# Chapter 4
# Environment and Biodiversity

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## Introduction

Research over many decades has documented numerous environmental effects of urbanization, ranging from the loss and reconfiguration of green space to dramatic changes in ecosystems and biodiversity. Rather less is known about how urban form, in particular the density of urban development, alters environmental patterns and processes *within* cities. Investigation of the relationships between urban form and environmental structure and performance is an important issue in the urban sustainability debate and here we use that work to illustrate some of the key ideas in this newly emerging field. After outlining the general effects of urbanization on environment and biodiversity, we then consider in turn the relationships between urban form and patterns of green space, the degree to which urban environments can provide useful ecosystem services to human populations, and finally the responses of biodiversity to urban development.

Urbanization transforms the ecology of an area. Such transformation can include: (i) the alteration of habitat, such as the loss and fragmentation of natural vegetation, and the creation of novel habitat types (Niemelä, 1999; Pickett et al., 2001; McKinney, 2002; Johnson and Klemens, 2005); (ii) the alteration of ecosystem services (e.g. air, water and climate regulation, pollination), and other resource flows, including reduction in net primary production, increase in regional temperature, and degradation of air and water quality (Henry and Dicks, 1987; Rebele, 1994); (iii) the alteration of disturbance regimes, typically an increase in disturbance frequencies (Rebele, 1994); and (iv) disruption of species occurrence and abundance patterns, commonly including the local extinction of many species that are habitat specialists, require large habitat patches, utilise the interiors rather than the edges of patches, or are associated with complex vegetation structures (Pickett et al., 2001; McKinney, 2002; Chace and Walsh, 2006). The extent and intensity of these effects depend largely on the extent, composition and management

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of green spaces in urban areas. However, both variation in these characteristics and the details of their interactions particularly with ecosystem services and biodiversity remain rather poorly understood. This is perhaps surprising, given that both the characteristics of green space and its ecological correlates also bear on a number of other important issues. In particular, the extent, composition and management of green space has been shown to have significant effects on urban economies, through a diverse array of impacts, including on house prices, the costs of heating and cooling buildings, and the ease of attracting businesses and employees to areas (e.g. CABE Space, 2004), and on human physical, mental and social well-being (Dunnett and Quasim, 2000; Kuo and Sullivan, 2001; de Vries et al., 2003; Fuller et al., 2007a).

A useful step in developing an improved understanding of the structure of green space and some of the benefits that it brings within urban areas is to determine how these depend on urban form (the structure of the built environment). At the most simplistic level, the structure of urban green space might be viewed as the converse of urban form. Indeed, both measures of green space and of the built environment have been identified as key environmental indicators in urban areas (e.g. Pauleit and Duhme, 2000; Whitford et al., 2001). However, for a given ratio of green space to coverage by impervious surface, the nature and density of the built form can vary widely, such that there is a great deal of scatter in the relationship between measures of green space cover and urban density (Tratalos et al., 2007a). Further, the spatial configuration of habitat patches, rather than simply the degree of coverage by green space, can be important in determining biological processes for certain groups of organisms and types of ecosystem function (Bastin and Thomas, 1999). Links between coverage by green space and the nature of urban form have not been well explored (Pickett and Cadenasso, 2006).

There is considerable variation in the structure of cities, with local and regional factors heavily influencing urban form. For example, Sheffield, one of the case study cities, has a long history as a market town with a distinctly urban character (Hey, 2005). The population grew from 10,000 inhabitants in 1736–83,000 in 1851, and 90,000 by 1901. In the second half of the nineteenth century steel manufacture became the major industry and remained so for nearly one hundred years. Access to water for this industry dictated much of the pattern of early urbanization, and prevailing winds from the west meant that affluent residential areas were typically sited in the western suburbs with cleaner air, leading to a westward sprawl of the city boundary and increased population growth (currently 513,000). During the 1970s and 1980s, manufacturing industry began to shrink drastically, the economy diversified, and by the mid-1990s two-thirds of jobs were in the service sector. Substantial areas were redeveloped under regeneration programmes that replaced many industrial sites with housing or office blocks (Hey, 2005). This history has, in part, shaped the configuration of urban Sheffield today, with the most heavily developed areas in the river valleys, forming a y-shaped pattern. Large green spaces are restricted to the zones of intermediate and low levels of urban development, and a band of high density housing is noticeable between the centre and the outer suburbs (Fig. 4.1). In the context of the environment and biodiversity, urban form is perhaps best measured in terms of the density of various elements of urbanization,


**Fig. 4.1** Satellite image of Sheffield. Image of Sheffield acquired June 2005, 17° off nadir angle, cloud cover 1%. The area shown is all Ordnance Survey 1 $km^2$ national grid squares, inside the administrative boundary which comprise 10% or more impervious surface (buildings, roads or other sealed surface)

the patterns of coverage of different land use types, and the degree to which different patches of land cover are connected to each other. Insofar as the road network alters the configuration of green spaces by dividing them into smaller fragments, and might form a barrier (or a conduit) for dispersal of animals and plants, the physical structure of the road network is an important component of urban form as it impacts environmental performance. Although they may remain of interest, the effectiveness or efficiency of other measures of connectivity, as reflected for example in social cohesion and the ease with which people can move between areas using transport networks, are of less direct relevance, because they do not bear directly on the amount and configuration of patches of usable habitat.

In this chapter, we review aspects of the relationships between urban form and green space extent, ecosystem service provision and biodiversity. In so doing, we use examples drawn principally, although not exclusively from empirical studies of these issues conducted within the five case study cities.

## Green Space and Urban Form

Broadly defined to include most soft rather than hard or impervious surfaces, local government statistics often indicate that green space covers a substantial proportion of most cities in the UK. However, values based on the administrative boundaries of cities are difficult to interpret as these often extend well beyond the actual limits of urbanization and may include substantial rural areas. For example, urban areas form only about one-third of the area within Sheffield's administrative boundary, while another third is agriculture, and the remaining area forms part of the Peak District National Park (Beer, 2005). To generate more comparable data, we define the urban area simply as all the 1 km$^2$ grid squares inside the administrative boundary that comprise 10% or more sealed surface (buildings, roads or other human-made surface). Within this urban area, green space covers 60–70% if gardens are included, and 30–45% if gardens are excluded (Table 4.1). The data in Table 4.1 make two interesting points. First, the figures are remarkably consistent across the five study cities, despite their very different histories and topographies. Secondly, the figures are perhaps surprisingly high; a substantial majority of the surface within urban areas comprises green space, if gardens are included (see also Pauleit and Duhme, 2000). In this context it is important to note that these analysis are based on high resolution vector mapping, which includes even very small areas of green space such as road verges and amenity plantings. Given that small patches of green space make a disproportionately large contribution to overall levels of urban green space (see below), analyses of green space coverage that use mapping at a coarser scale might substantially underestimate its extent. In addition, these figures treat gardens as purely green space, whereas a proportion of the coverage of most gardens comprises impervious surfaces (e.g. in Sheffield, about one-third of garden area comprises impervious surfaces such as paths, patios, etc.; Tratalos et al., 2007a).

**Table 4.1** Total Green Space Coverage within the Five Case Study Cities

| City | Total urban area (km$^2$) | Total green space (km$^2$) | Gardens (km$^2$) | Non garden green space (km$^2$) | % coverage by green space incl. gardens | % coverage by green space excl. gardens |
|---|---|---|---|---|---|---|
| Edinburgh | 124.02 | 80.75 | 27.63 | 53.12 | 65.1 | 42.8 |
| Glasgow | 197.60 | 120.27 | 32.47 | 87.80 | 60.9 | 44.4 |
| Leicester | 70.09 | 41.84 | 18.88 | 22.96 | 59.7 | 32.8 |
| Oxford | 37.28 | 25.06 | 8.18 | 16.88 | 67.2 | 45.3 |
| Sheffield | 158.93 | 104.61 | 39.56 | 65.05 | 65.8 | 40.9 |

Green space is defined as any land parcel classified as "natural surface" by Ordnance Survey's MasterMap dataset (Ordnance Survey 2006), while gardens are those parcels classified as "multiple". The urban area is defined as that area inside the city's administrative boundary intersecting Ordnance Survey 1 km$^2$ national grid squares with 10% or more of their area comprising impervious surfaces (buildings, roads or human made surface).

