Urban form, biodiversity potential and ecosystem services

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Received 24 October 2006; received in revised form 24 May 2007; accepted 26 May 2007
Available online 5 July 2007

Abstract

Using data from selected areas in five UK cities, we studied the relationships between urban form and the following measures of ecosystem performance: availability and patch characteristics of tree cover, gardens and green space; storm-water run-off; maximum temperature; carbon sequestration. Although most measures of ecosystem performance declined with increasing urban density, there was considerable variability in the relationships. This suggests that at any given density, there is substantial scope for maximising ecological performance. The social status of residents was related to measures of tree cover. Housing type was significantly associated with some types of ecosystem service provision, indicating that the type of development may be important independent of its density. These findings have implications for understanding the distribution of ecosystem services and biodiversity across urban landscapes, and the management of development aimed at meeting UK government housing density targets. © 2007 Elsevier B.V. All rights reserved.

Keywords: Carbon sequestration; Green space; Housing density; Patch size; Run-off; Tree cover

1. Introduction

More than half the world’s population now lives in cities, compared with about 14% a century ago (United Nations, 2001). This increasing urbanisation radically modifies the ecology of landscapes. The effects include alteration of habitat, such as loss and fragmentation of natural vegetation, and the creation of novel habitat types (Davis, 1978; Niemelä, 1999a,b; Wood and Pullin, 2000); the alteration of 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; Donovan et al., 2005; Bonan, 2000); the alteration of disturbance regimes, with many habitats experiencing more frequent disruption (Rebele, 1994); the alteration of species composition, species diversity, and proportions of aliens (Davis, 1978; Ruszczyk and de Araujo, 1992; Rebele, 1994; Roy et al., 1999; Hardy and Dennis, 1999; McKinney, 2002).

One approach to reduce the impact of increasing urbanisation is to minimise the spatial extent of urban areas by developing more compact city forms. There has been much recent debate over the “compact city” paradigm, with its aims of centralising services and reducing urban land take (Jenks et al., 1996; Williams et al., 2000; Jenks and Dempsey, 2005). Such developments reduce urban sprawl, and significant long-term social and ecological benefits have been claimed (Burton, 2000; Jenks and Burgess, 2000). However, whilst the focus has been on the benefits of reducing urban area, we know much less about how urban densification changes the ecosystem characteristics of the urban areas themselves. For example, is it possible to build dense, compact cities that maintain areas of natural habitat and provide useful levels of ecosystem services, such as carbon sequestration and storm-water interception? The net ecological effect of moving towards higher-density urban forms clearly depends on the balance of the benefits of reduced land take against the changes in ecosystem function of the higher density urban areas.

In the UK, current policy is to build new developments at high net density, partly as a response to increasing urban populations and social and demographic pressures resulting in a reduction in average household size (ODPM, 2002). One possible ramification of increased urban density might be deterioration in ecosystem service provision in urban areas and declines in both urban biodiversity and the quality of life of the urban human population. As ecosystem services such as carbon sequestration, storm-water interception, climate regulation and biodiversity...
potential are influenced by the availability and type of vegetated ground cover, increased densification, if it brings with it a reduction in the proportion of such cover and changes in its spatial configuration, may have undesirable effects on these services (Arnold and Gibbons, 1996; McPherson, 1998; Simpson, 1998; Xiao et al., 1998; Weng, 2001; Whitford et al., 2001). This would be a particular concern for at least three reasons. First, many of the ecosystem services provided by urban green spaces carry with them significant economic implications, both locally and regionally (McPherson, 1992; Chee, 2004; Farber et al., 2006). These include implications for house prices, the costs of lighting, cooling and heating of buildings, and the ease of attracting businesses and employees (e.g. Luttik, 2000; Tyrväinen and Miettinen, 2000; Morancho, 2003; CABE Space, 2004). Second, in the face of intensive agriculture in the wider landscape, in some regions urban green spaces now act as important havens for native plant and animal populations (Mörberg and Wallentinus, 2000; Gregory and Baillie, 1998; Mason, 2000; Gaston et al., 2005). Third, there is growing evidence that local green spaces contribute to both the physical and mental well-being of people living in urban areas (Hartig et al., 1991; Chiesura, 2004; Takano et al., 2002; de Vries et al., 2003), and that the pattern of provision of such spaces is an important issue for social equity (Whitford et al., 2001; Pauleit et al., 2005).

