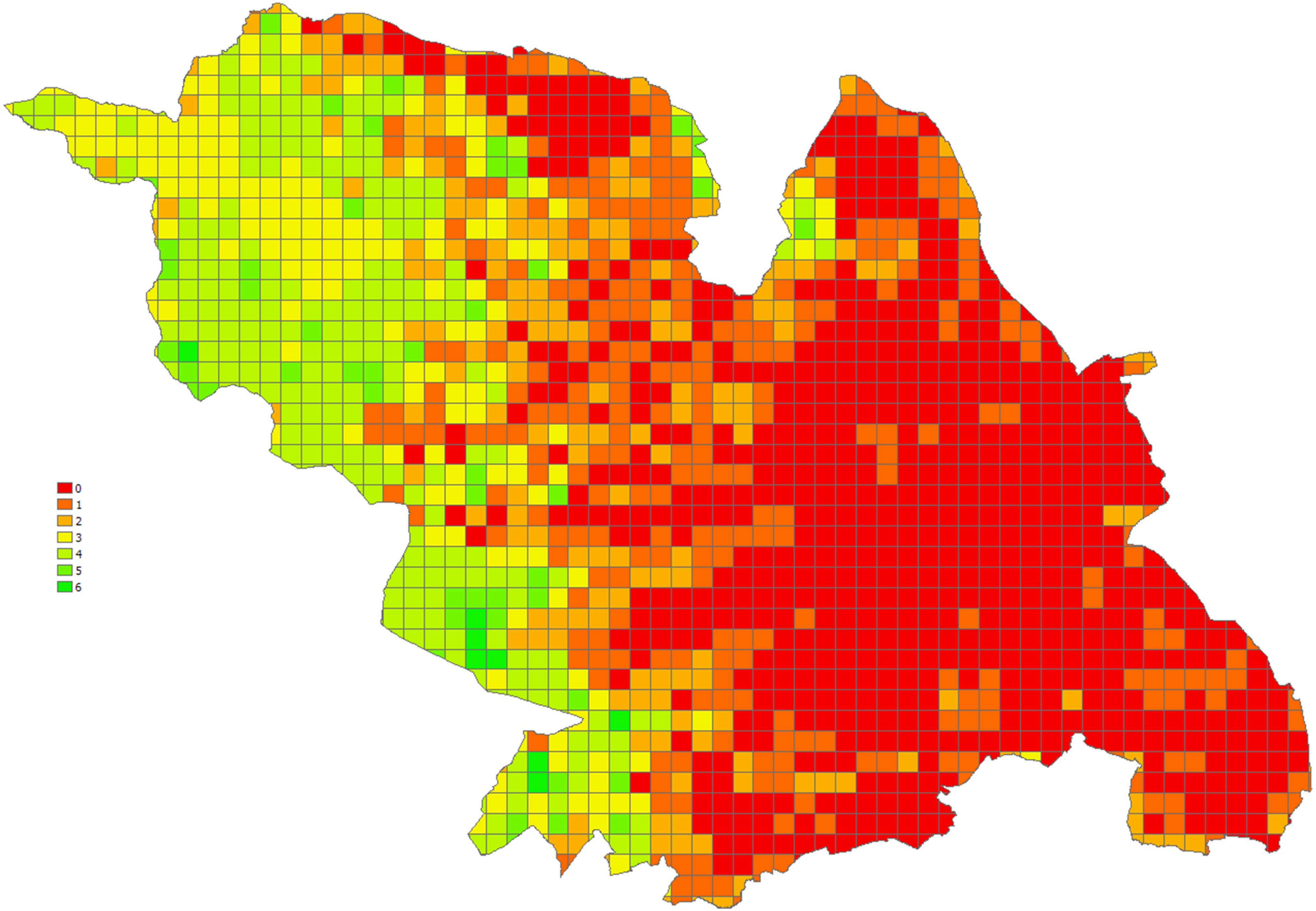
Map 21. Index of habitat for flora and fauna, aggregated to Output Area boundaries.



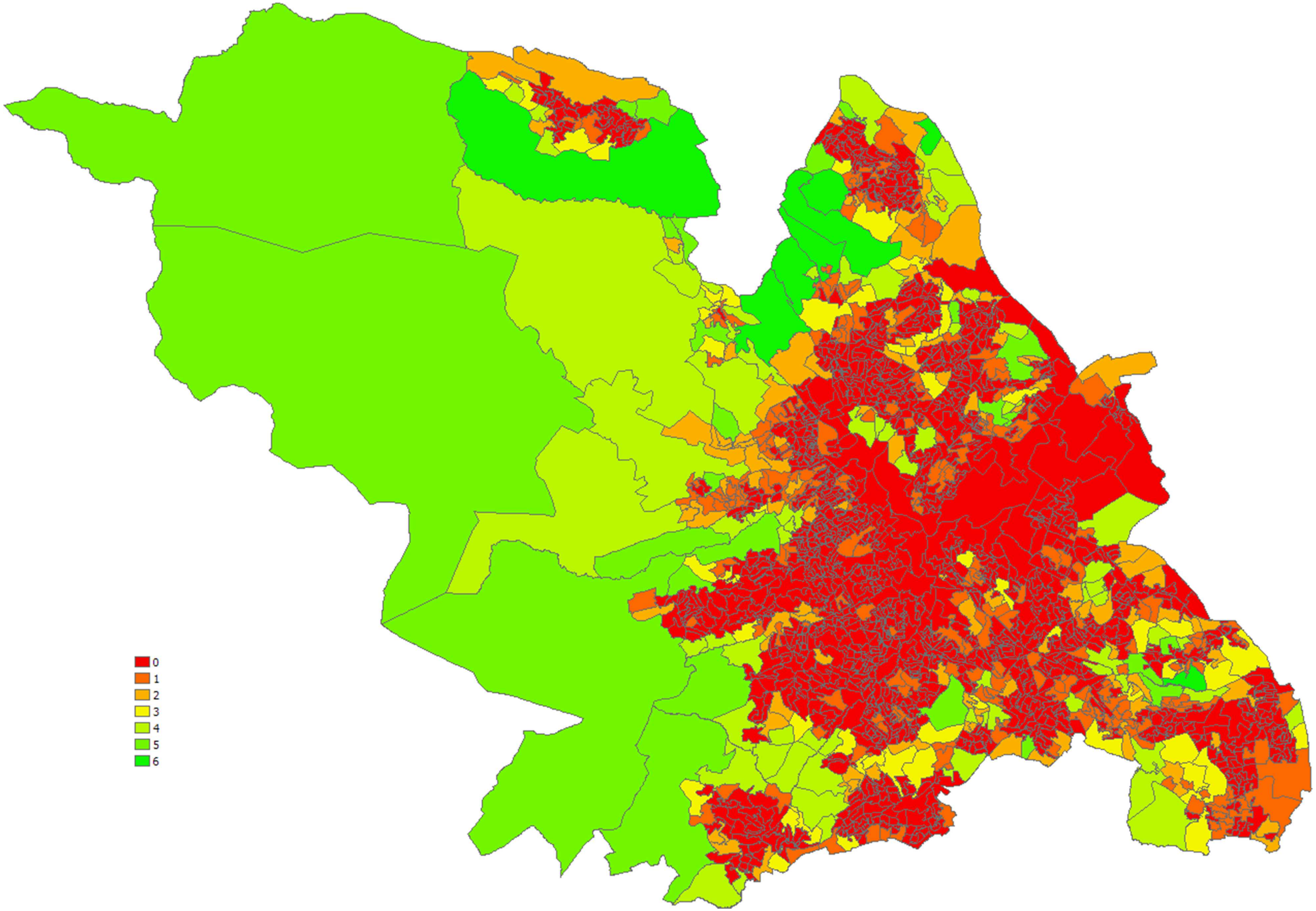
Map 22. Index of habitat for flora and fauna, aggregated to Historic Environment Character area boundaries.