A detailed grid-based analysis of green space across the five case study cities (Fig. 4.2) reveals that in each case coverage by green space is highly heterogeneous, with a long left tail of areas with very low levels of green space coverage, and a significant number of cells with almost complete coverage, representing large parks, well-wooded remnants and encapsulated patches of undeveloped land. The general pattern is remarkably consistent across the five cities, with the peak in proportional coverage generally occurring around 0.6. The frequency of cells with very high coverage varies markedly though, with Edinburgh and Oxford showing distinct peaks, indicating large patches of internal green space within those cities,

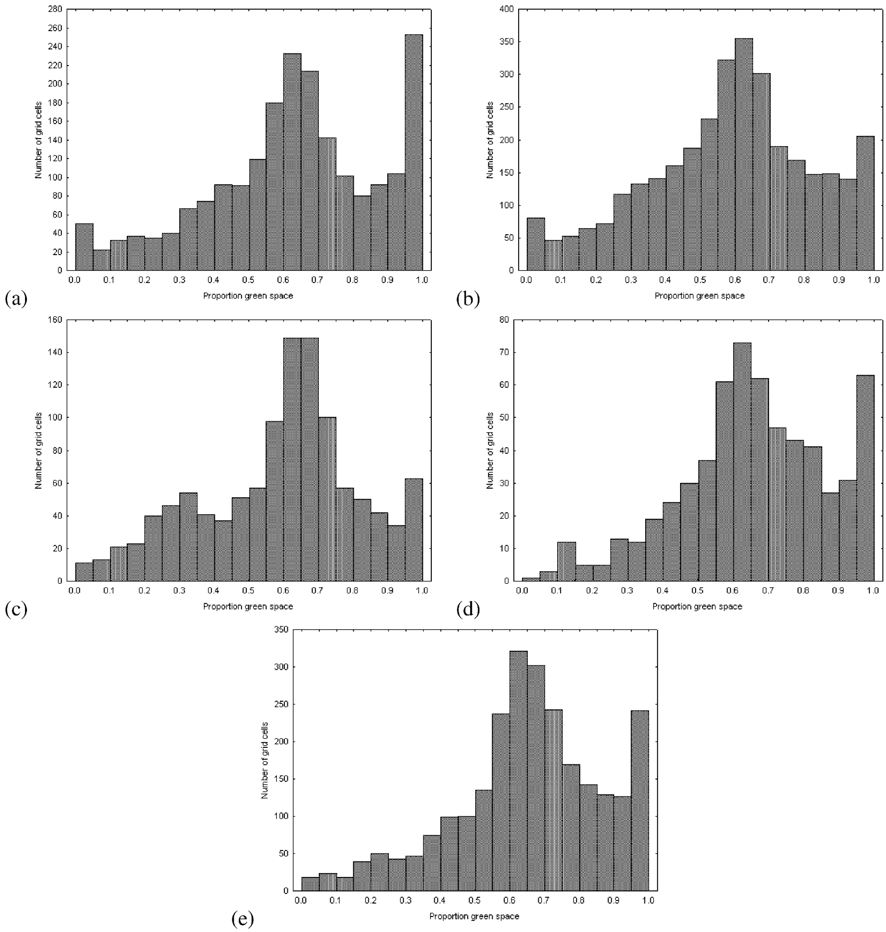


**Fig. 4.2** Proportions of green space across case study cities at 250 × 250 m cell resolution (**a**) Edinburgh, (**b**) Glasgow, (**c**) Leicester, (**d**) Oxford and (**e**) Sheffield. Green Space is vegetated surface including domestic gardens

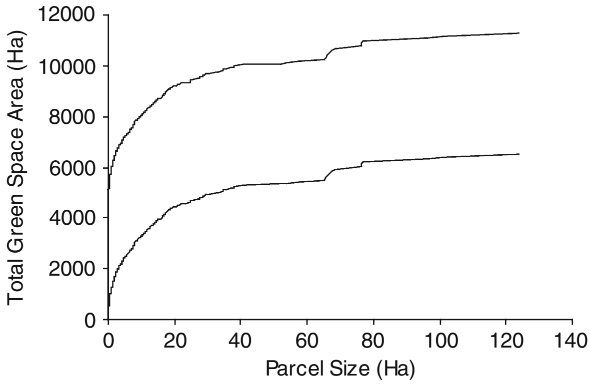


**Fig. 4.3** Cumulative amount of green space in Sheffield as parcels are added beginning with the smallest. The lower line excludes garden parcels

and Glasgow and Leicester showing a much less pronounced peak indicating that few large patches of green space exist within the boundaries of those cities.

A high proportion of urban green space in UK cities is composed of small patches, such as domestic gardens and roadside verges, rather than large patches, such as public parks and playing fields. For example, there are 326,147 separate parcels of green space within the Sheffield urban area, of which 93% are front or rear garden. When arranged in rank order of increasing area, the contribution to overall cumulative area of green space decelerates strongly as parcel size increases, a pattern that persists even when gardens are excluded (Fig. 4.3). Across urban Sheffield, 50% of green space comprises the parcels with an area less than 0.59 ha. Excluding gardens, the equivalent figure is 9.9 ha. This is significant, as much strategic planning of urban green space is focussed on the large green spaces, and ignores the smaller ones. In practice, however, the vast network of small patches may make substantial contributions to the net environmental benefits provided by urban green space, and may be vital to maintaining the contributions of some of the larger patches (e.g. through influences on overall habitat availability and on the degree of connectivity among larger patches). Without the smaller patches the larger ones would genuinely resemble the isolated habitat islands that strategic planning often portrays them to be.

The size of a patch of green space is an important variable in ecological terms, because larger patches tend to support more species and individuals, buffer those populations from local extinctions, and allow the organisms within them to disperse across the landscape (Bastin and Thomas, 1999). As well as overall green space, average patch size tends to decline with increasing building/housing density, with some evidence that the relationship is markedly non-linear, being steeper at lower densities (Tratalos et al., 2007a). This effect may be exacerbated by the larger numbers of land parcels in areas of higher density. Crucially though, the ecological impact of changes in habitat patch sizes depends on whether habitat characteristics of patches also vary with their size, the minimum patch areas that particular species can tolerate, and the ability of species to disperse across the landscape to other patches (Watson et al., 2005; Bierwagen, 2007).

Green space is not distributed evenly within cities (Fig. 4.4a, b). Yet, despite variation in geographic settings, socio-economic and developmental histories, topographies and sizes, there are often marked similarities between cities in this distribution, and some common correlates with key measures of urban form. For example, across the five case study cities the overall extent of green space tends to increase with elevation and topographic slope, and decline with number of road nodes, length of roads, and the density and area of housing and other buildings (Davies et al., unpublished data). The slopes of these relationships are often broadly similar between these cities. The relationships between overall green space and building/housing density result in large part from the net trade-off between two processes, the decline and the increase respectively of non-garden and garden green space with increasing housing density. Indeed, this is arguably a key driver of patterns of green space in urban areas. This is particularly apparent for relationships between green space coverage and the density of given housing types (flats, terraced, semi-detached, detached; Fig. 4.4i–l), with the net balance between the two processes shifting, such that for flats it is dominated by the loss of non-garden green space whilst for detached houses it is dominated by the gain of garden green space.

Whilst interesting patterns are evident in the distribution of green space treated as a whole, it is also clearly the case that all green space is not the same. One marked difference is in the occurrence of trees. Trees can contribute to some, though not all, of the key advantages of green space disproportionately to the area they occupy on the ground because of their complex structure that includes a canopy. This may be important in urban areas where ground space is at a premium. Therefore the numbers and distribution of trees are potentially important aspects of greenspace provision in urban areas. In what seems likely to be a general pattern, across Sheffield there is an approximately triangular relationship between tree cover and extent of green space, such that whilst the maximum amount of tree cover increases with the extent of green space, below this limit the full range of variation in tree cover is exhibited (Davies et al., 2008). There are also marked declines in tree cover with increasing housing density (Iverson and Cook, 2000). This is a particular concern at the present time given that the occurrence of trees in individual private gardens declines as those gardens become smaller (Smith et al., 2005), and that decreases in garden size are occurring in urban areas of the UK as a consequence of increased densification both in the form of high density new developments and infilling development.

