Environmental Pollution 183 (2013) 133–142
Contents lists available at SciVerse ScienceDirect
Environmental Pollution
journal homepage: www.elsevier.com/locate/envpol

Identifying potential sources of variability between vegetation carbon storage estimates for urban areas

Zoe G. Davies [a,*], Martin Dallimer [b], Jill L. Edmondson [c], Jonathan R. Leake [c], Kevin J. Gaston [d]

[a] Durrell Institute of Conservation and Ecology (DICE), School of Anthropology and Conservation, University of Kent, Canterbury, Kent CT2 7NR, UK
[b] Department of Food and Resource Economics, Center for Macroecology, Evolution and Climate (CMEC), Faculty of Science, University of Copenhagen, 1958 Copenhagen, Denmark
[c] Department of Animal and Plant Sciences, University of Sheffield, Sheffield S10 2TN, UK
[d] Environment and Sustainability Institute, University of Exeter, Penryn, Cornwall TR10 9EZ, UK

ARTICLE INFO

Article history:
Received 10 October 2012
Received in revised form
14 June 2013
Accepted 16 June 2013

Keywords:
Above-ground
Ecosystem services
Spatial ecology
Urban forest
Urbanisation

ABSTRACT

Although urbanisation is a major cause of land-use change worldwide, towns and cities remain relatively understudied ecosystems. Research into urban ecosystem service provision is still an emerging field, yet evidence is accumulating rapidly to suggest that the biological carbon stores in cities are more substantial than previously assumed. However, as more vegetation carbon densities are derived, substantial variability between these estimates is becoming apparent. Here, we review procedural differences evident in the literature, which may be drivers of variation in carbon storage assessments. Additionally, we quantify the impact that some of these different approaches may have when extrapolating carbon figures derived from surveys up to a city-wide scale. To understand how/why carbon stocks vary within and between cities, researchers need to use more uniform methods to estimate stores and relate this quantitatively to standardised 'urbanisation' metrics, in order to facilitate comparisons.

1. Introduction

Globally, the urban human population has expanded rapidly in recent decades, with over half of people now living in towns and cities (United Nations, 2012). In turn, this has been accompanied by high rates of land conversion to urban areas (Seto et al., 2012). With urbanisation set to continue, the need to understand and quantify ecosystem service provision within cities is increasingly acknowledged as being highly apposite to the lives of inhabitants, and essential in helping to tackle the environmental and social challenges they experience (Gaston, 2010a).

One particular ecosystem service that has become a high-profile feature of climate change mitigation efforts is carbon storage within soils and vegetation (e.g., Schimel, 1995; Grimm et al., 2008). Indeed, to fulfil international reporting obligations (e.g., UN Convention on Climate Change and Kyoto protocol) and national reduction targets, many countries must produce inventories of greenhouse gas emissions by sources and removal by sinks, including accounting for biological carbon losses and sequestration arising from different land-uses and their conversion (Dyson et al., 2009). As the bulk of carbon emissions can be attributed to urban areas (International Energy Agency, 2008; Satterthwaite, 2008; Kennedy et al., 2010), the policies and actions of the local authorities that administer towns and cities are central to meeting the required cuts. However, in order to achieve measureable reductions in the long-term, reliable baseline assessments of carbon stocks need to be available. Only then can it be established whether interventions such as tree planting strategies and land development policies (e.g., Churkina et al., 2010; Escobedo et al., 2011; Pataki et al., 2011; Raciti et al., 2012a) can be advocated as effective tools that go some way to offsetting the emissions of urban inhabitants.

Although considerably smaller than carbon emissions per unit area, there a is growing consensus that urban biological carbon stocks warrant further investigation, as they are more substantial than previously assumed (e.g., Nowak and Crane, 2002; Pataki et al., 2006; Davies et al., 2011; Hutyra et al., 2011; Raciti et al., 2012b). However, as this relatively new field of research begins to expand and more urban carbon density measurements are derived, variability between estimates is becoming apparent. Whilst this is not unexpected, because carbon densities will be influenced by a range

* Corresponding author.
E-mail addresses: z.g.davies@kent.ac.uk (Z.G. Davies), mada@ifro.ku.dk (M. Dallimer), j.edmondson@sheffield.ac.uk (J.L. Edmondson), j.r.leake@sheffield.ac.uk (J.R. Leake), k.j.gaston@exeter.ac.uk (K.J. Gaston).

0269-7491/$ – see front matter © 2013 Elsevier Ltd. All rights reserved.
http://dx.doi.org/10.1016/j.envpol.2013.06.005

Table 1
The landcover map resolutions, landcover definitions and biomass estimation procedures used in the 13 independent studies that have generated vegetation carbon storage estimates for urban areas since 2000. The landcover terminology used in each individual article is retained and denoted by capitalisation and highlighting in italics; please refer to the relevant paper for detailed definitions.