In this paper we investigate how urban form affects the ecological performance of the urban environment through an evaluation of the relationships between urban form and measures of environmental quality and biodiversity potential, over 15 sites distributed across five UK cities. This work forms part of a much broader consortium project to assess multiple dimensions of the sustainability of a variety of urban forms using these study areas (Jones, 2002; http://www.city-form.com).

2. Methods

2.1. Data

In each of five UK cities, Edinburgh, Glasgow, Leicester, Oxford and Sheffield, three study sites were selected, each containing ca. 2000 households (Fig. 1). Sites were selected on the basis that each city should contain a city centre site (Inner), an outer suburban site (Outer) and a site situated between the centre and suburbs (Middle). This is similar to the urban gradient approach widely used in urban ecology (see, for example, McDonnell and Pickett, 1990; Hahs and McDonnell, 2006). It was done on the assumption that it would allow the examination of a variety of urban forms, rather than that all measures of urbanisation (such as housing density) would necessarily decrease

Fig. 1. The 15 study sites, shown at the same scale. ©Crown copyright Ordnance Survey. All rights reserved.
with distance from the centre. Each site was delineated along the boundaries of output areas from the UK 2001 census (see Boyle and Dorling, 2004), to allow matching to census data. Output areas are irregular polygons mapped around the addresses of households included in the census, and together cover the entire UK mainland. Mean area of the study sites was 1.33 km² (range: 0.43–4.86 km²), and each of the study sites comprised between 13 and 53 output areas from the 2001 UK census.

The land cover characteristics of each output area within the study sites were determined in a GIS, based on the classification of surface cover polygons by Ordnance Survey within the MasterMap digital cartographic dataset at a 1:1250 scale (Murray and Shiell, 2003). The MasterMap classifications were grouped into seven categories—woodland, scattered trees, scrub, garden, other vegetated areas, sealed surfaces and water bodies. For a small number of cases (ca. 2%), Ordnance Survey had described a polygon as either “Unknown” or “Unclassified”. Aerial photographs at 25 cm resolution, produced by Cities Revealed (http://www.citiesrevealed.com), were used to classify these polygons by eye. Tree cover was mapped in a GIS for each of the study sites, by manually tracing around each tree or group of trees shown in the aerial photographs.

Seven measures of urban form were used. Four of these came directly from the UK national census, a survey of all UK households, which was last conducted in 2001 (Rees et al., 2002). (i) Population density: the number of residents per hectare; (ii) housing density: the number of households per hectare; (iii) proportion detached/semi-detached: proportion of houses that were detached or semi-detached. Detached properties are where the house does not share a wall with a house belonging to an adjacent property (although the land belonging to it will in most cases share a boundary with the land belonging to other houses). In semi-detached properties, the house shares a wall with a house on one side of the property but not on the other; (iv) proportion in social group AB: proportion of residents classified in social group AB, comprising the more affluent and professionally qualified sectors of society. This last variable was included in an attempt to separate the effects of urban built form from potentially confounding socio-economic factors.

Because the UK census provides data only for residential properties, three further measures of urban form were derived from MasterMap building polygons and Ordnance Survey Addresspoint data: (v) address density: the number of addresses per hectare; (vi) building density: the number of buildings per hectare; (vii) density of buildings with addresses: the number of buildings with one or more associated addresses per hectare.

Biodiversity potential was evaluated using four measures of habitat cover and four measures of habitat patch size. The habitat cover measures were the proportion cover of (private/domestic) gardens; proportion cover of green space; proportion cover of gardens and green space; proportion tree cover over gardens or green space. Green space is here defined as all areas classified as natural but not as garden in the MasterMap dataset. Gardens were treated separately because they are a distinctive and extensive feature of urban environments (Gaston et al., 2005). The patch size measures were the average patch size of: green space, gardens and green space, non-sealed areas (green space and gardens), tree cover, and tree cover over gardens and green space. Areas of cover <10 m apart were grouped as part of the same patch for the purposes of the patch size analyses, as small areas of sealed surface, such as paths through parks, or narrow roads, seem unlikely to act as effective barriers to dispersal for most plant and animal species, although the degree to which this is true will clearly vary with the biota under consideration.