Such densification, both in the UK and elsewhere, also results in general reductions of green space (e.g. Pauleit and Duhme, 2000; Pauleit et al., 2005; Yli-Pelkonen and Niemelä, 2005). This occurs both as a consequence of infill development in areas which historically have had lower densities of buildings, and through development in those areas which recently have had higher levels of brownfield land. For Sheffield, a trend of increasing urban density is apparent from a generally positive relationship between the date of urbanization and current levels of green space (Fig. 4.5a). However, a flattening of the relationship since the

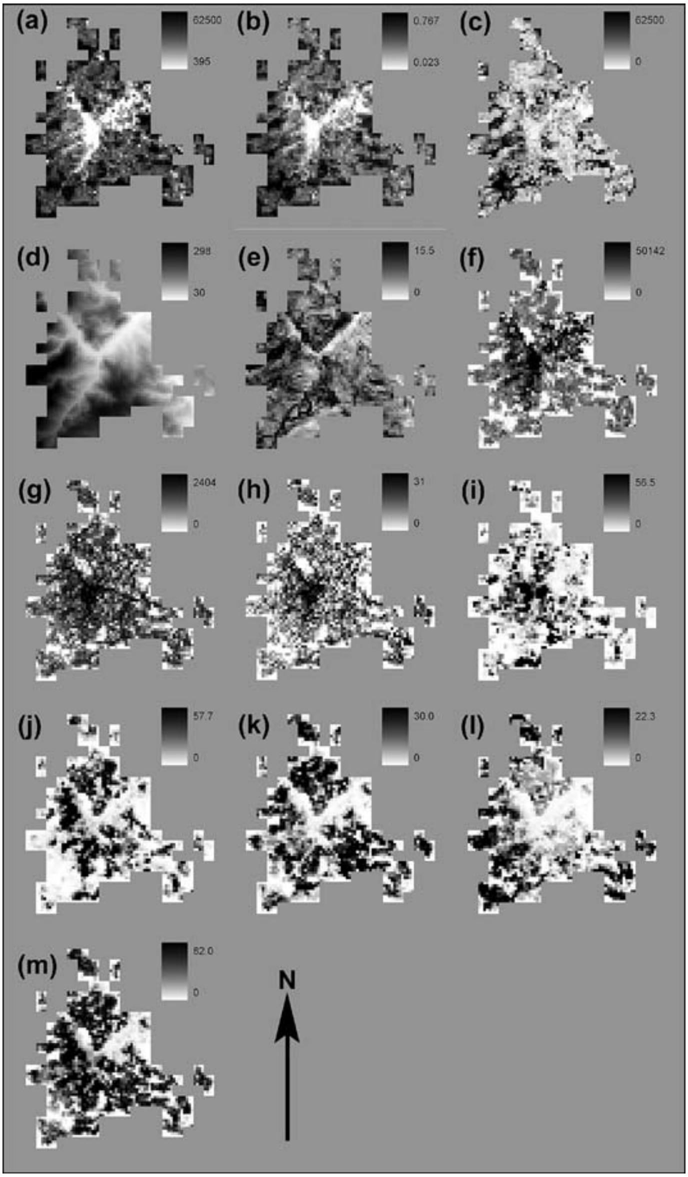


**Fig. 4.4** Environmental Surfaces for Sheffield at 250 × 250 m cell resolution: (**a**) extent of green space ($m^2$); (**b**) vegetation cover (Normalized Difference Vegtation Index: NDVI); (**c**) tree-cover ($m^2$); (**d**) elevation (m); (**e**) degree angle of slope; (**f**) buildings area ($m^2$); (**g**) road length (m); (**h**) road nodes (junctions); (**i**) density of households in blocks of flats; (**j**) terraced housing density; (**k**) semi-detached housing density; (**l**) detached housing density; and (**m**) total housing density. All housing densities were measured as households $ha^{-1}$

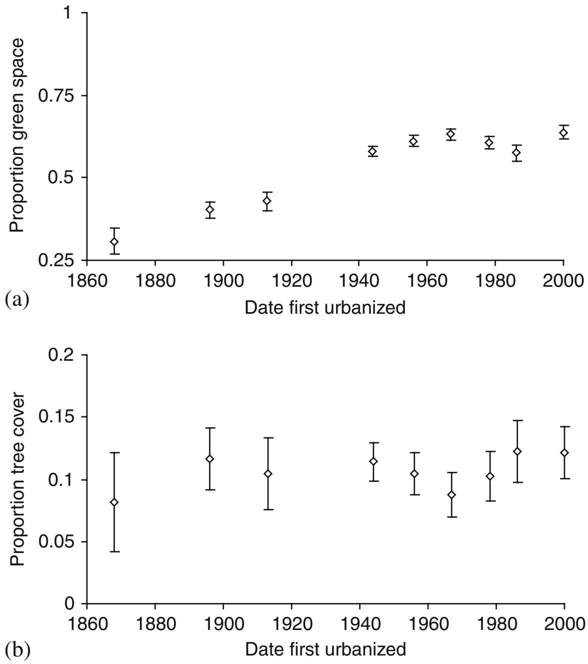


**Fig. 4.5** The Relationship between Date of first Urbanization and the Proportion of each 250 × 250 m cell in Sheffield that currently comprises (**a**) Green Space and (**b**) Tree Cover. Date first urbanized is the date of the first Ordnance Survey map series for which each grid cell became dominated (>50% coverage) by urban built form, and as such represents the latest date at which urbanization was known to have occurred. Grid cells not yet urbanized are excluded from the plot. Green space is vegetated surface including domestic gardens. Tree cover is derived from digitising tree cover from aerial photographs taken in 1998

1960s could indicate that recently developed areas have suffered a disproportionate loss of green space. An alternative explanation for this pattern is that urbanization of an area results initially in the loss of a certain proportion of green space, which is then gradually further eroded over time. Tree cover shows a much less pronounced, and more variable relationship with date urbanized (Fig. 4.5b). This suggests that, while more recently developed areas have similar levels of tree cover, a greater proportion of it is over impervious surface. In Sheffield, a tendency for larger trees in the western part of the city partly results from their being planted in the nineteenth century during planting initiatives associated with creation of parks and avenues. Tree cover as seen in aerial imagery is therefore probably associated with age of urbanisation, since a mature species will have a large crown area.

## Ecosystem Services and Urban Form

Recent emphasis on assessing the value of the environment in terms of the benefits that ecosystems provide to humans (ecosystem services), has resulted in increasing interest in evaluating the role of urban areas in providing ecosystem services (e.g. Bolund and Hunhammar, 1999; Pataki et al., 2006; Tratalos et al., 2007a). This is partly because local provision of some of these services is valuable to human communities in those areas, for example services such as temperature and water regulation, pest control, pollination, and recreation, all have implications for urban economies and human well-being. It is also partly because although urban systems have traditionally been accorded rather low value where environmental metrics focus on naturalness and lack of human influence, this does not necessarily mean their contribution to other types of ecosystem service such as carbon sequestration, is also low.

In order to examine the relationship between ecosystem service provision and urban form within and between cities we largely take an approach based on simple general models of the relationships between variables on the ground that can be measured across entire cities, and specific services. This allows patterns to be explored, but may not pick up all the local, and city-specific, detail that direct measurement of the services themselves would allow. However the data demands of the latter at the scales of comparison we are interested in here make the general approach the only one practical at the present time for most services.

## *Temperature Regulation*

Urban areas experience heat island effects, in which temperatures are elevated compared with surrounding landscapes, particularly at night and in cold weather (Pickett et al., 2001; Baker et al., 2002). Although they are a function of several factors, across urban areas both air and land surface temperatures tend to increase with proportional coverage by impervious surfaces and to decrease as proportional coverage of green surfaces increases (Chen and Wong 2006; Jenerette et al., 2007). Levels of tree cover in particular can have a marked influence on temperatures, in major part by generating energy loss, and therefore cooling, through water loss to the atmosphere (Stone and Rodgers, 2001).