Study	Resolution of underlying landcover map	Public and private land surveyed?	Definitions of forest/canopy landcover(s)	Inclusion of forest height/age into landcover definition	Minimum tree size recorded	Plot size	Allometric equations	Use of urban tree biomass correction factor[a]	Inclusion of root biomass into calculation	Inclusion of herbaceous vegetation carbon stocks
Jo (2002)	225 m^2 (100 m^2 for Junglang district)	Yes	Defined as: *Agricultural*; *Natural*; *Institutional Vegetation Dominated*, and; *Recreational Vegetation Dominated*	No	All woody plants measured (defined as shrubs for DBH < 2 cm)	225 m^2–600 m^2	From the literature. Equations developed for four tree and five shrub species	No	Yes	Estimate not stated separately
Nowak and Crane (2002)	1 km^2	Not stated	A single forest category	No	Methods follow Nowak and Crane (2000) who use a DBH > 2.54 cm	400 m^2	From the literature	Yes	Yes	Not estimated
Guan and Chen (2003a, b)	Not stated	Not stated	A single forest category	No	Not stated	100 m^2	From the literature		Not stated	Not estimated
Yang et al. (2005)	Landsat (30 m)	No - plots could not be surveyed on government land	A single tree/shrub category	No	Not stated	400 m^2	From the literature	Yes	Yes	Not estimated
Golubiewski (2006)	NA	No - private greenspaces only	NA	NA	All woody plants measured	387 m^2–22028 m^2	From the literature, including those for urban trees	Yes	No	0.282 kg C m^{-2}
Escobedo et al. (2010)	Not stated	Not stated	A single forest category	No	DBH > 2.5 cm	400 m^2 (100 m^2 for "even-aged, dense pine rockland, mangrove and *Melaleuca quinquinervia* plots in Miami-Dade)	From the literature	Yes	Yes	Not estimated
Zhao et al. (2010)	Not stated	Not stated	Forests defined based on age and species composition	Age	DBH > 4 cm	Not stated	Biomass equations stated in paper	No	No	Not estimated
Davies et al. (2011)	0.25 m^2	Yes	Three categories based on height: < 2 m *Shrub*; 2–5 m *Tall Shrub*, and; > 5 m *Trees*	Height	DBH > 1 cm	25 m^2	From the literature	No	No	0.14 kg C m^{-2}
Hutyra et al. (2011)	Landcover map (30 m); canopy cover map (0.46 m)	Yes	*Mixed* or *Conifer* forest categories	No	DBH > 5 cm	707 m^2 (15 m radius circle)	From the literature	Yes. For field plots containing < 7 trees	No	Not estimated
Ren et al. (2011)	1:10000 map	Not stated	Nine forest types based on species composition	Age	Not stated	Not stated	Biomass expansion factors convert landcover into biomass		No	Not estimated
Liu and Li (2012)	QuickBird (0.6 m)	Not stated	Five forest types based on forest function	No	Not stated	800 m^2–9300 m^2	From the literature	Yes. For large trees (DBH > 30 cm)	Yes	Not estimated
Raciti et al. (2012b)	30 m	Yes	A single forest category	No	DBH > 5 cm	707 m^2 (15 m radius circle)	From the literature	No	No	Not estimated
Strohbach and Haase (2012)	Land-cover map (minimum patch size 0.25 ha); canopy cover map (0.4 m)	Yes	Seven forest/woodland types, based on species composition/structure	No	DBH > 5 cm	707 m^2 (15 m radius circle)	From the literature	Yes. For trees growing in human dominated landcovers	No	Not estimated

[a] An arbitrary tree biomass correction factor has been used in some studies since Nowak 1994, to account for the fact that urban trees are often open-grown and/or maintained, which may result in a lower biomass than would be predicted by allometric equations derived from forest-grown trees of the same DBH.

of intrinsic and extrinsic spatio-temporal factors (e.g., interactions with the prevailing climate, regional patterns and histories of urbanisation, human population densities, land management), it is currently difficult to compare values across studies meaningfully, due to the assortment of methodological approaches used. A recent study has illustrated the problem by highlighting the discrepancies that may arise as a result of inconsistent definitions of 'urban' land-use (Raciti et al., 2012b). In this paper, we review additional procedural differences, evident in the literature, which are potential sources of variability in vegetation carbon density assessments. Furthermore, we quantify the impact that some of these different approaches may have when extrapolating carbon estimates derived from surveys up to a city-wide scale.