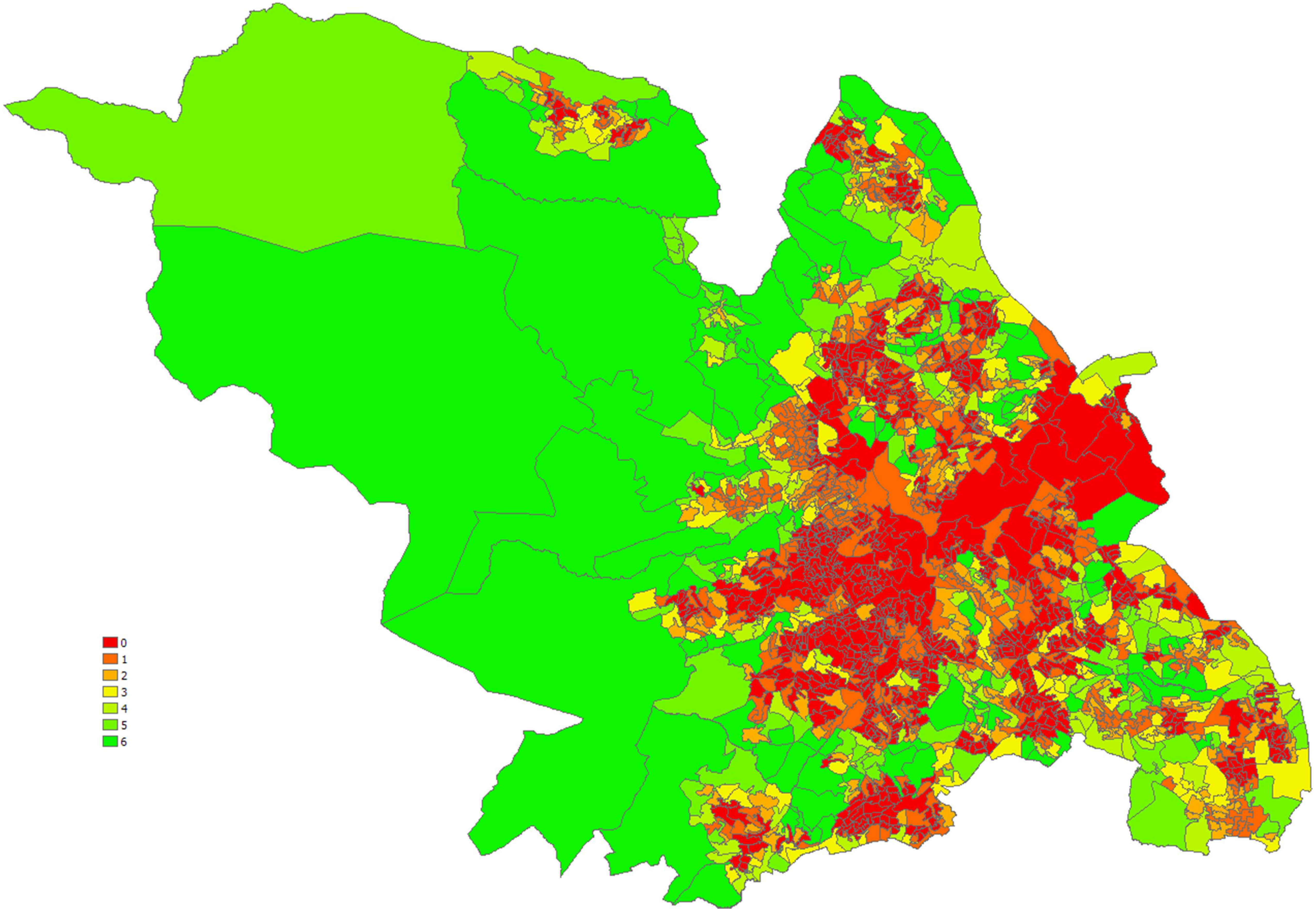
Map 23. Number of ecosystem services for which each 500m grid square is a hotspot according to the top 10% threshold.



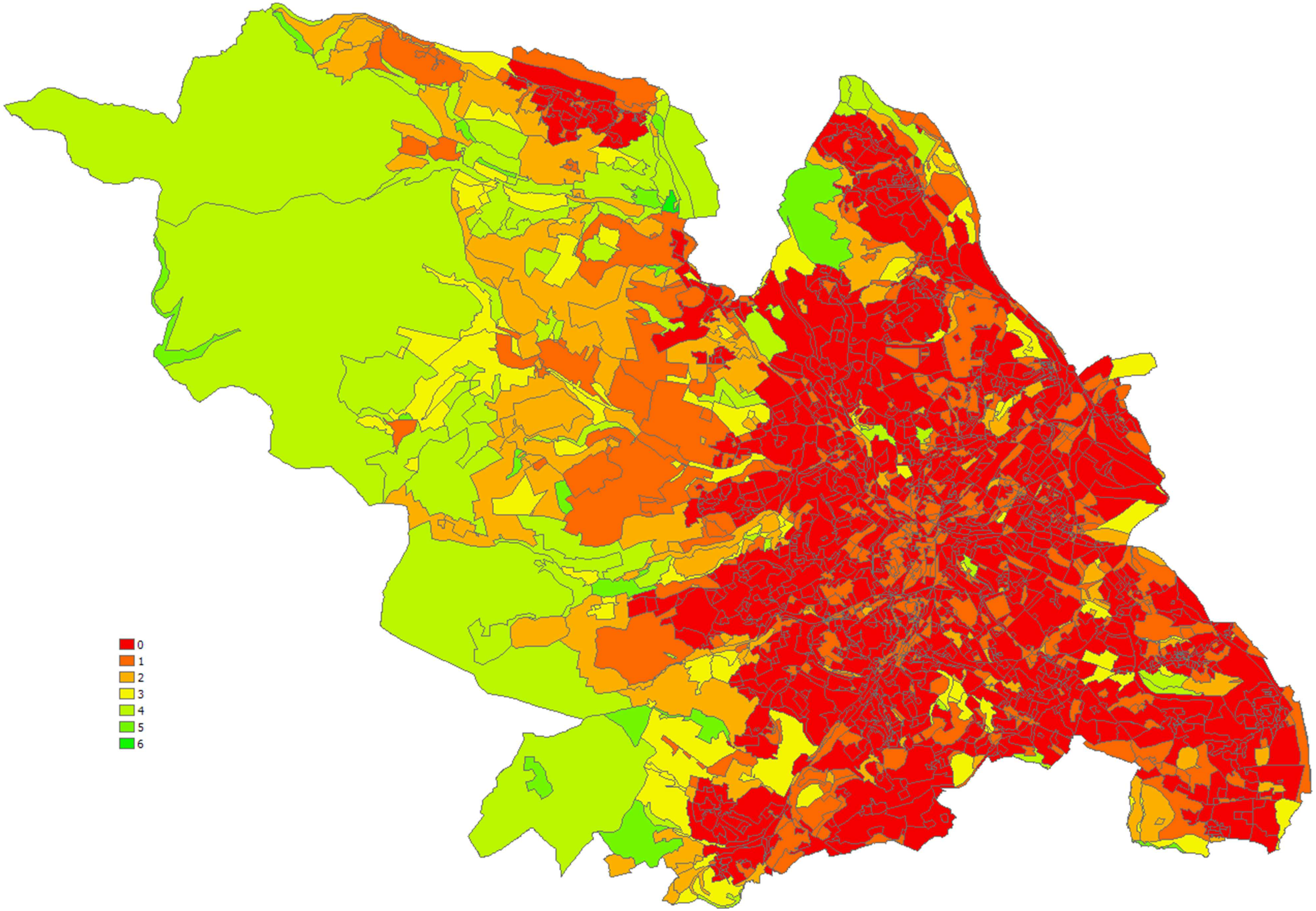
Map 24. Number of ecosystem services for which each 500m grid square is a hotspot according to the top 25% threshold.



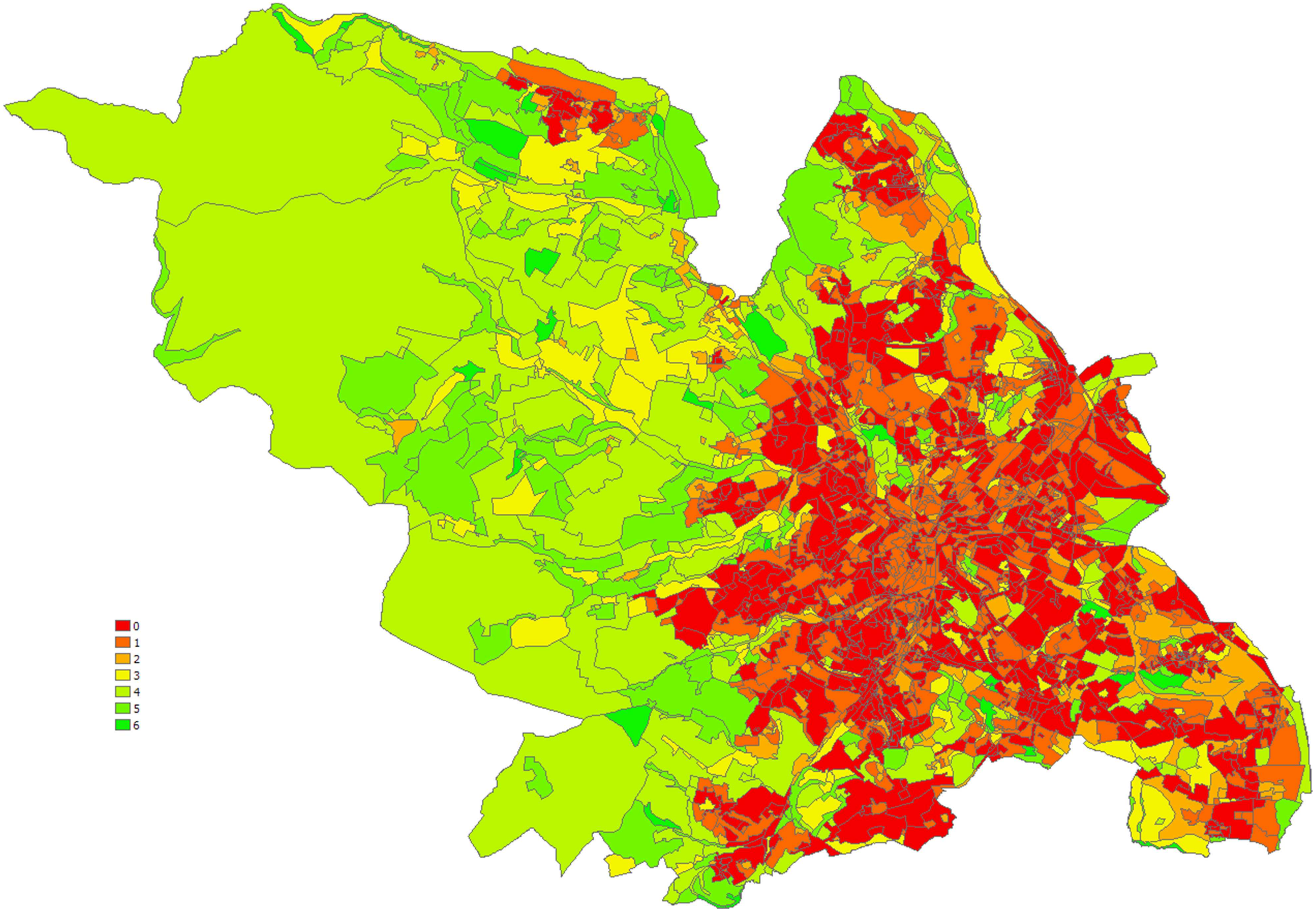
Map 25. Number of ecosystem services for which each Output Area is a hotspot according to the top 10% threshold.