As well as wholesale changes in temperature regimes wrought by urbanization, variation in urban form within a built up area can have significant thermal impacts. Data from an array of temperature loggers placed across urban Sheffield revealed that elevated temperatures in both summer and winter were associated largely with the city centre, declining toward the less built up suburbs (Fig. 4.6a, b). This effect was still apparent at a local scale in the environs around each temperature logger; there was a positive relationship between temperature and impervious surface coverage within 100 m of a location, in both summer and winter (Fig. 4.7a, b).

Diurnal temperature range also varied markedly, with areas near the city centre showing a smaller difference between minimum and maximum daily temperature (Fig. 4.6c, d), although local coverage by impervious surface was not related to daily temperature range (Fig. 4.7c, d). These general patterns demonstrate marked buffering of temperatures in heavily developed areas, and show that the effect persists throughout the year. It is crucial to note, however, that while the amount of heat energy emitted per unit area might be higher in places dominated by impervious surfaces, the per household contribution to these emissions depends

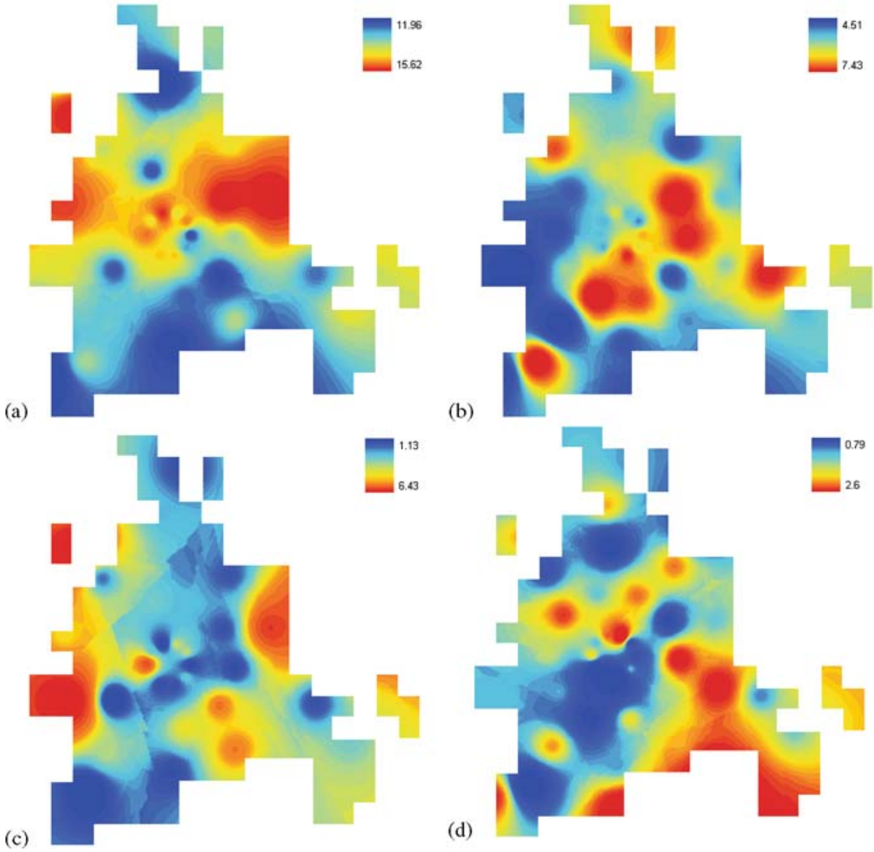


**Fig. 4.6** Temperature surfaces across Sheffield (**a**) Mean summer temperature, (**b**) mean winter temperature, and mean diurnal temperature range in (**c**) summer and (**d**) winter. °C interpolated from hourly data collected by an array of 50 temperature loggers arranged in five concentric rings of 10 loggers each centred on the city centre. Loggers were buried in soil at a depth of 20 mm, and data were downloaded every three months

on the absolute coverage by impervious surface within the housing parcel (Stone and Rodgers, 2001). Although smaller housing parcels tend to have a higher proportional coverage by impervious surface (Smith et al., 2005), this is unlikely to cancel out the reduction in absolute coverage by impervious surface as housing parcel size declines. Multi-dwelling buildings will further reduce the per household contribution to urban heating. Consequently, although higher density residential urban forms might generate higher levels of heating per unit area, the total amount of heat energy released from such developments is likely to be less than that produced by a low density neighbourhood comprising the same number of households.

Temperature changes through urbanization can have a variety of ecological consequences, some of which extend beyond the urban area itself, including changes

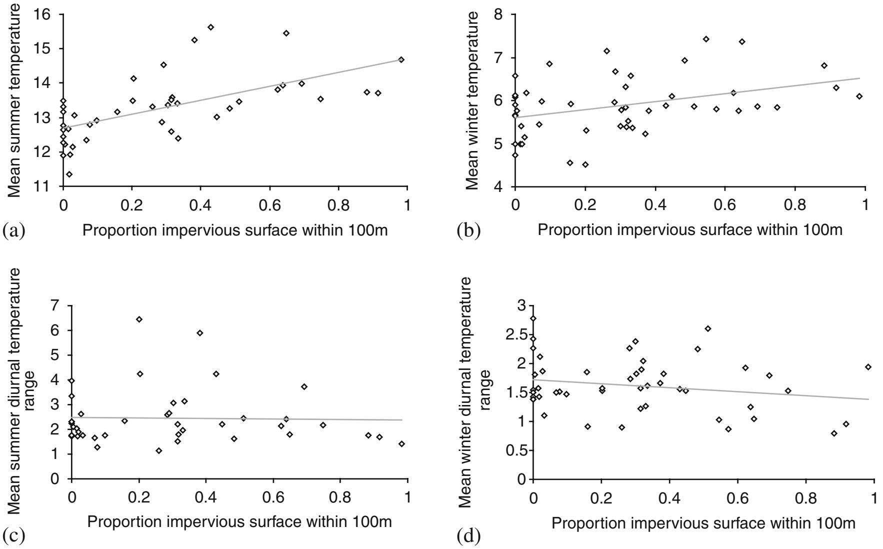


**Fig. 4.7** Relationships between Cover by Impervious Surface within a 100 m radius of each Temperature Logger and (**a**) Mean Summer Temperature (**b**) Mean Winter Temperature, and Mean Diurnal Temperature Range in (**c**) Summer (**d**) Winter. Based on linear regressions

relative to rural areas in the timing of germination, leaf flush, leaf drop, and flowering of plants, and in the breeding and survival of animals (Zhang et al., 2004a, 2004b; Partecke et al., 2005; Neil and Wu, 2006). It seems likely that similar changes may also occur within the boundaries of urban areas, and be particularly influenced by variation in coverage by green space, although disentangling the effects of green space per se and its influences on temperature may not be straightforward.

## *Water Regulation*

Cities in the UK almost invariably draw the bulk of their water supplies from outside the urban area, and have rather low coverage by standing water. However, urban areas have to deal with water influx, either directly on the area, or in rivers and streams that pass through it. Flood control and reduction of storm-water runoff are therefore key components in the ecosystem service of water regulation. Water runs directly over impervious surfaces, increasing the frequency and severity of urban flooding. In recognition of this, much research has focussed on the relationship between impervious surface coverage and water regulation (Arnold and Gibbons, 1996). However, the converse of this is that increasing the extent of green spaces enhances water regulation, providing that soil surfaces within those green spaces are not heavily compacted. Green spaces also increase water loss from the ground through evapotranspiration (the transfer by soil and plants of water to the atmosphere as water vapour), with additional beneficial effects on climatic conditions in urban areas.

The distribution of green space within cities therefore becomes crucial for explaining variation in water regulation across the urban landscape. Although urbanization dramatically reduces green space coverage, significant levels (60–70%) remain in many typical urban areas (Table 4.1). It has been estimated that 59% of the surface of urban Manchester is evapotranspiring (Gill et al., 2007). For the case study neighbourhoods across the five cities these reveal non-linear relationships with the extent of green space: runoff increases as greenspace declines, but at low levels of greenspace changes in the remaining greenspace have little effect (Tratalos et al., 2007a). This is a consequence of the relatively low infiltration rates of even many of the non-sealed surfaces when building densities become high.