A technique adapted from that used by Whitford et al. (2001) and Weng (2001) was used to estimate run-off for each of the sites. This technique has its basis in studies conducted by the Soil Conservation Service (1972) and further developed by Pandit and Gopalakrishan (1996). It calculates surface run-off as

\[ P_r = (P - 0.25S^2)/(P - 0.8S) \]

where \( P_r \) denotes surface run-off, \( P \) precipitation and \( S \) the maximum potential rainfall retention of the catchment. \( S \) is calculated as 2540/CN – 25.4, where CN is the curve number calculated by the Soil Conservation Service for each combination of land cover and soil type. Although soil type is known to affect hydrological characteristics, urban soils are spatially complex and poorly mapped (Effland and Pouyat, 1997), and as our study aimed to isolate the effect of urban form from other predictors of run-off, the same soil type (sandy loam) was assumed for all study sites. This was selected as it is a common soil type and its free draining nature means that land cover has a large influence on surface run-off. Using the Soil Conservation Service classifications as a basis, soil curve numbers were assigned to the various cover types as follows: sealed surfaces (including rock), 98; woodland and other tree cover, 55; scrub, 66; rough grassland, 58. Natural vegetation which was not classified as woodland, scrub or rough grassland, and which lacked tree cover, was assigned a curve number of 61, typical of both turf grass and flowerbeds. Areas designated as “scattered trees” were assigned a curve number of 55 in areas mapped as tree cover from aerial photos and 61 in areas where no tree cover was present. Gardens were assigned a curve number of 74.5, on the basis of the percentage cover estimated from a sample of 70 randomly selected gardens located in the three Sheffield study sites (ca. 33% sealed, 37% grass, 12% scrub, 10% herbaceous, 8% earth or gravel). Areas classified as “rock with scrub” (<0.01% of the surface cover of the study areas) were assigned the same curve number as gardens, as they comprised a mixture of sealed and unsealed surfaces. Following Whitford et al. (2001), average run-off from a 12 mm storm event, a common occurrence in UK cities, was calculated for the area of each study site. Areas of water bodies were not included (water bodies made up less than 0.7% of the total area studied, with a maximum of 3.9% in the Oxford Middle site).

Calculations for carbon sequestration were based on percentage tree cover digitised from the aerial photographs. The formula of Rowntree and Nowak (1991) was used: tonnes of carbon sequestered acre⁻¹ year⁻¹ = 0.00335(% tree cover). As the relationship between carbon sequestration and tree cover is therefore linear, results for carbon sequestration can therefore be interpreted as matching those for tree cover, and therefore reflecting additional ecosystem services associated with tree cover such as temperature and noise buffering, air quality improvement and aesthetic value.
The model of Whitford et al. (2001), developed from Tso (1991) was used to calculate maximum temperature, using as inputs the evaporative fraction (the proportion of each study site consisting of vegetation and water bodies, based on the MasterMap data and digitised tree cover) and the built mass, using the assumption of Tso (1991) of 777 kg m$^{-2}$ of the built environment. Cover by sealed surface was calculated as the proportion of land area classified by MasterMap as man-made + 0.33 x proportion of the land area covered by gardens (0.33 being the proportion of the sample of Sheffield gardens which was found to be sealed surface). The same climatic inputs used by Whitford et al. (2001), typical of a warm summer day in the UK, were used throughout.

The run-off and temperature models used in this study assume that all gardens have the same proportion of sealed surfaces, lawn, shrubs and beds, whereas these proportions are likely to vary to some degree with garden size and the sociological characteristics of the study area. To get some idea of the degree of error which might be introduced by this simplification, for two of the study areas, the Outer and the Inner sites in Sheffield, aerial photographs were used to classify each of the 12,000 MasterMap polygons by eye into percentage tree, shrub, herbaceous plants, turf, rough grass and sealed surfaces, and the surface run-off and maximum temperature were calculated using these data. Run-off values differed only slightly, by 3.4% for the inner area (aerial photo method = 5.914, map classification method = 6.119) and 8.1% for the outer areas (aerial photo method = 8.409, map classification method = 7.775). Maximum temperature values differed by 0.6% for the inner (aerial photo method = 26.32, map classification method = 26.48) and 9.8% for the outer locations (aerial photo method = 21.4, map classification method = 23.72).