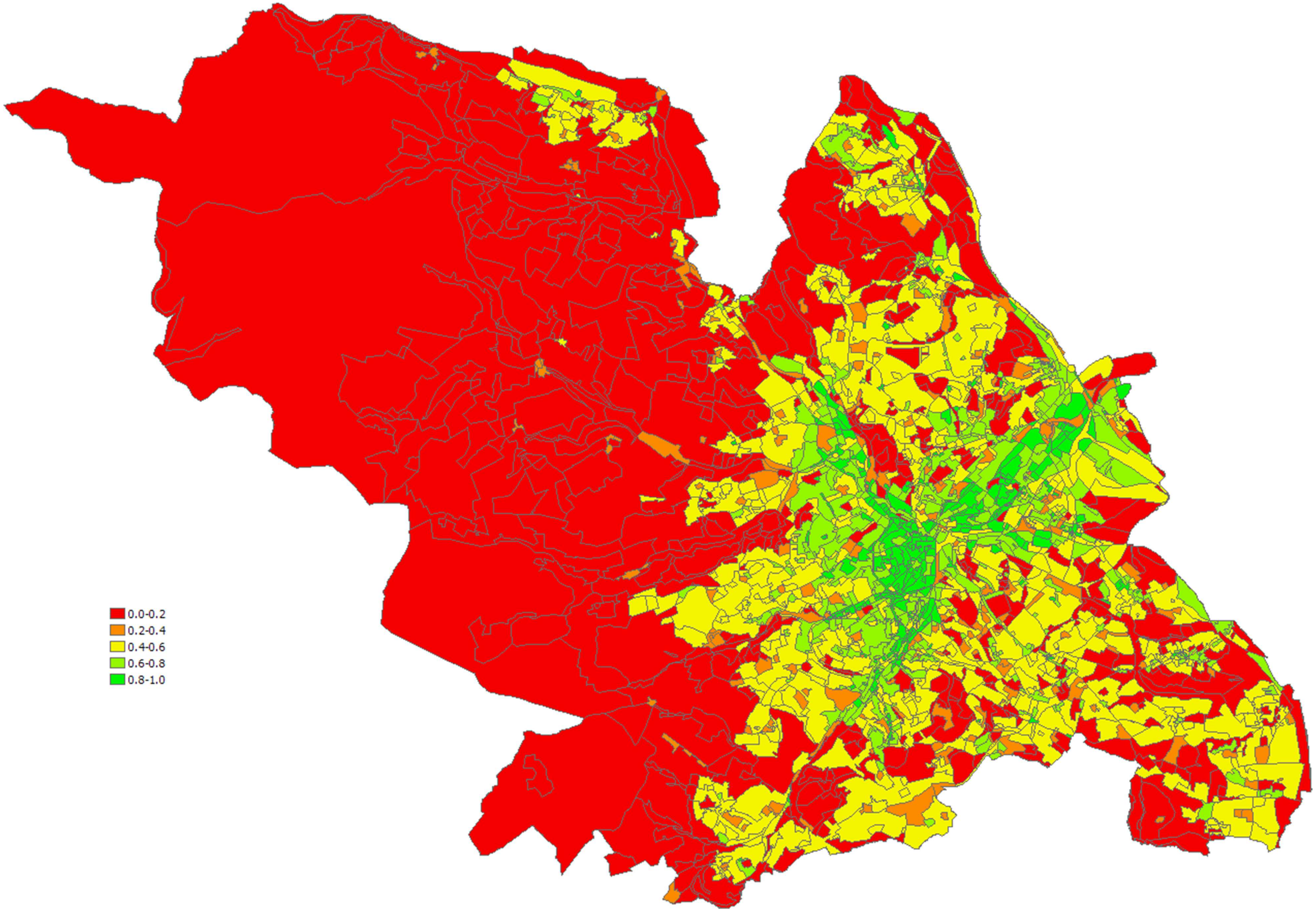
Map 26. Number of ecosystem services for which each Output Area is a hotspot according to the top 25% threshold.



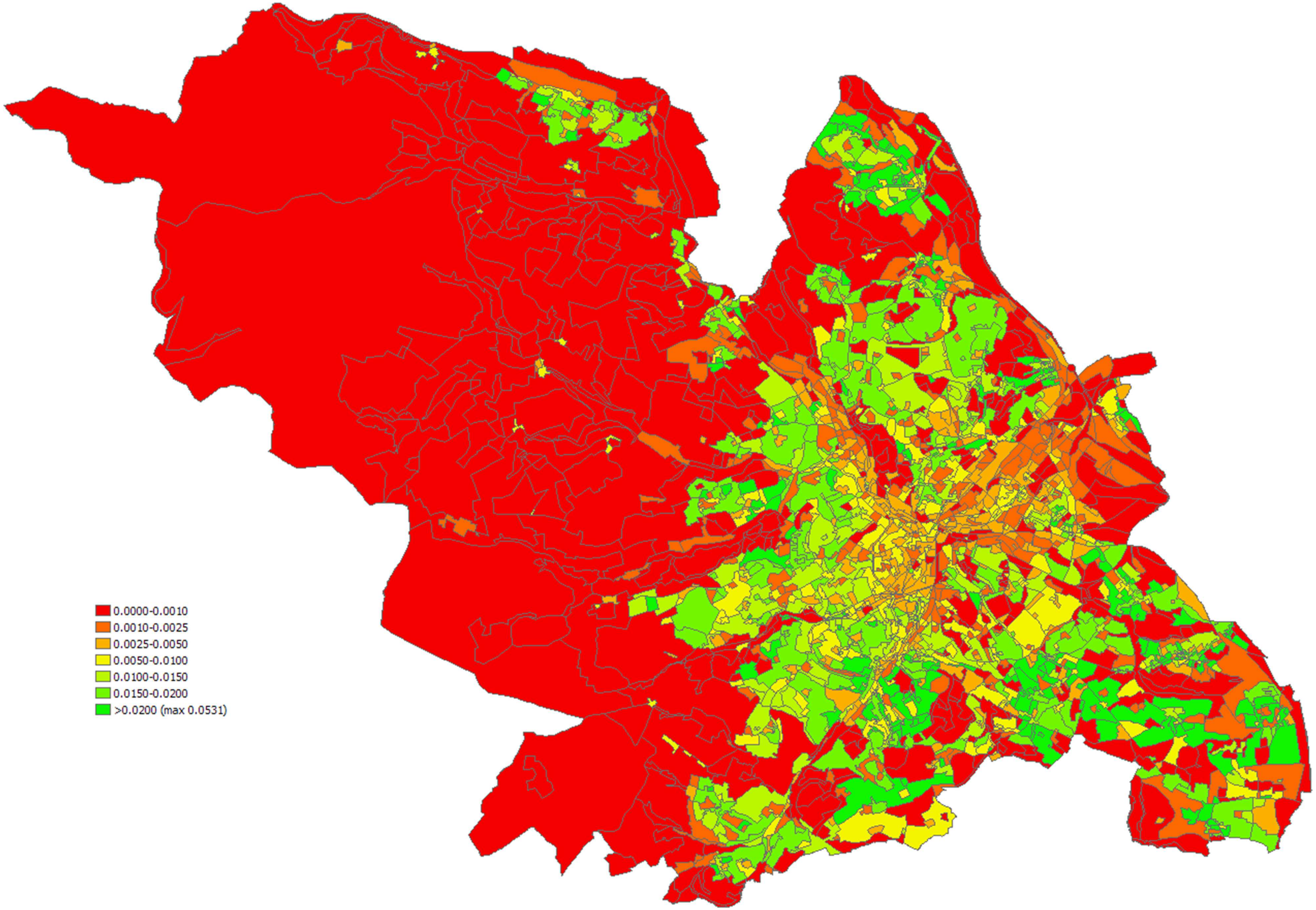
Map 27. Number of ecosystem services for which each Historic Environment Character area is a hotspot according to the top 10% threshold.