As well as problems associated with the quantity of run-off generated in urban environments, transport of pollutants can occur as storm water washes over impervious surfaces such that suspended particulate matter within the run-off might include anthropogenically derived materials. Such contaminants can represent a significant non-point source of pollution in urban areas (Characklis and Wiesner 1997; Bibby and Webster-Brown, 2005).

## *Carbon Sequestration*

It is becomingly increasingly apparent that, at least in some regions, carbon sequestration in urban areas may not be as trivial a consideration as some have suggested (Jo and McPherson 1995, Golubiewski 2006; Pataki et al., 2006). Indeed, whilst obviously small compared with carbon emissions the per unit area and the gross sizes of urban carbon pools (reservoirs of stored carbon) can nonetheless be substantial (Nowak and Crane 2002; Kaye et al., 2005; Lorenz et al., 2006). There are two major natural carbon pools in urban green spaces, comprising respectively vegetation and soils.

The levels of carbon sequestration by trees in urban areas, resulting from the fixation of carbon during photosynthesis and its storage as biomass, tend to be estimated using simple functions of tree cover (e.g. Rowntree and Nowak, 1991; Whitford et al., 2001; Tratalos et al., 2007a). Detailed mapping in Sheffield has shown that the surface of the city is about 15% tree-covered (see also Fig. 4.4c). This is notably lower than in some other cities in the northern hemisphere (e.g. Nowak and Crane, 2002). An analysis of carbon sequestration across the five case study cities indicated that more densely urbanized areas are associated with a lower predicted rate of carbon sequestration (Tratalos et al., 2007a). Because calculations for carbon sequestration were based on a linear relationship with tree cover (see Rowntree and Nowak, 1991), results for carbon sequestration can typically be interpreted as matching those for tree cover. However, while carbon sequestration will generally increase with tree cover, in practice the relationship will depend, among other things, on the demographic structure and species composition of the urban forest. This problem is compounded by the fact that tree cover will itself vary with the age of the properties which make up the urban form, and tree growth rates will vary according to soil compaction, pollution, impervious surface area under tree crown and water potential (Quigley, 2004).

A sample of 2170 trees from Sheffield indicates clear systematic patterns in tree species richness and size (Fig. 4.8). Broadly speaking, tree richness increased with distance from the city centre, along with an associated increase in tree size. The sample comprised 96 tree species, although the ten most abundant species accounted for 64% of all individuals recorded. Trees nearer the city centre tended to be taller relative to their girth than those on the margins of the city, although there was substantial variation in this (Fig. 4.8d). While some very mature trees were present in the sample, with girths of 3–4 m, the size distribution of the trees was strongly right-skewed (Fig. 4.9), with 50% of individuals having a girth <0.5 m. Continual

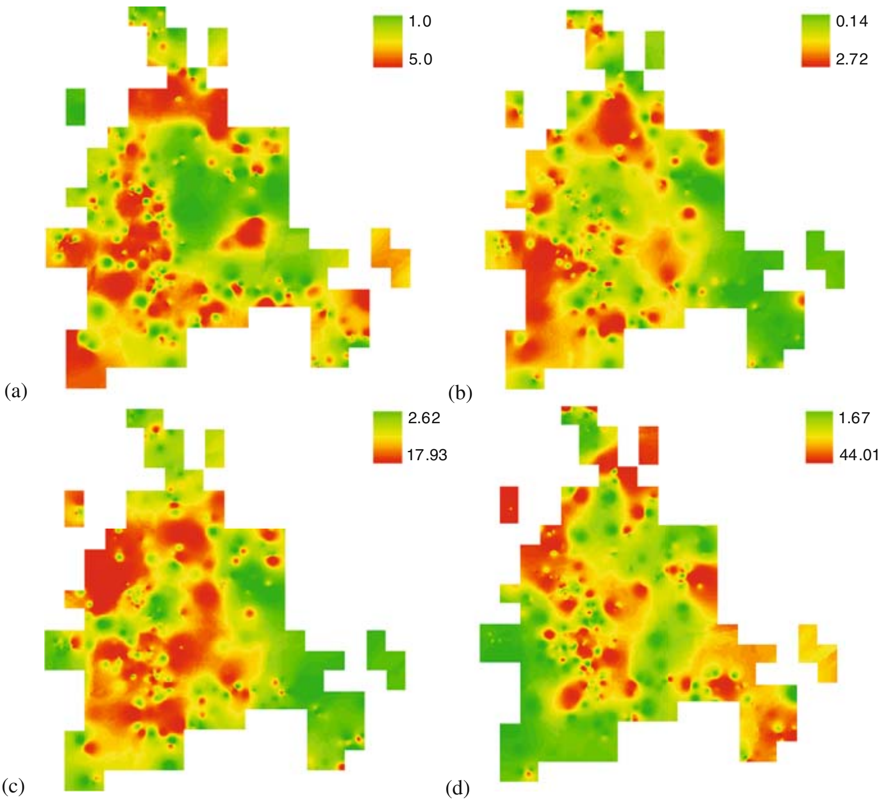


**Fig. 4.8** Surfaces for Sheffield of (**a**) Tree Species Richness, (**b**) Average Tree Circumference (m), (**c**) Average Tree Height (m) and (**d**) Height/Circumference. Data are derived from field measurements of 2170 trees (the five trees nearest to 434 randomly-chosen locations across the city)

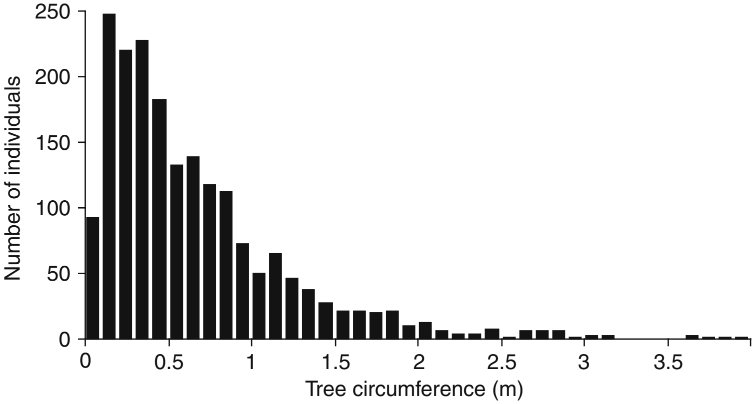


**Fig. 4.9** Distribution of Circumferences of Trees across Urban Sheffield. Data are derived from field measurements of 1924 trees selected as the five trees nearest to 434 randomly-chosen locations across the city (circumference could not be measured for 246 individuals because of access difficulties)

replacement of mature urban trees with smaller species that are more manageable, and less prone to cause damage through deadfall, windthrow and root intrusion has recently been documented in London (London Assembly Environment Committee, 2007).

Management of the urban forest is further complicated by the influence of land ownership. Recent data from Sheffield indicate that, within a 13 km$^2$ study area, approximately 69% of tree cover occurred on privately owned land (Dennis, unpublished data). About 73% of the land was privately owned, indicating that proportional levels of tree cover on private and public lands were broadly similar. These data highlight the very limited extent to which adequate management of the urban forest can emerge simply from policies focusing on land under direct public control. There is a need to recognise that much urban land is under dispersed ownership, with small private parcels representing domestic gardens being managed in markedly different ways (Gaston et al., 2005). Furthermore, smaller private gardens are less likely to contain trees, and there are concerns that continuing densification of urban areas through infilling development might lead indirectly to further reductions in tree cover through this route (Smith et al., 2005). In public lands, street trees may be removed for public safety reasons, or as a result of subsidence claims by insurance companies. Despite no change in overall tree numbers in a five year study period, data from London indicate a rapid turnover of street trees for these reasons, and an ongoing disproportionate loss of mature native trees (London Assembly Environment Committee, 2007).