2. Materials and methods

2.1. Review of the urban vegetation carbon storage literature

To identify potential sources of variability in published urban vegetation carbon density estimates, we review the procedural differences evident in the methods sections of peer-reviewed literature (e.g., resolution of the underlying spatial data, definitions of urban areas, use of correction factors to be applied to urban tree biomass estimates). The following search terms and Boolean operators were used in the ISI Web of Science database to identify studies suitable for inclusion: *urban** AND *above* AND *ground* AND *vegetation*; *urban** AND *above-ground* AND *vegetation*; *urban** AND *aboveground* AND *vegetation*; *urban** AND *above* AND *ground* AND *carbon*; *urban** AND *above-ground* AND *carbon*; *urban** AND *aboveground* AND *carbon*; *urban** AND *vegetation* AND *carbon*; *urban** AND *forest** AND *carbon*. All English language journal articles published since 2000 (up to June 2012) were evaluated. Whilst we acknowledge that this approach is restricted (as it is bounded by date and limited to searching one major bibliographic database), the purpose of this exercise was not to undertake an extensive systematic review (Pullin and Stewart, 2006), but to sample the methods that have been applied to date in this field of study in a manner which was free of the bias that can be associated with 'selecting' literature.

The searches returned 703 unique records. The title of each paper was checked for relevance, removing those focussed solely on airborne/water pollutants, soil carbon or landcover mapping. In total, 523 studies were discarded, before the abstracts of the remaining 180 articles were filtered for exclusion by two independent reviewers, based on the same criteria. Subsequently, the full text of 73 studies was scrutinised for suitability. However, most of the material examined net primary production or carbon sequestration and, therefore, is not included in the final selection of publications. We also removed studies that were either reviews of previous work or just extrapolated carbon estimates originating from existing literature. Finally, we picked a single paper from research teams that have published multiple articles based on the same methodological approach and broadly within the same geographic region. The final sample comprised 13 articles.

From each study, we extracted information pertaining to the resolution of the underlying landcover maps, landcover definitions and methods employed to convert field survey data into vegetation carbon storage estimates (Table 1). In addition, for every surveyed city, we summarised the outcomes of the research, including tree and carbon density estimates, as well as the figure for total carbon stored across the urban area (Table 2).

2.2. Assessing the impact of spatial data resolution and extent of landcover categorisation on urban vegetation carbon storage estimates

From the literature review, three potential sources of variability in vegetation carbon density estimates were identified that could be examined quantitatively using the dataset from Davies et al. (2011): (i) the resolution of the underlying landcover map; (ii) the land tenure of surveyed areas, and; (iii) the extent to which forest/canopy landcover classes are refined.

Leicester was used as a case study for this research. It is a typical mid-sized British city, covering an area of approximately 73 km^2 and is home to a human population of c. 310 000 (Leicester City Council, 2012). Geographically, it is located in central England (52°38′N, 1°08′W) and we considered the urban area to comprise all land falling within the unitary authority (municipality) boundary (Fig. 1). Full details regarding the material and methods followed to generate above-ground carbon storage estimates for the city can be found in Davies et al. (2011), and are summarised in Tables 1 and 2.

Landcover characteristics across the city of Leicester were established using a Geographic Information System (GIS) which consisted of a digital vector cartographic dataset produced by Infoterra called LandBase (http://www.infoterra.co.uk/landbase). Within LandBase, each individual polygon (accurate to 0.25 m^2) is assigned into one of eight landcover classes (*Inland Water*, *Bare Ground*, *Artificial Surface*, *Buildings*, *Herbaceous Vegetation*, *Shrubs*, *Tall Shrubs* and *Trees*). The four types of above-ground vegetation categories are effectively stratified by maximum vegetation height (classified using high resolution, 4–8 points per metre, LiDar data): *Herbaceous Vegetation* (grasses and non-woody plants), *Shrubs* (woody bushes and trees with a mean height typically <2 m), *Tall Shrubs* (woody bushes and trees with a mean height generally 2–5 m) and *Trees* (trees >5 m tall). Davies et al. (2011) justified this system of categorisation as vegetation height is indicative of biomass, especially when refined using measurements of tree density (Mette et al., 2003).

To quantify the effect of using coarse resolution spatial data to underpin the extrapolation of field-derived carbon densities up to a city-wide value, the LandBase landcover information was aggregated into six progressively larger scale maps, with grid square sizes of: (i) 10 × 10 m; (ii) 25 × 25 m; (iii) 50 × 50 m; (iv) 100 × 100 m; (v) 250 × 250 m, and; (vi) 1000 × 1000 m. For each map, these 'pixels' were categorised according to the predominant landcover (Fig. 2), and the areal extent of each landcover across Leicester was subsequently obtained by multiplying together the number of assigned squares and grid square area. These landcover totals could then be multiplied by the appropriate estimated carbon density (0.00 kg C m^{-2} for *Inland Water*, *Bare Ground*, *Artificial Surface* and *Buildings*; 0.14 kg C m^{-2} for *Herbaceous Vegetation*; 10.54, 13.41 and 28.38 kg C m^{-2} for *Shrubs*, *Tall Shrubs* and *Trees* respectively; Davies et al., 2011) and summed to produce a vegetation carbon store for the entire urban area. To make this analysis more directly comparable to the rest of the literature we did not account for land tenure (Table 1).