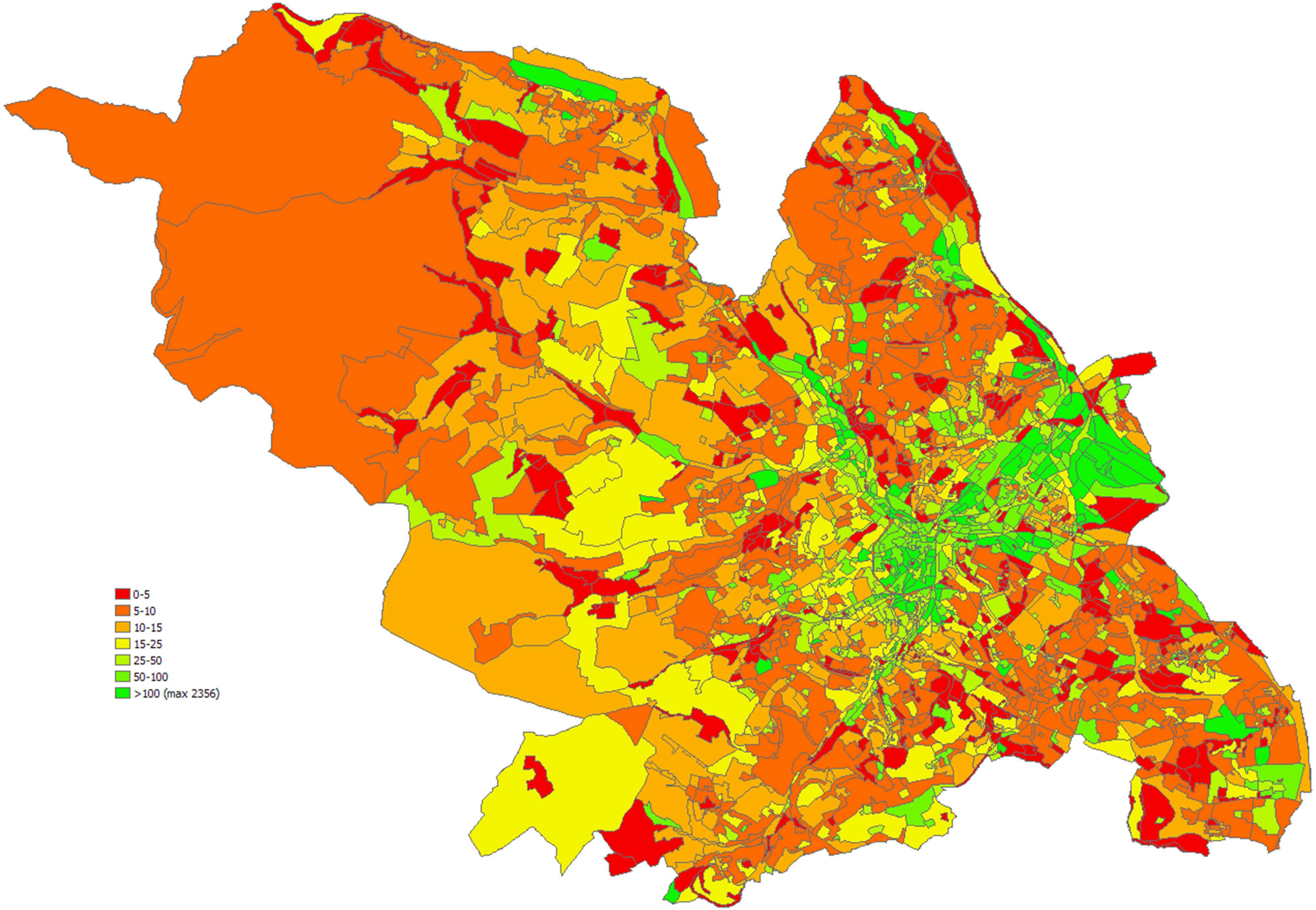
Map 28. Number of ecosystem services for which each Historic Environment Character area is a hotspot according to the top 25% threshold.



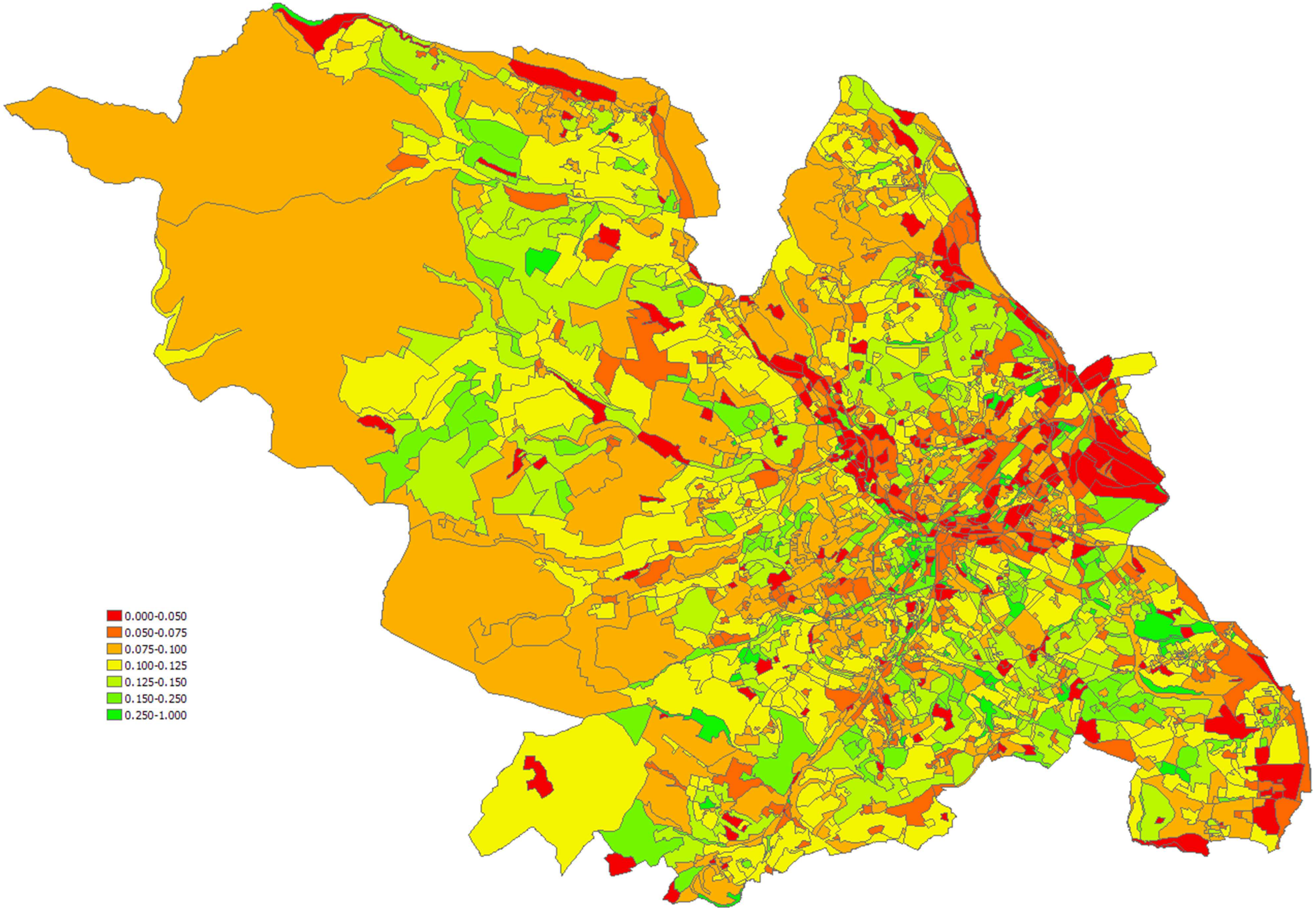
Map 29. Proportion of impervious cover over Historic Environment Character areas.