The other significant carbon pool in urban areas is that contained within soils. The sizes of such carbon pools have been surprisingly little explored, in large part because urban soils (i) are typically extremely heterogeneous both spatially and temporally, comprising a mix of islands of apparently natural soils within a matrix of highly human-altered soils; (ii) have almost invariably been poorly mapped; and

(iii) often have altered processes of decomposition and nutrient cycling, for a range of reasons including the urban heat island effect (Effland and Pouyat, 1997; Pouyat et al., 1997; Carreiro et al., 1999; Pickett et al., 2001). Where soil carbon pools have been estimated in urban areas, this has typically involved extrapolation from data collated from just a handful of cores and tiny quantities of soil (e.g. Pouyat et al., 2006). Nonetheless, such work suggests that these pools may be substantial. Legal protection of the ecological functions of urban soils in Germany attests to this importance, despite the rather rudimentary current understanding of the properties of urban soil (Lorenz et al., 2006).

The significance of carbon sequestration in urban areas depends fundamentally on how vegetation and soils are managed. Carbon emissions associated with management (e.g. from chain saws, chippers, lawn mowers and transport of cut vegetation) could, for example, negate any positive sequestration effects, although the extent to which this is a problem can be influenced by the choice of management approaches and the fate of vegetation that is removed (e.g. landfill, bio-fuels). However, even where this is the case, urban trees in particular may typically bring a number of other advantages (e.g. control of storm water, reducing energy use in buildings and human health benefits), and a decided net benefit (Akbari, 2002; McPherson et al., 2005).

## *Pollination*

Many plant species are pollinated by insects, which transfer pollen from one plant to another. Some plant species cannot produce fertile seeds without pollination by insects, and the yield of most plants is improved where insect pollination occurs. Concern has been expressed about regional and local reductions in the numbers of pollinators in a variety of areas of the world (Biesmeijer et al., 2006; Klein et al., 2007). As well as impacting on agricultural crop production (Allen-Wardell et al., 1998), declines in pollinators adversely affect the functioning of natural and anthropogenically-perturbed ecosystems (Kearns et al., 1998; Cheptou and Avendaño, 2006; but see Ghazoul, 2005). Additionally, some people with private domestic gardens rely on, and derive economic benefit from, pollination of garden plants (Nabhan and Buchmann, 1997). Although several studies have documented declines in insect pollinators in response to urbanization, others have found elevated bee species richness in urban conditions, attributed to increased temperatures, reduced exposure to agricultural chemicals, and a wider variety of microhabitats being present in urban landscapes (Eremeeva and Sushchev, 2005 and references therein).

Pollinator faunas can also change systematically within urban areas. Across private gardens in Sheffield, the number of species of an important group of pollinators, bumblebees, is influenced both by immediate local and larger scale factors, increasing with the habitat diversity within the gardens and with the area of green space in the environs of the gardens (Smith et al., 2006a). However, no such relationships were apparent for the abundance of this group (Smith et al., 2006b).

As well as changes to pollinator faunas, impacts of urbanization on the distribution and population dynamics of plant species can affect plant-pollinator relationships. Fragmentation of the natural environment through urbanization frequently leads to small, isolated populations of plants, and diminished population densities, which can result in few pollinators finding a particular patch of plants. Cheptou and Avendaño (2006) studied an urban plant (*Crepis sancta*: Asteraceae), showing that the number of pollinating insects visiting a patch of plants increased as the number of plants in the patch increased. This suggests that where urbanization leads to a plant becoming rarer, visits by pollinators might also decline, potentially compounding the effect of urbanization. This will be particularly important for those plant species that are wholly dependent on insects for pollination, and cannot reproduce unless they are visited by pollinators. *Crepis sancta* is able to self-fertilize, but it was not able to increase its levels of selfing in response to a low number of visits by pollinators. More research is needed on the adaptations of plants to urbanization.

## *Provision of Pest Control*

Effective functioning of ecosystems depends on the maintenance of interactions between species, including competition, predation, parasitism and mutualism. By differentially altering the abundance of particular species as areas are progressively developed, urbanization often results in new combinations of species that have not previously interacted. There is little information on how species interact in such new combinations, and how these interactions vary across urban landscapes. In one well studied example, urbanization of Phoenix, Arizona, led to a dramatic increase in available water, a limiting resource in the surrounding desert landscape. This has resulted in a shift in species interactions, including a greater top-down influence of predators on the system (Faeth et al., 2005). Urbanization can also lead to asymmetric changes in the abundances of particular species, such as rapid increases in species that might be regarded as “pests” (Alberti et al., 2003).

To assess the potential for such changes in response to increases in urban density, recent work in Sheffield examined rates of mortiality in a widespread herbivorous insect, the holly leaf-miner *Phytomyza ilicis* Curtis (Diptera, Agromyzidae). The holly leaf-miner is the most common insect herbivore of European holly *Ilex aquifolium*, feeding inside the leaves with the large blotch mines visible on the upper surface of occupied leaves. A pest population of holly leaf-miners was successfully controlled by introducing a parasitic wasp in Canada (Clausen, 1978), highlighting the potential economic importance of understanding such species interactions. While in the leaf, the larva is subject to a number of potential mortalities, including miscellaneous death during the larval or pupal stages, e.g. starvation caused by low plant quality, parasitism (always fatal) by various species of wasp, and bird predation (Cameron, 1939).

Sampling across Sheffield indicated only rather weak effects of impervious surface cover, housing and tree cover on the abundance and mortality of the holly

**Table 4.2** Holly Leaf-Miner Demographics in relation to Urban Form in Sheffield

| Urban form variable | Prop. mined leaves | Misc. larval death | Larval parasitism | Bird predation | Pupal parasitism | Misc. pupal death | Successful emergence |
|---|---|---|---|---|---|---|---|
| Impervious surface cover | –0.1 | 0.04 | 0.20** | –0.12 | –0.06 | –0.19* | –0.12 |
| Housing density | 0.14* | –0.11 | 0.01 | –0.19** | 0.14* | –0.04 | 0.09 |
| Tree cover | –0.15* | 0.21** | –0.002 | 0.23*** | –0.20** | –0.10 | –0.22** |

Of 460 sampling locations across the city, holly plants were found within the survey area (approx 1 ha) at 276. The proportion of mined leaves on each plant was estimated by haphazardly sampling The table reports Spearman rank correlation coefficients of the relationships of these demographic outcomes with three urban form variables, measured within a 100 m radius from the holly plant location.
* indicates statistically significant at the 5% level ** statistically significant at the 1% level
*** statistically significant at 0.01% level

leaf-miner (Table 4.2). The strongest relationships ($r_S > 0.2$) were those with tree cover. Both miscellaneous larval death and bird predation increased with tree cover, while pupal parasitism declined. This translated into an overall decline in successful emergence with increasing urban tree cover. The other measures of urban form (impervious surface cover and housing density) were only weakly related to rates of mortality and parasitism, and were not associated with any overall change in successful emergence. These results show that at least some complex interactions among species can remain apparently largely intact even within highly developed sites.

## *Recreation*

For many people, green spaces in urban areas provide their primary contact with biodiversity and the “natural” environment (Jorgensen et al., 2002), may influence their physical and mental well-being (Ulrich et al., 1991; Jackson, 2003), and, in the case of public green space, can offer broader social benefits as meeting places that give a shared focus to diverse communities and neighbourhoods (Germann-Chiari and Seeland, 2004; Martin et al., 2004). In consequence, regulatory and advisory agencies have made various recommendations for the minimum provision of urban green space, usually expressed as the walking distance or time to access the resource (e.g. Stanners and Bourdeau, 1995).

Across Sheffield, there is enormous variation in the distances through the road network that separate households from their nearest accessible public green space (Barbosa et al., 2007). Many households do not enjoy the levels of access recommended by governmental agencies, with the distribution of distances being strongly right-skewed such that for some households these distances are particularly large. The mean level of access varies significantly across different sectors of society. As these sectors tend to occupy areas characterised by different urban form, levels of access are also likely to vary systematically with urban form, although this has not explicitly been tested. These distance-based measures of access could usefully be refined to include travel constraints, such as physical and psychological barriers to pedestrian movement (Handy, 1996).