2.2. Analyses

For each of the 15 study sites, Spearman rank correlations were calculated between each of the urban form variables and each of the environmental indicator variables. Separate correlations were calculated for the 10 primarily residential Middle and Outer study sites, as in these areas one would expect a closer relationship between densification of built form and measures of population and housing density derived from the UK census, which contains information only from residential properties.

For the primarily residential Middle and Outer sites, each of the component output areas was used as a data point, and backwards stepwise OLS regressions were calculated to assess the relationship between the environmental variables and two of the urban form variables—household density and proportion detached/semi-detached. The patch characteristics variables were not included in these analyses, as the difference in size between the largest and smallest output areas (394 m$^2$ < 3,118,762 m$^2$) would have had a large effect on the results. The following transformations were used for the dependent environmental variables, and their respective autoregressive terms, in order to achieve approximately normal distributions: carbon sequestration, square root; proportion green space, arcsine square root; proportion tree cover over gardens or green space, arcsine square root; proportion garden, arcsine. In each of these models, a spatial autoregressive term, consisting of the mean value for all neighbouring output areas, was incorporated into the model, to take account of spatial autocorrelation in the data. Standard spatial models, which take account of the distance decay of the autocorrelation process, were not calculated, as the output areas were so irregular in shape and variable in size that an accurate assessment of the autocorrelation structure was not feasible.

3. Results

3.1. Comparison of ecosystem performance among 15 study sites

The Inner, Middle and Outer site categorisations were intended to reflect variations in urban form within cities rather than between them, and to provide an overall continuum in urban form across all the cities, and thus formal comparisons between these groups were not conducted (almost all variables did indeed exhibit continuous variation across the 15 study sites; see below). Thus, whilst cover of green space and gardens generally increased from Inner to Outer sites, the proportion of this area made up of green space as opposed to garden differed from site to site (Fig. 2a). The Outer area of Sheffield showed a particularly high proportion of garden, and the Outer area of Glasgow a particularly high proportion of green space. The proportion of the study areas with tree cover over green space or gardens followed a similar pattern to tree cover over all surfaces. The study sites in Edinburgh showed particularly large variability in this measure, with the Inner site having the lowest value from the 15 study areas (2.9%) and its Outer site the highest (27.7%).

Habitat patch sizes differed markedly among sites. Mean patch size of non-sealed surfaces was greatest in the Outer areas, possibly because of their tendency to have large gardens and narrow roads (Fig. 2b). This did not apply universally to green space patch size, which was often quite large in the Middle and Inner areas, partly due to the effect of urban parks (Fig. 2b). Tree patch size, and the patch size of trees over green space or gardens were also often lower in Outer than Middle areas (Fig. 2c), and tree cover over green space or gardens followed a similar pattern (Fig. 2c). Inner sites were generally low in tree cover, and hence carbon sequestration, but in three out of five cases Middle sites showed a greater predicted rate of carbon sequestration than Outer sites (Fig. 2d).

Predicted run-off was in all cases higher in the Inner site of the five cities than in the Middle or Outer sites. In all cases except Oxford, Middle sites all had higher run-off than Outer sites (Fig. 2e). Maximum temperature followed a similar pattern to run-off, partly because the proportion of non-sealed surfaces was a major component in both models (Fig. 2f). Population density and housing density showed a decline from Inner to Outer sites (Fig. 2g). Proportion detached/semi-detached increased from Inner to Outer sites, although the pattern for proportion in social group AB was less marked (Fig. 2h).

Among the measures of urban form there were particularly strong correlations between population density and household
density ($r_s = 0.971$), and between the density of buildings with addresses and building density ($r_s = 0.954$), and so the first of each pair of these variables was dropped from further analyses. Likewise, among the measures of patch size, the patch size of tree cover and of tree cover over gardens and green space were particularly strongly correlated ($r_s = 0.975$) and the former was dropped from further analyses.