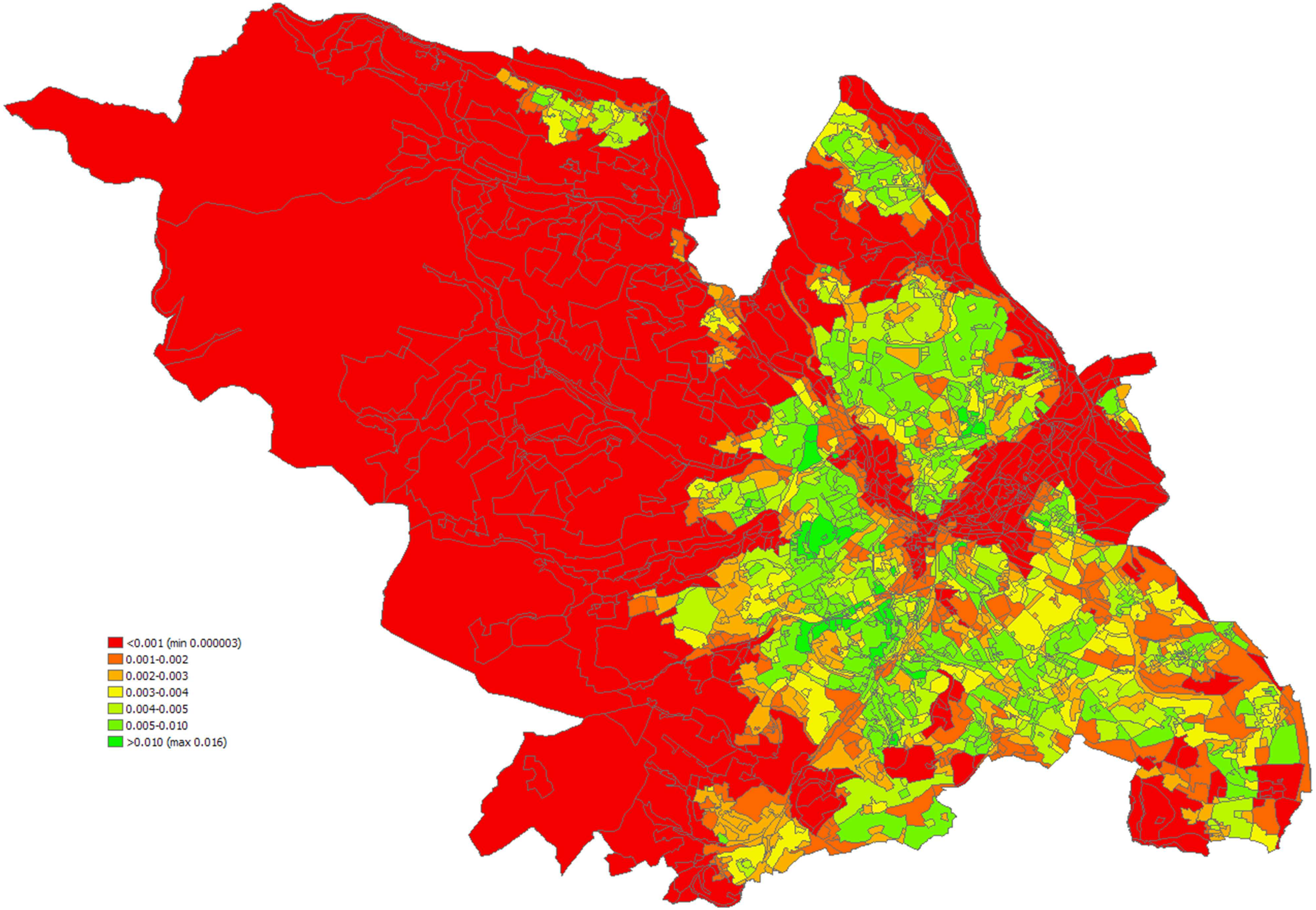
Map 30. Building density over Historic Environment Character areas. Note non-linearities in legend. (Units: discrete buildings per m^2)



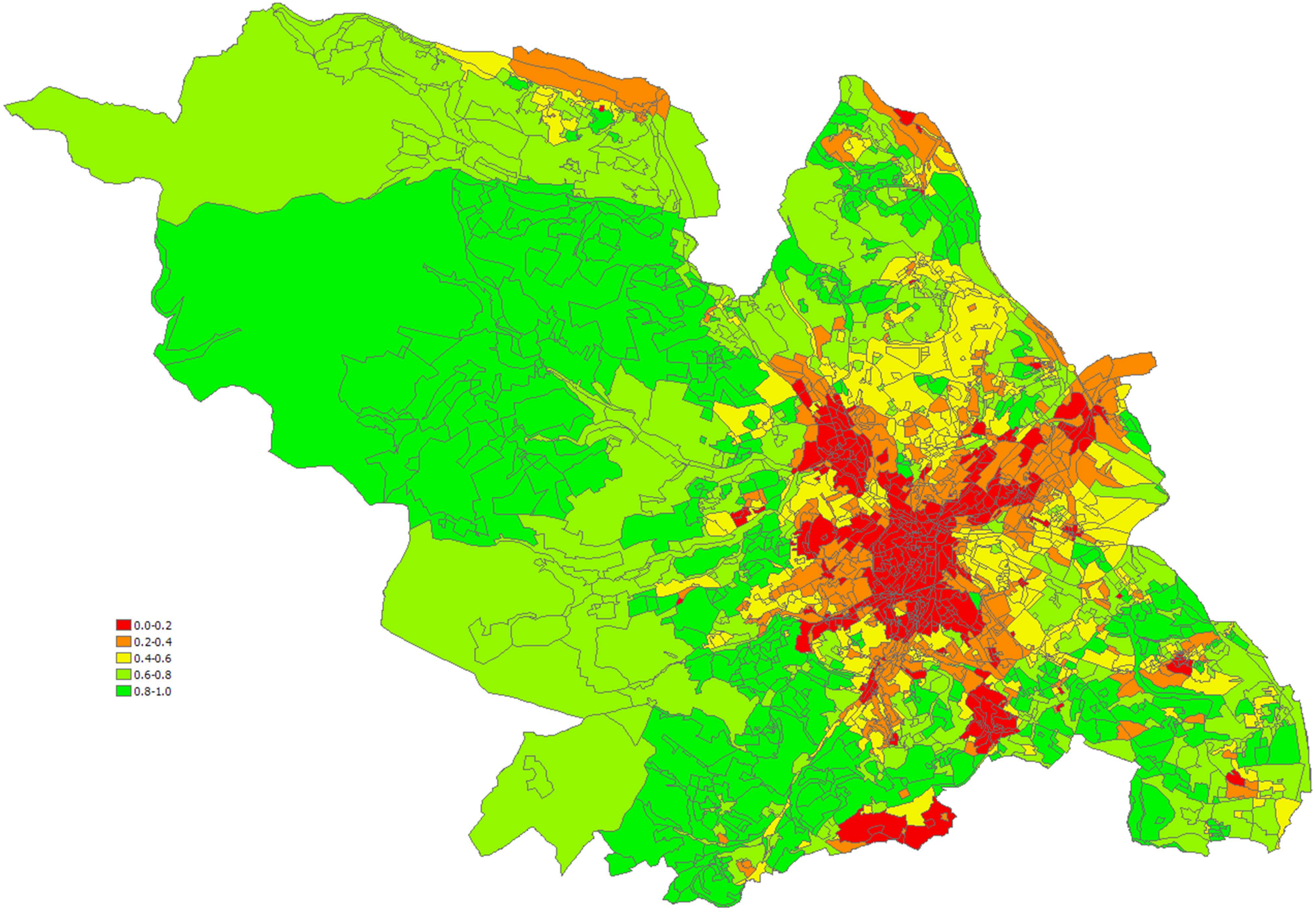
Map 31. Mean building size over Historic Environment Character areas. Note non-linearities in legend. (Units: m^2)