Even if green space is locally accessible, its quality is extremely variable. Work in Sheffield has demonstrated that benefits to psychological well-being of visitors to urban parks are positively correlated with the species richness of those spaces, suggesting that the biological complexity of urban green space is important in enhancing human well-being, as well as for the conservation of biodiversity itself (Fuller et al., 2007a; see Chapter 10).

The private domestic garden has long been considered an important part of human health and well-being (see e.g. Gerlach-Spriggs et al., 1998). Access to a garden reduced self-reported sensitivity to stress (Stigsdotter and Grahn, 2004), while lack of access has been associated with increased self-reported levels of depression and anxiety (Macintyre et al., 2003). When asked to identify the contribution of spending time in the garden to overall well-being, 57% of householders in Perth, Australia indicated it was very important or the most important factor (ARCWIS, 2002). Interactions between people and nature frequently happen in private gardens, with wildlife-friendly gardening practices now receiving greater attention, and a large proportion of the population specifically providing food and shelter for birds (Lepczyk et al., 2004; Fuller et al., in press). Because of the diverse array of benefits to human well-being of gardens on the one hand, and public green spaces on the other, it is unclear to what extent they are substitutable in urban planning (Barbosa et al., 2007). This is especially important given that increasing housing density reduces non-garden green space and expands overall coverage by gardens, although individual gardens become smaller.

## Biodiversity and Urban Form

Biodiversity can be thought of as the variety of life, at all levels of organisation from genetic through species diversity to ecolosystem diversity. Biodiversity has variously been considered to be a contributor to some ecosystem services, to be an ecosystem service itself, and to be a product of some ecosystem services (Gaston and Spicer, 2004; Millennium Ecosystem Assessment, 2000). Doubtless, in some senses it is all three. However, in the context of urban systems, biodiversity has typically been considered in its own right, and largely with respect to how it relates to the physical level of urbanization, with little consideration of its relevance to ecosystem services or how it is influenced by variation in human activity across the urban landscape.

A large number of studies has now been conducted examining the relationships between the species richness of selected taxonomic groups (species richness is a key measure of biodiversity) and levels of urban development, usually along a rural-urban gradient. Generally, studies have shown declines in overall species richness at very high levels of urbanization, but a mixed response at low and intermediate levels of urban development, where the observed pattern depends strongly on taxonomic group (Marzluff, 2001; Chace and Walsh, 2006; McKinney, 2008). Some studies have shown no simple pattern with increased urban development (e.g. Roy et al., 1999; Niemelä et al., 2002; Mason, 2006). A decline in richness as urbanization intensifies is usually attributed to the loss of suitable habitat and resources. A peak in richness at intermediate levels is often associated with a greater number of land use types in areas of intermediate levels of development, intermediate levels of disturbance, and the multiple private ownership of land that leads to variation in management styles. Increases in richness with urban development seem usually, at least in part, to occur because of the relatively high numbers of alien/introduced species in more heavily developed areas (e.g. Kowarik, 1990; Roy et al., 1999; Marzluff, 2001; Wittig, 2004).

Across Sheffield, both plant and bird diversity shows strong heterogeneity (Fig. 4.10). Species richness of breeding birds is concentrated around the edges of the city, with generally low values in the centre (Fig. 4.10a). Breeding bird density shows a superficially similar pattern (Fig. 4.10b), although values appear to peak inside the margin of the city, an effect particularly evident along the south-eastern and northern fringes of the city. This pattern suggests that breeding bird densities peak some distance inside the edge of the city. Patterns of plant species richness are less clear. There are high values of native plant richness near the edge of city, particularly in the south and east (Fig. 4.10c), although there is substantial heterogeneity in the distribution. Alien plants show very low species richness around the edge of the city, peaking at intermediate levels of urban development, and declining again toward the city centre (Fig. 4.10d).

Despite the large number of studies documenting the responses of biodiversity to urbanization, understanding of the relationship between biodiversity and urban form per se remains poor. The different components of the rural-urban gradient are often not well differentiated in analyses, and comparison tends to be focussed on the difference between rural and urban areas rather than between urban areas of differing structure. This is particularly significant at the present time, when on the one hand urban areas in the UK are becoming havens for some species that have undergone marked declines in the wider countryside (e.g. blackbird *Turdus merula*, song thrush *T. philomelos*; Gregory and Baillie, 1998; Mason, 2000), largely as a consequence of intensive agriculture, and on the other hand some previously common and widespread species are undergoing marked declines in urban areas (e.g. starling *Sturnus vulgaris*, house sparrow *Passer domesticus*; Cannon et al., 2005). There are strong suggestions that both of these trends are more apparent in some urban forms than in others, but empirical evidence remains scant.

In Sheffield, housing density was strongly negatively associated with breeding bird richness (Fig. 4.11a), while the relationship with breeding bird abundance appeared hump-shaped, with bird abundance peaking at intermediate levels of housing density (Fig. 4.11b). This is presumably related to the overall loss of green space with increasing urban development, as well as a reduction in the range

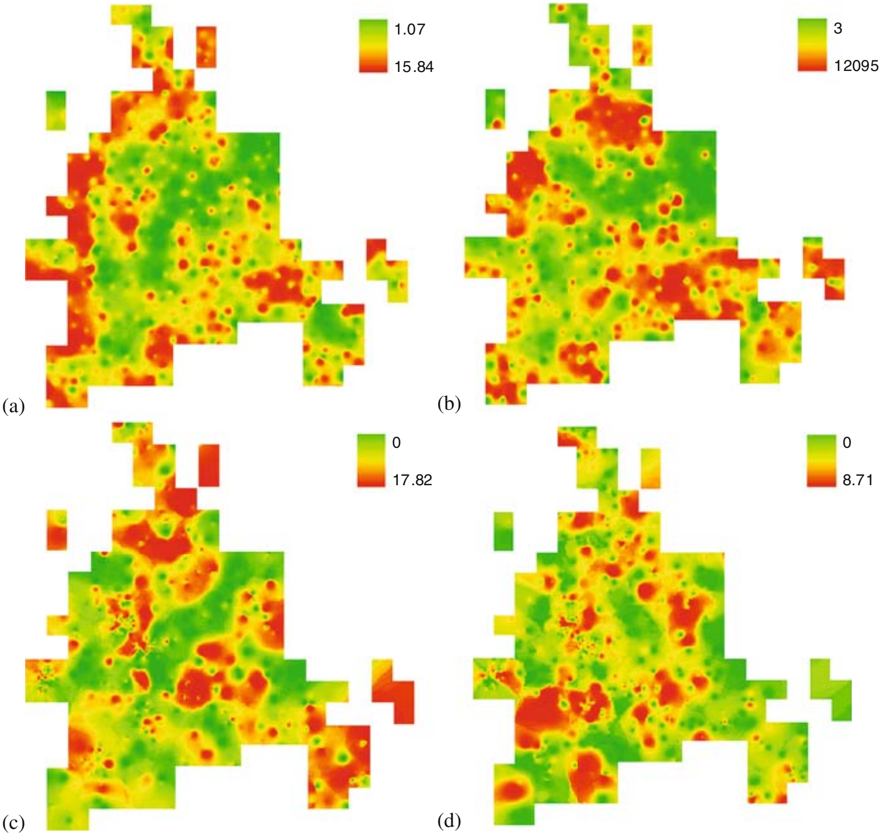


**Fig. 4.10** Surfaces for Sheffield of (**a**) Breeding Bird Species Richness, (**b**) Breeding Bird Density (individuals km$^{-2}$), (**c**) Native Plant Species Richness and (**d**) Non-native Plant Species richness. Bird data are derived from 5-minute point transects carried out at 640 locations across Sheffield (a point randomly located within each 500 × 500 m grid cell across the city). Plant data are derived from 1000 quadrats (1 m$^2$) placed across the city

of available habitat types. A range of processes may contribute to such changes in abundance, including natural food availability, the availability of artificially-provided supplementary food, nest site availability and quality, predation pressure and interspecific competition (Clergeau et al., 1998; Thorington and Bowman, 2003; Shochat, 2004; Faeth et al., 2005). The ratios of different types of habitat available are likely to change systematically with increasing urban development. For example, as its overall coverage declines, green space changes markedly in composition, with a greater proportion being made up of gardens rather than other forms of vegetated surface.