### 3.2. Correlates of biodiversity potential

When the analysis was conducted over all 15 study sites, in almost all cases the three urban density variables were significantly negatively correlated with the biodiversity potential/habitat variables, and of these address density was the most strongly related (Table 1). There was one exception to this, proportion cover by gardens, which was more closely correlated with the density of buildings (Table 1). The proportion of surface cover by gardens was also positively correlated with the proportion of detached/semi-detached housing, and when green space was added to this cover variable this relationship became yet more marked (Table 1). The proportion of detached/semi-detached housing had a significant relationship with three of the patch size variables, patches being larger in areas where the proportional coverage of these housing types was greater (Table 1). The proportion of the population in social groups AB was significantly correlated with only one of the biodiversity potential variables: proportion tree cover over gardens or green space. In the main, all of these results held up when constrained to just the Middle and Outer study sites, with some being strengthened and others weakened (Table 1).

### 3.3. Correlates of ecosystem performance

For the 15 study areas, address density was significantly positively correlated with both run-off and maximum temperature, and negatively correlated with carbon sequestration (Table 1).
Conversely, the proportion of detached/semi-detached housing was significantly negatively correlated with run-off and temperature and positively correlated with carbon sequestration. The latter also increased significantly with the proportion of detached/semi-detached housing. Proportion detached/semi-detached housing was selected and positively related with garden cover, and negatively related with green space cover and carbon sequestration, although it did not have a strong effect on the fit of these models.

4. Discussion

In this study across five cities in the UK, we have shown that high-density urban developments were generally associated with poor environmental performance, as measured by green space patch size and the levels of provision of key environmental services. More densely urbanised areas had less coverage by green space and gardens, smaller habitat patch sizes, greater predicted run-off, higher predicted maximum temperatures and lower predicted carbon sequestration (and hence tree cover) (Table 1, Figs. 3 and 4). In residential areas, a very close relationship was found between urban densification and a variety of the environmental indicators. Some of these relationships were clearly non-linear, with coverage by green space and its patch size particularly tending to decline most rapidly at lower levels of urbanisation (Fig. 3). Our results indicate, however, that ecosystem quality tends to decline continuously as urban density increases, although the scatter evident in many of these relationships suggests that for any given urban density, and with appropriate consideration to the proportion and configuration

### Table 1

Spearman rank correlations between measures of urban form and measures of biodiversity potential and ecosystem performance