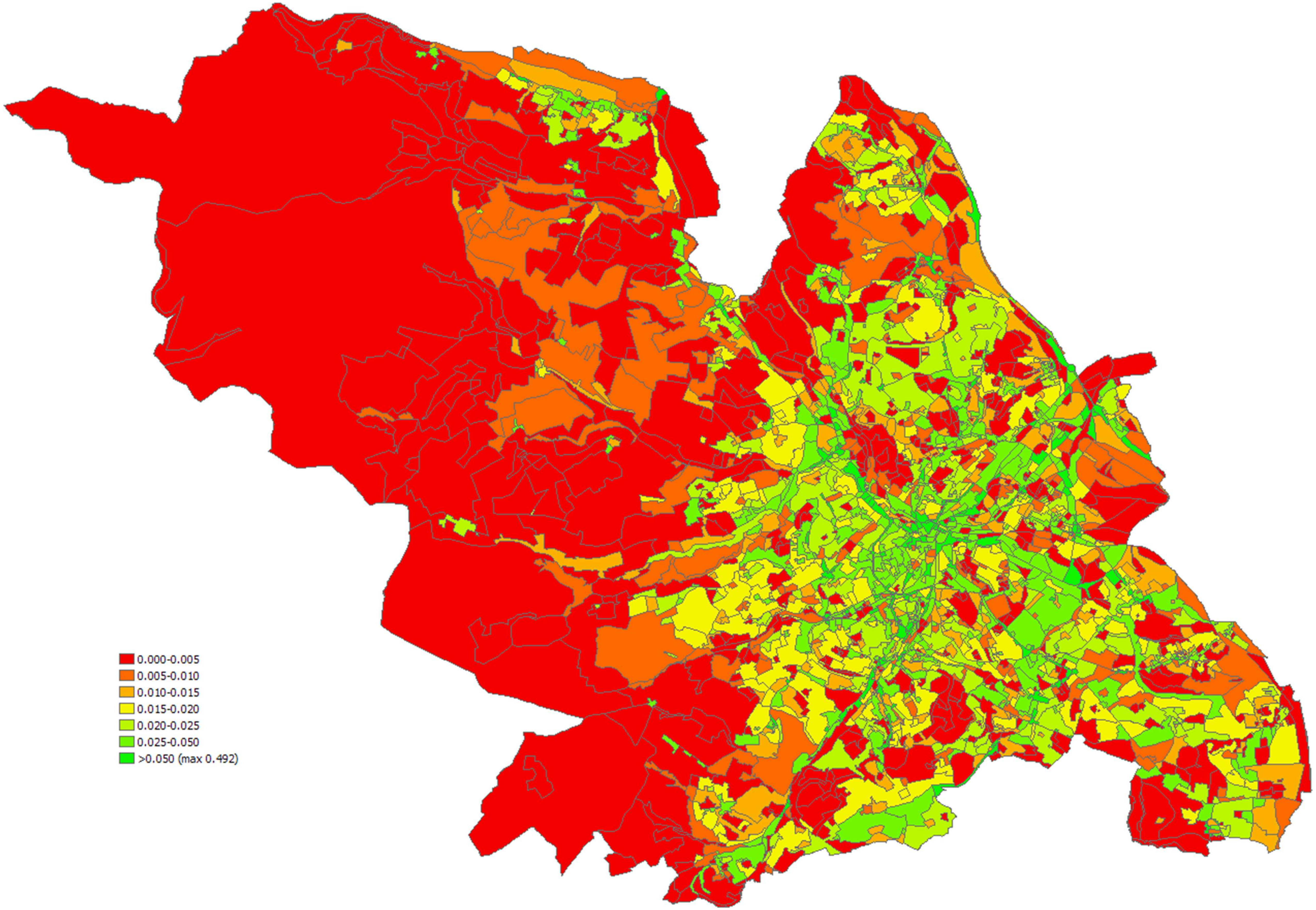
Map 32. Impervious surface normalised landscape shape index (nLSI) over Historic Environment Character areas. Note non-linearities in legend.



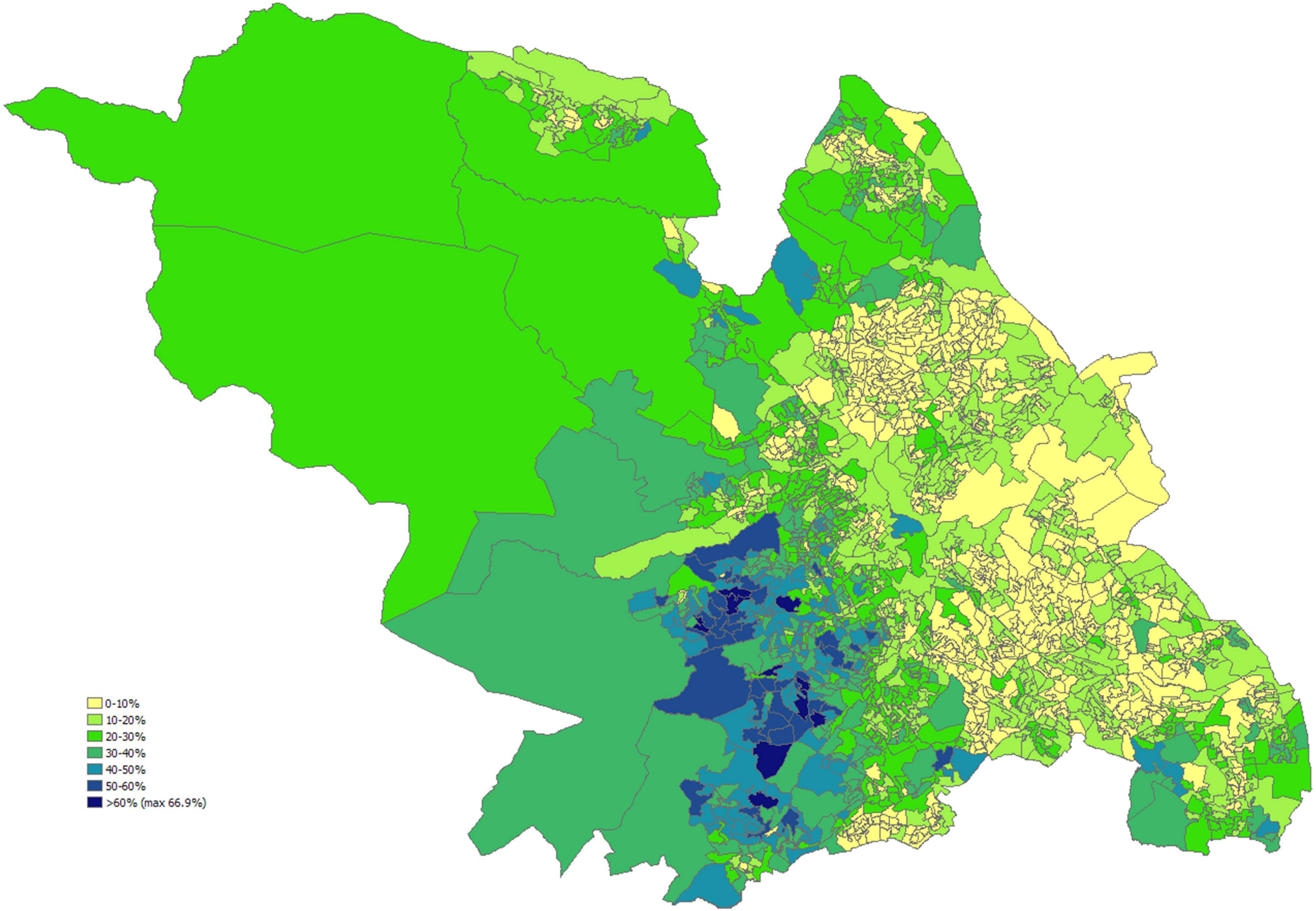
Map 33. Population density over Historic Environment Character areas. Note non-linearity in legend. (Units: people per m^2)



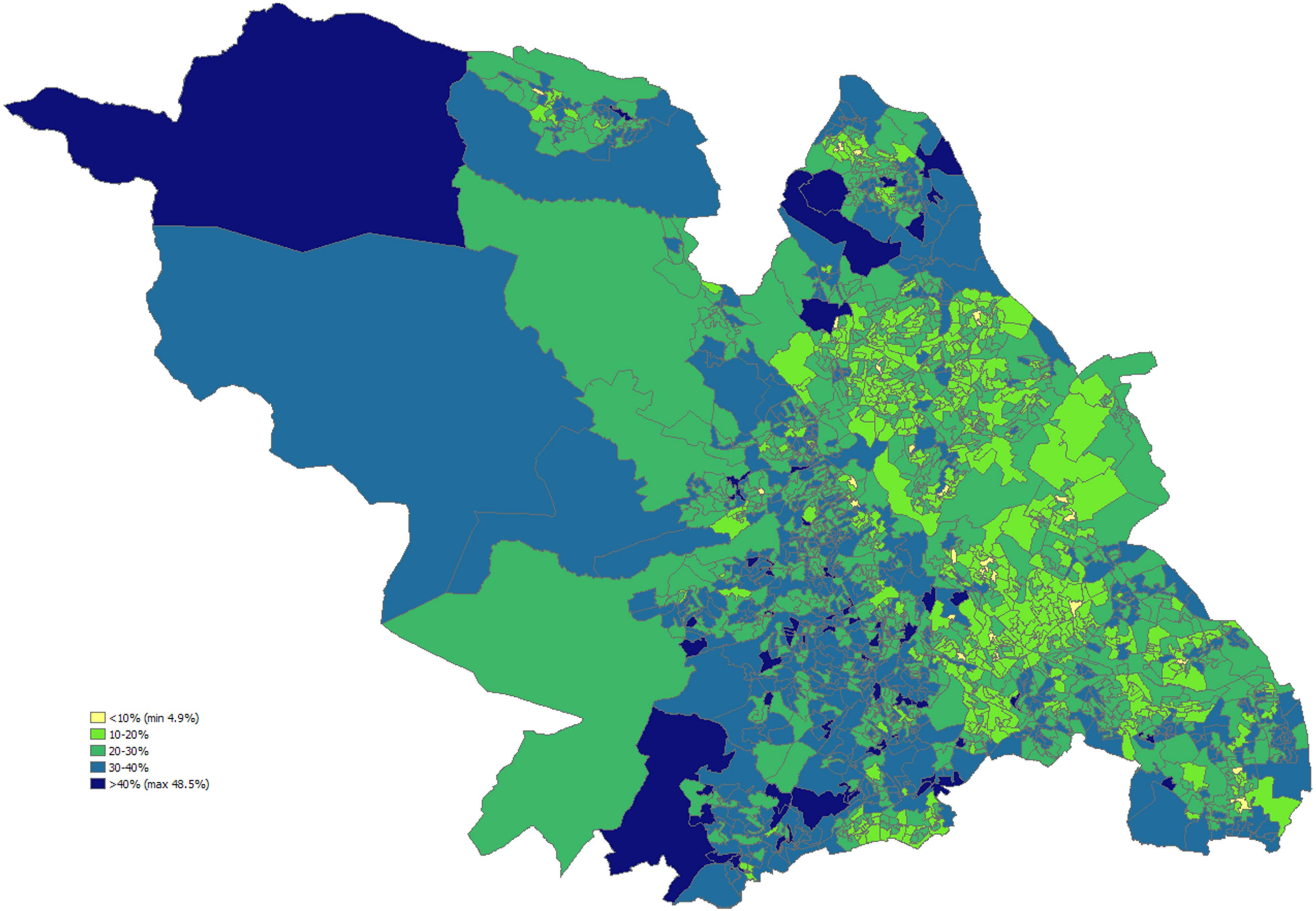
Map 34. Proportion households in detached or semi-detached houses (proportion detached houses) over Historic Environment Character areas.