The diversity of plants was more weakly related to housing density (as measured by the number of addresses per unit area). Raw plots show a negative relationship between housing density and native plant species richness (Fig. 4.11c), and a

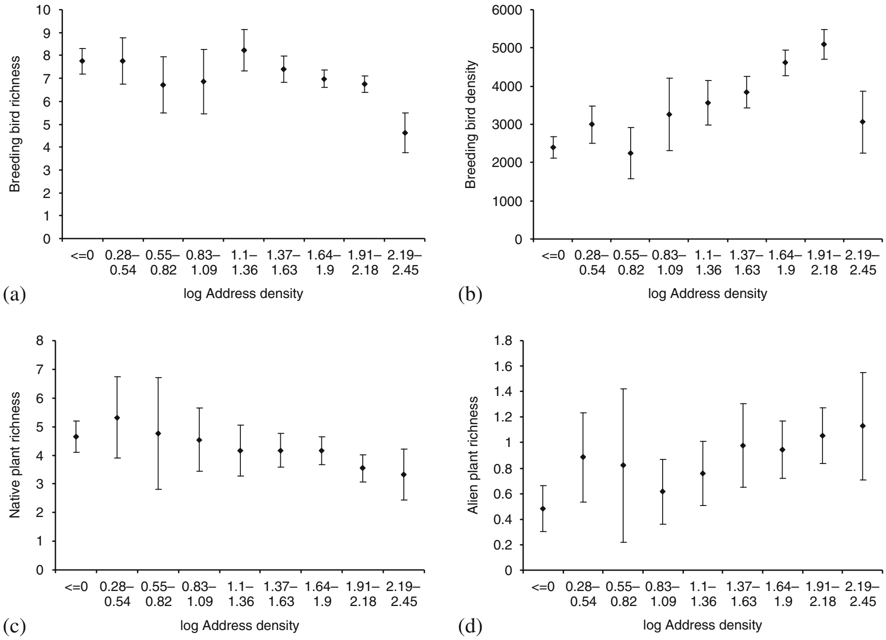


**Fig. 4.11** Relationships between Housing Density and (**a**) Breeding Bird Species Richness, (**b**) Breeding Bird Density, (**c**) Native Plant Species Richness and (**d**) Non-Native Plant Species Richness. Housing density is log address density within a 100 m buffer around each sampling location. Bird density is individuals km$^{-2.}$ Error bars are 95% confidence intervals

positive relationship between housing density and alien plant species richness (Fig. 4.11d). Many urban alien plants in northern Europe have natural ranges in regions with Mediterranean climates and dry soils, rather akin to those found within highly urbanized environments (Sukopp and Wurzel, 2002). Moreover, propagule pressure resulting from introductions and escapes of alien plants from gardens and amenity plantings is likely to be much higher in towns and cities than in rural areas, and disturbed urban habitats may promote the establishment of non-native weedy species (Smith et al., 2006c; Dehnen-Schmutz et al., 2007).

Work in Sheffield has also revealed an intriguing behavioural response to urban noise (arguably either a measure, or a close correlate, of urban form). Sites where European robins *Erithacus rubecula* sang nocturnally tended also to be those places with high noise levels during the day, suggesting that the birds were singing at night to avoid acoustic competition with daytime urban noise (Fuller et al., 2007b). As noise levels are closely related to certain components of urban form, such as where transport networks are located, the times at which birds choose to sing may in turn be affected by urban design. (Warren et al., 2006). Daytime noise at a particular urban location is strongly positively correlated with the proportion of impervious surface in the surrounding area (Fig. 4.12). Integrative studies that can disentangle these multiple ecological and behavioural effects at a city-wide scale are needed.

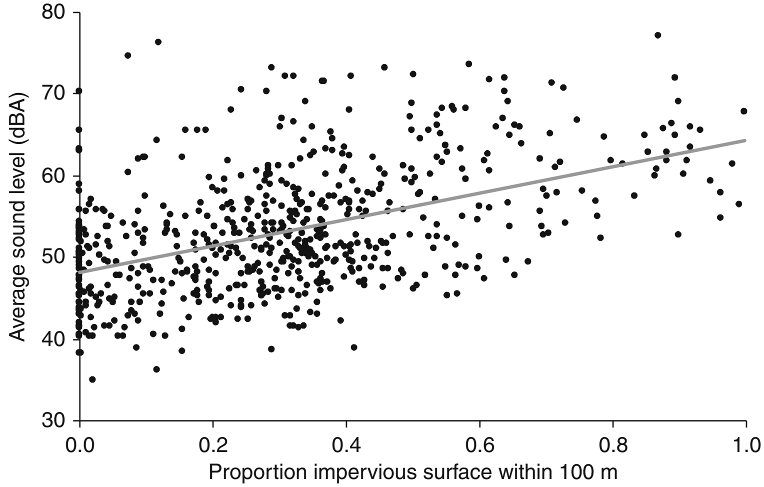


**Fig. 4.12** Relationship between Noise Levels and Impervious Surfaces. Noise levels are the mean of 10 measurements taken with a handheld digital sound meter at 30 s intervals at 628 locations across Sheffield and the proportion of impervious surface within a 100 m radius around these points ($r = 0.48$, $n = 628$, $p < 0.001$)

Looking more widely across Britain as a whole, the species richness and abundance of breeding birds responds systematically to variation in housing density (Tratalos et al., 2007b). At a resolution of $1 \times 1$ km squares, avian species richness at first increases strongly with housing density, but then declines rapidly at higher housing densities. A similar pattern is seen in the richness of 27 urban specialist species used as urban health indicators by the UK government (DEFRA, 2002, 2003), suggesting that even those species best able to exploit urban environments are impacted at high urban densities. Standardised abundances of all species, including the urban indicator species, are positively associated with housing density, but decline at very high urban densities (Tratalos et al., 2007b).

## Conclusions

The analysis has shown that the density of urban development, a key measure of urban form, is strongly associated with a reduction in total green space coverage, and changes to the connectivity of vegetated patches within the urban landscape. This has ramifications for the ecosystem services that are mediated by green space, including the regulation of water and temperature regimes, carbon sequestration and the provision of pest control and pollinators across the urban landscape. Moreover, changes to the amount and quality of green space will have significant consequences for recreation within urban areas, access to an experience of nature, and for human quality of life more generally. Levels of biodiversity generally increased initially with housing density, but then declined sharply at highly developed sites.

An understanding of the distribution of ecosystem services and the associated pattern of biodiversity across the urban landscape is crucial for predicting the consequences of the increases in urban density required by current UK legislation. Although many of these responses point to a decline in ecosystem function and biodiversity potential with increasing urban density, there is substantial scatter around many of these relationships, suggesting scope for maximizing the environmental and ecological performance of urban areas for any given level of urban density. Perhaps rather more troubling are the observed declines in abundance at high urban densities even of those species most able to exploit urban environments. This suggests a difficult trade-off between on the one hand, minimizing the conversion of land for new development, and on the other maintaining meaningful levels of biodiversity and ecosystem function around the places where most of us live. There is a need for longitudinal studies documenting past changes in the spatial configuration of urban landscapes in response to urbanization, and using that information to predict the future consequences of alternative modes of development and regeneration at regional and even national scales. Spatially explicit area selection exercises, of the kind that have been used extensively in the conservation planning literature (e.g. Pressey et al., 1997; Cabeza and Moilanen, 2001) will help identify areas that are crucial to maintaining effective ecosystem function, and those that might efficiently be used for high density residential developments.

With more than half the world's population now living in urban areas, changes in how we plan, manage and develop such areas has potentially profound impacts. A better understanding of the best strategies for managing the trade-offs among environmental functions and urban form is urgently required if we are to ensure that increasing urban density and concomitant declines in green space and biodiversity are not to lead to impaired ecosystem function, reduced provision of ecosystem services and the degradation of human experiences of nature.

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