<table>
<thead>
<tr>
<th>Urban form</th>
<th>Address density</th>
<th>Building density</th>
<th>Household density</th>
<th>Proportion detached/semi-detached</th>
<th>Proportion social group AB</th>
</tr>
</thead>
<tbody>
<tr>
<td>Biodiversity potential</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Proportion cover of gardens</td>
<td>−0.186</td>
<td>0.443</td>
<td>0.186</td>
<td>0.671***</td>
<td>0.446</td>
</tr>
<tr>
<td></td>
<td>0.091</td>
<td>0.248</td>
<td>0.091</td>
<td>0.261</td>
<td>0.491</td>
</tr>
<tr>
<td>Proportion cover of green space</td>
<td>−0.639*</td>
<td>−0.329</td>
<td>−0.514*</td>
<td>0.357</td>
<td>−0.279</td>
</tr>
<tr>
<td></td>
<td>−0.661*</td>
<td>−0.636*</td>
<td>−0.661*</td>
<td>0.261</td>
<td>−0.406</td>
</tr>
<tr>
<td>Proportion cover of green space and garden</td>
<td>−0.696***</td>
<td>0.025</td>
<td>−0.329</td>
<td>0.918***</td>
<td>0.121</td>
</tr>
<tr>
<td></td>
<td>−0.830***</td>
<td>−0.552</td>
<td>−0.830***</td>
<td>0.758*</td>
<td>−0.164</td>
</tr>
<tr>
<td>Proportion tree cover over gardens or green space</td>
<td>−0.529*</td>
<td>0.104</td>
<td>−0.282</td>
<td>0.546*</td>
<td>0.586*</td>
</tr>
<tr>
<td></td>
<td>−0.358</td>
<td>−0.382</td>
<td>−0.358</td>
<td>0.079</td>
<td>0.539</td>
</tr>
<tr>
<td>Average patch size of green space</td>
<td>−0.646*</td>
<td>−0.114</td>
<td>−0.357</td>
<td>0.800***</td>
<td>0.096</td>
</tr>
<tr>
<td></td>
<td>−0.721*</td>
<td>−0.624</td>
<td>−0.721*</td>
<td>0.576</td>
<td>−0.127</td>
</tr>
<tr>
<td>Average patch size of green space and garden</td>
<td>−0.682**</td>
<td>−0.064</td>
<td>−0.586*</td>
<td>0.586*</td>
<td>−0.061</td>
</tr>
<tr>
<td></td>
<td>−0.576</td>
<td>−0.370</td>
<td>−0.576</td>
<td>0.430</td>
<td>−0.079</td>
</tr>
<tr>
<td>Average patch size of non-sealed areas</td>
<td>−0.660*</td>
<td>0.146</td>
<td>−0.296</td>
<td>0.950***</td>
<td>0.150</td>
</tr>
<tr>
<td></td>
<td>−0.745*</td>
<td>0.309</td>
<td>−0.745*</td>
<td>0.879**</td>
<td>−0.091</td>
</tr>
<tr>
<td>Average patch size of tree cover over gardens and green space</td>
<td>−0.567*</td>
<td>−0.257</td>
<td>−0.473</td>
<td>0.357</td>
<td>0.389</td>
</tr>
<tr>
<td></td>
<td>−0.576</td>
<td>−0.612</td>
<td>−0.576</td>
<td>0.139</td>
<td>0.321</td>
</tr>
<tr>
<td>Ecosystem performance</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Run-off</td>
<td>0.754***</td>
<td>0.061</td>
<td>0.414</td>
<td>−0.861***</td>
<td>−0.186</td>
</tr>
<tr>
<td></td>
<td>0.891***</td>
<td>0.685*</td>
<td>0.891**</td>
<td>−0.612</td>
<td>0.018</td>
</tr>
<tr>
<td>Carbon sequestration</td>
<td>−0.550</td>
<td>0.064</td>
<td>−0.296</td>
<td>0.593</td>
<td>0.514*</td>
</tr>
<tr>
<td></td>
<td>−0.358</td>
<td>−0.382</td>
<td>−0.358</td>
<td>0.079</td>
<td>0.539</td>
</tr>
<tr>
<td>Temperature</td>
<td>0.750***</td>
<td>0.071</td>
<td>0.396</td>
<td>−0.854***</td>
<td>−0.200</td>
</tr>
<tr>
<td></td>
<td>0.903***</td>
<td>0.709*</td>
<td>0.903***</td>
<td>−0.576</td>
<td>−0.079</td>
</tr>
</tbody>
</table>

In each case the upper figure is for the relationship across all study areas (n = 15) and the lower figure across just the Middle and Outer sites (n = 10). Asterisks after correlation coefficients indicate the significance level (*p < 0.05; **p < 0.01; ***p < 0.001).
of green space and tree cover, there is substantial scope for maximising ecological performance.

At the level of output areas, housing type (proportion of houses that were detached/semi-detached) had significant relationships independent of housing density with carbon sequestration/tree cover, cover by green space and cover by gardens (Table 2). This indicates that relationships between urban form and ecological performance do not simply arise as a result of housing density. Indeed, housing type was the only significant predictor of garden cover apart from the spatial autoregressive term (Table 2), indicating a greater saturation with gardens in areas of predominantly detached or semi-detached housing and a relatively small effect of individual garden size. Gardens are typically heterogeneous, with wide variation among gardens in relative cover by impervious surface, vegetation characteristics, presence of features specifically intended to promote biodiversity (Gaston et al., 2005; Smith et al., 2005). Individual garden size is a strong predictor of land cover composition, with smaller gardens supporting fewer land cover types and less likely to contain trees (Smith et al., 2005). Domestic gardens covered more than 20% of the urban areas of the five UK cities studied by Loram et al. (2007), indicating substantial potential for them to contribute to city-wide biodiversity potential and ecosystem performance.