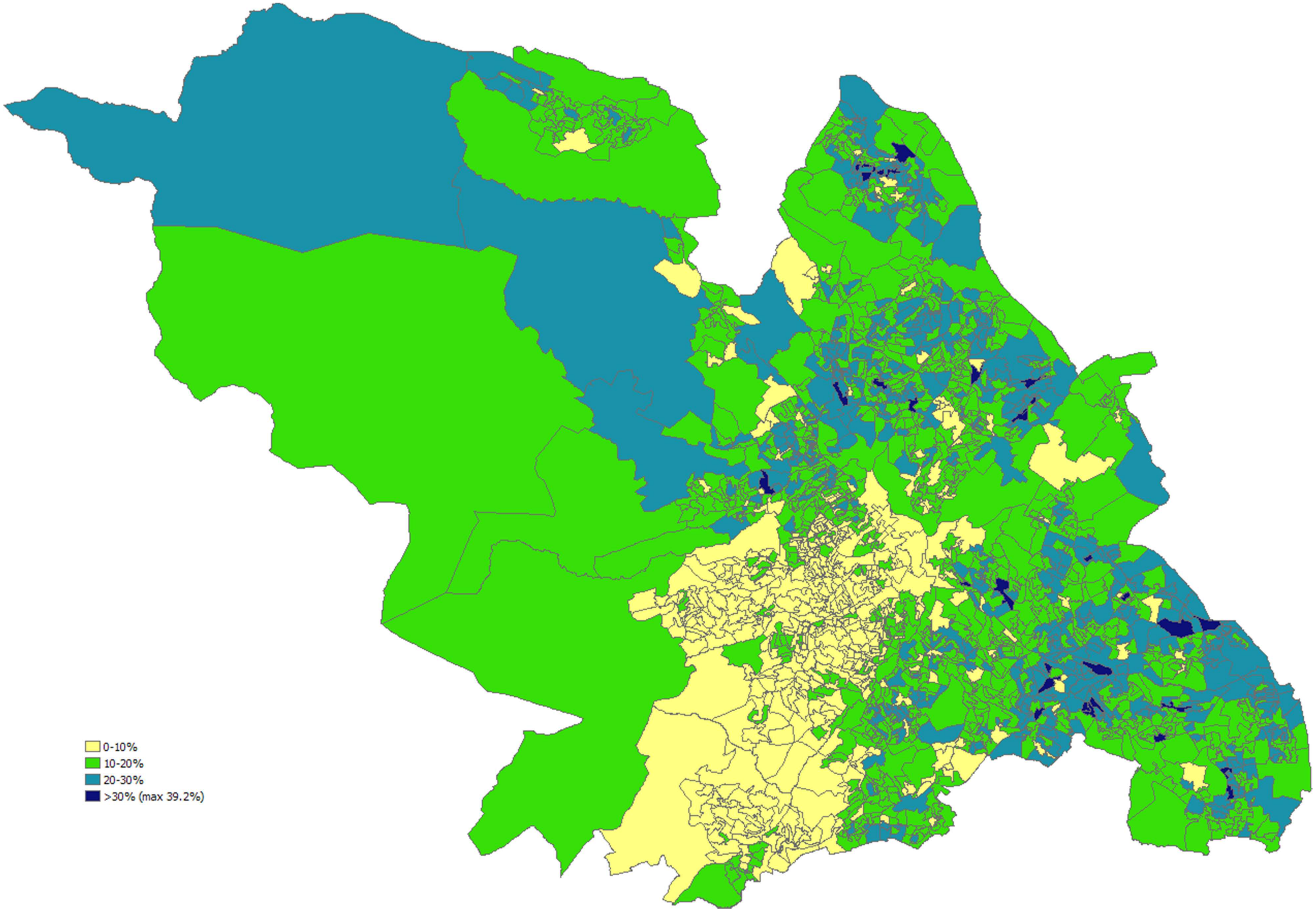
Map 35. Weighted road length density index (LDI) over Historic Environment Character areas.



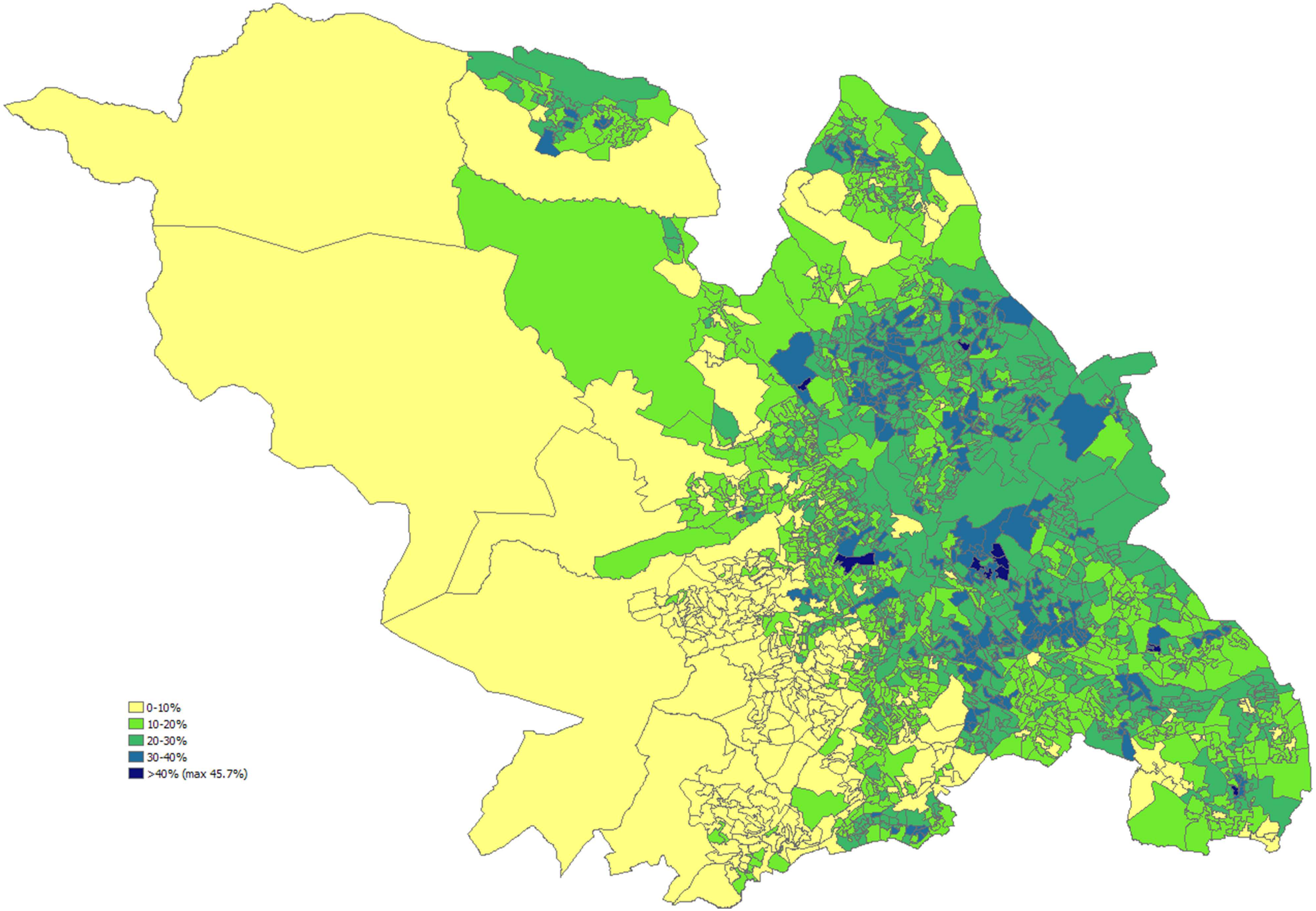
Map 36. Proportion of population (16 years+) in Output Areas belonging to approximated social grade AB.



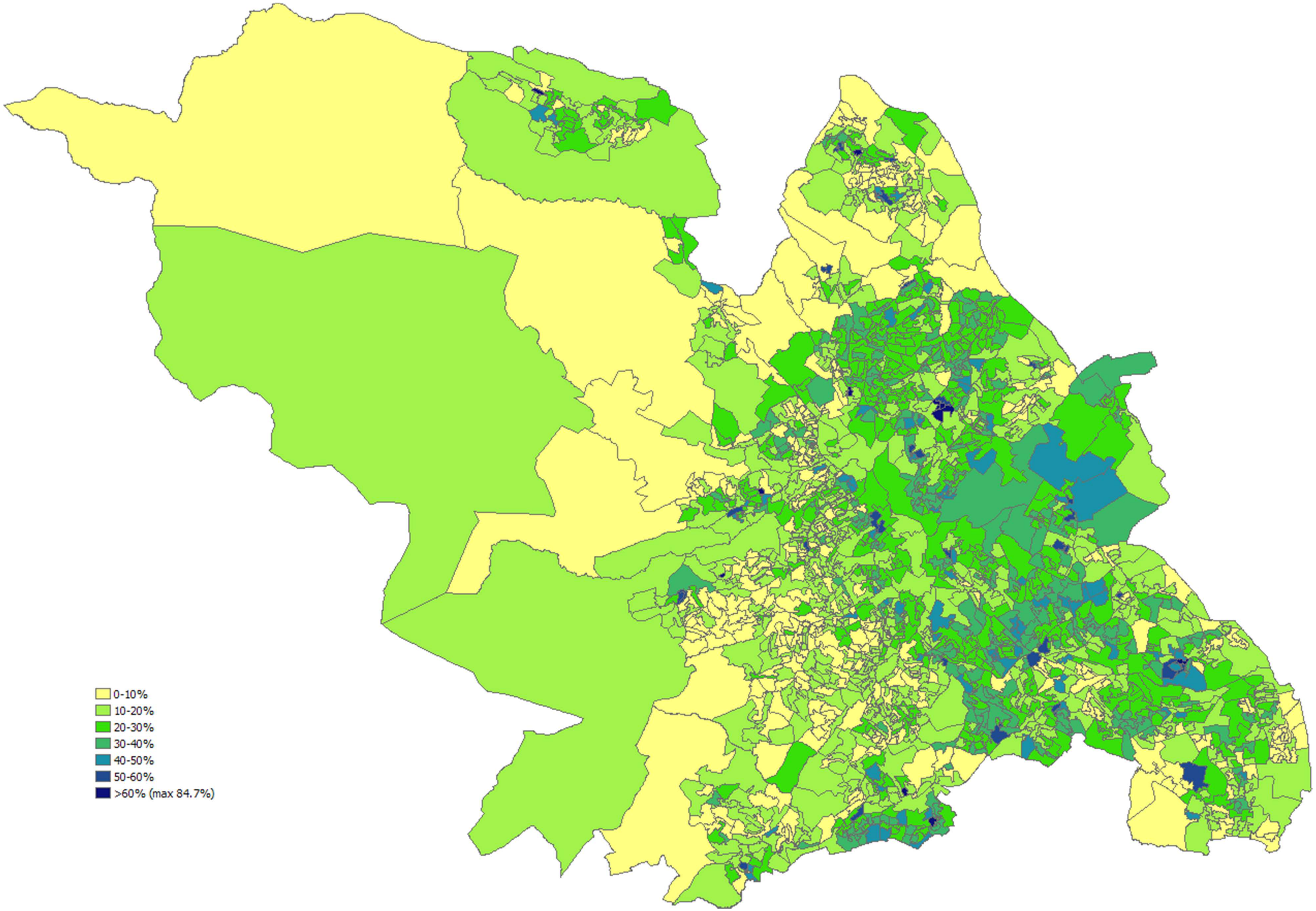
Map 37. Proportion of population (16 years+) in Output Areas belonging to approximated social grade C1.



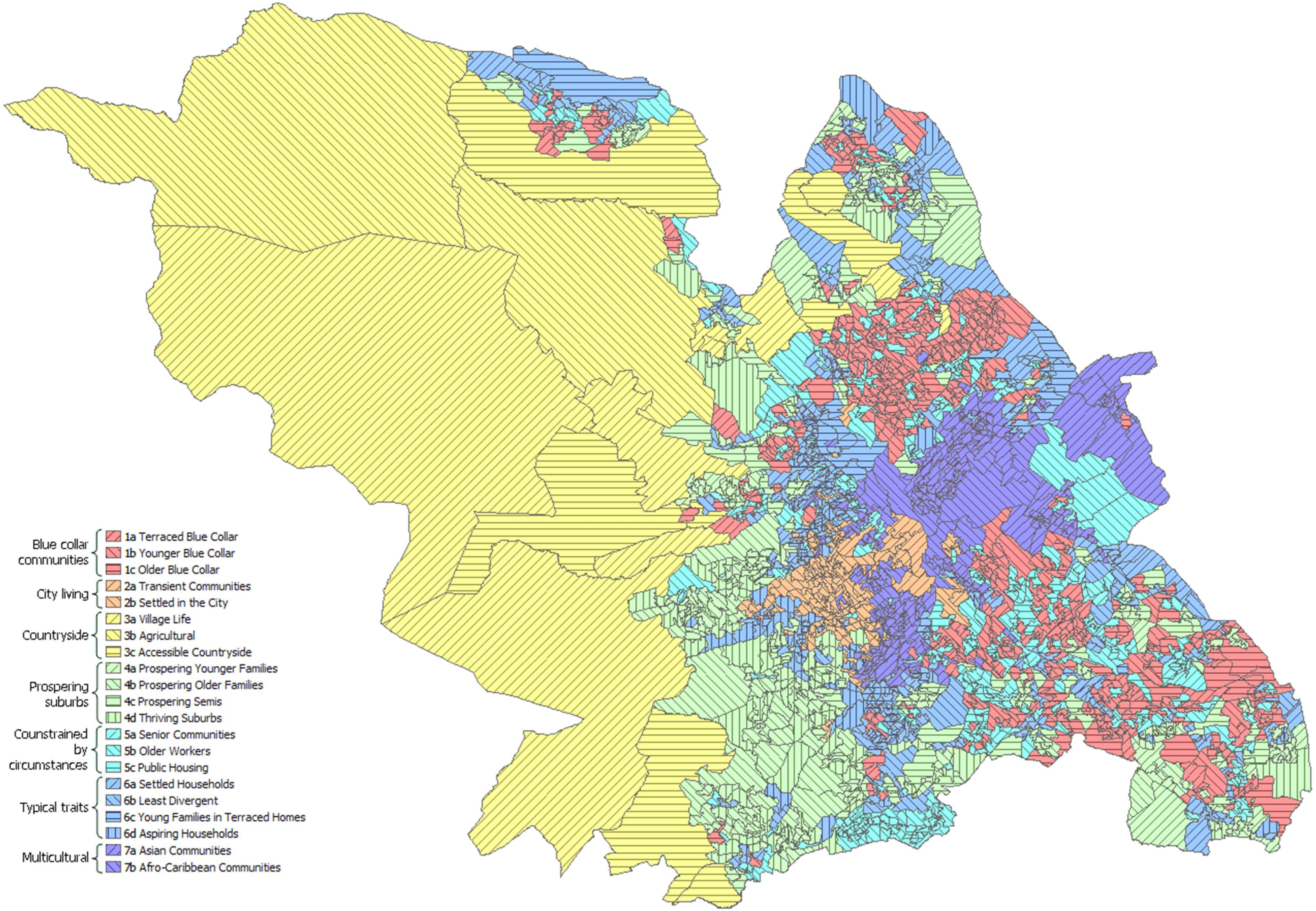
Map 38. Proportion of population (16 years+) in Output Areas belonging to approximated social grade C2.



Map 39. Proportion of population (16 years+) in Output Areas belonging to approximated social grade D.



Map 40. Proportion of population (16 years+) in Output Areas belonging to approximated social grade E.



Map 41. Area classification of Output Areas for 2001 Census data. Colours indicate supergroup membership while hatching differentiates groups within supergroups.