Despite the general relationships, there was a large degree of variability in the environmental quality of urban areas with apparently similar urban form characteristics (Fig. 2). Identifying drivers of this variation may be a key to designing cities that optimise environmental and ecological benefits for given housing densities, where modification of the latter is not possible. Examples of this might include the planting of trees on city centre roads and the maintenance of grass verges and other vegetated areas in city centres, community initiatives to increase the biodiversity potential of gardens, reduced use of pesticide in key areas of habitat, and increased management of key open spaces for wildlife.

In interpreting these results, it should be borne in mind that although they provide a broad indication of how environmental quality changes with urban form, the specific values calculated for the various indicators are likely to match only approximately those that would be found by studies incorporating a more detailed knowledge of the study environment and that of the broader regional environment. For example, the carbon sequestration model of Rowntree and Nowak (1991) assumes a uniform relationship between tree cover and net carbon uptake. However, the relationship between tree cover and carbon sequestration will depend, among other things, on the demographic structure and species composition of the urban forest (Nowak and Crane, 2002; Nowak et al., 2002). 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 the tree crown and water potential (Quigley, 2004; Donovan et al., 2005). Another example is that the temperature model of Whitford et al. (2001) does not take into account mixing of air from areas adjacent to the study areas. Thus, although the model may be a good indicator of temperature differences due solely to urban form, it is likely to overestimate maximum temperatures in highly urbanised locations and underestimate them in urban areas with comparatively little surface sealing.

Our data highlight challenges in measuring urban density, and indicate that several measures are required for a good...
understanding of how urban densification is related to ecological performance. For example, the use of household density as a single measure of urban densification will underestimate urban density in city centres because of the relatively small proportion of buildings that are residential. This effect is presumably responsible for the fact that many of the relationships with household density evident across the residential areas, were not significant when *Inner* sites were included in the analyses. Where analyses included non-residential areas, the number of addresses per hectare, which is also a measure of densification but includes non-residential buildings, showed a stronger relationship with the environmental variables.

Our results indicate that areas with a high proportion of residents in social group AB showed greater ecological performance as measured by carbon sequestration and tree cover over green...
space or gardens, and previous work in Liverpool (Whitford et al., 2001; Paullet et al., 2005), suggested that there is a tendency for social inequity to be matched with inequity in the environmental quality of the urban landscape. However, many of the our environmental variables showed only weak correlations with the proportion of residents in social group AB, suggesting that caution is needed before accepting this as a general rule. Kinzig et al. (2005) distinguish bottom-up factors such as management of private spaces by residents and top-down factors such as landscaping policies at local authority level. They suggest that bottom-up factors are particularly affected by how residents manage their private land and therefore their economic constraints and cultural ideals about how, for example, gardens should be managed. Tree cover is the variable in our study most open to manipulation by private management (a bottom-up factor), whereas the other variables depend more on urban design (top-down factors). It is therefore interesting that in our study, socio-economic status was significantly correlated with the proportion of tree cover, and hence carbon sequestration, supporting the distinction made by Kinzig et al. (2005).

For this study, we chose seven measures of the ecosystem services provided by urban landscapes. We chose the seven measures because they are important services, broadly indicative of environmental quality, and because they could be measured across five cities using the data available to us. However, there are many more ways in which urban landscapes are likely to differ in the level of ecosystem services they provide, for example, provision of habitat or nest sites for species of conservation importance, access to flowering plants for pollinators, provision of wetland habitats, pollution abatement, nitrogen fixation. We therefore believe that further work should be encouraged to analyse the relationships between a wider range of ecosystem services and urban form.

The results here suggest that there are marked potential impacts on ecological and environmental performance of urban areas associated with very high-density urban development. The implications of current urban development strategies, such as compact cities, for environmental quality should be seen in the light of these results. Given that development at increasing density is currently a central plank of UK government housing policy (ODPM, 2002), our results point to ways in which the impacts on environmental services and biodiversity potential can be assessed and minimised.

Acknowledgements

This work was supported by EPSRC grant GR/S20529/1 to the CityForm consortium. Ordnance Survey kindly provided MasterMap data under license to CityForm. We are grateful to I. Fishburn and C. Gascoigne for assistance, and to two anonymous referees for helpful suggestions to improve the paper.

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