3. Results and discussion

Across the city of Leicester, the principal landcover classes within the high-resolution LandBase vector dataset were *Herbaceous Vegetation*, *Artificial Surface*, *Buildings* and *Trees*, with city-wide areal extents of 37.5, 27.4, 15.2 and 10.6% respectively (Fig. 1). Landcover categories with smaller percentage contributions included *Shrubs* (7.1%), *Tall Shrubs* (1.3%), *Inland Water* (0.6%) and *Bare Ground* (0.3%). If the landcover classes throughout the city were derived using increasingly coarse resolution maps, with each pixel being defined by the predominant landcover present in the cell, these figures polarise, with *Herbaceous Vegetation* and *Artificial Surface* comprising ever more of the urban area at the expense of the other less well represented categories (Table 3, Figs. 2 and 3). Indeed, when grid squares of 250 × 250 m were considered, over 70% of the coverage by *Trees*, *Tall Shrubs*, *Shrubs*, *Inland Water* and *Bare Ground* had been effectively 'lost'. This highlights the degree to which broad scale spatial data fail to capture the finely grained landcover mosaic that typifies urbanised landscapes (Cadenasso et al., 2007; Gill et al., 2008). For example, trees are often distributed widely across urban areas, even being planted individually in the centre of cities where there are no 'greenspaces' per se. Consequently, high resolution satellite imagery is frequently recommended for characterising urban landcover (e.g., Nichol and Lee, 2005; As-syakur et al., 2010; Zheng et al., 2012).

The knock-on implications for extrapolating field-derived vegetation carbon densities up to a city-wide value are far from trivial (Table 4, Fig. 4); at the 250 × 250 m grid square scale, the quantity of above-ground carbon stored across Leicester would be underestimated by 76%. The impact of this phenomenon may be greater within the UK and some other parts of Europe, where the pattern of urbanisation has a propensity towards the densification of existing urban areas (e.g., Goode, 2006; Dallimer et al., 2011; Siedentop and Fina, 2012), and thereby reducing/constraining the size of vegetation patches, rather than the more dispersed 'sprawl' of settlements at the periphery of towns and cities as exemplified by North American urban expansion (e.g., Hansen et al., 2005; White et al., 2009; Gaston, 2010b). More recent studies focussed on generating vegetation carbon densities have used higher resolution maps as the underlying basis for their calculations, probably reflecting the fact that such digital information is becoming less prohibitive in terms of cost and easier to access/process (Table 1). Nonetheless, the wide range of spatial scales that have been employed to date reduces the appropriateness of comparisons between published estimates (Table 1).

In addition, there are inconsistencies in regard to how tree cover is characterised across urban areas (Table 1), with categories

Table 2
The urban area definitions, tree densities, vegetation carbon densities and total stored carbon estimates generated by the 13 independent studies published on this topic since 2000. The landcover and sample size terminology used in each individual article is retained and denoted by capitalisation and highlighting in italics; please refer to the relevant paper for detailed definitions. All units have been converted to tonnes of carbon (t C) for total carbon stored, trees per hectare (trees ha^{-1}) for tree density, and kilograms of carbon per metre squared (kg C m^{-2}) for carbon density, to facilitate comparisons across studies. Values in parentheses represent standard errors (unless otherwise stated), if reported in the original paper.

Study	Country	City	Definition of urban extent	Urban area (km^2)	Sample size	Canopy/tree cover (%)	Tree density: urban (trees ha^{-1})
Jo (2002)	Korea	Chuncheon	Political	52	149 (107 in *Urban* areas; 42 in *Natural* areas)	12–13% for *Urban* areas	150 (20)
	Korea	Kangleung	Political	76	173 (108 in *Urban* areas; 65 in *Natural* areas)	12–13% for *Urban* areas	150 (20)
	Korea	Kangnam district (Seoul)	Political	39.6	111 (80 for *Urban* areas; 31 for *Natural* areas)	12–13% for *Urban* areas	330 (40)
	Korea	Junglang district (Seoul)	Political	18.5	86 (65 for *Urban* areas; 21 for *Natural* areas)	12–13% for *Urban* areas	310 (40)
Nowak and Crane (2002)[a]	USA	Atlanta	Population density	Not stated	*Approximately 200*	36.7% (2.0)	276 (22)
	USA	Baltimore	Population density	Not stated	*Approximately 200*	12.5% (2.5)	136 (29)
	USA	Syracuse	Population density	Not stated	*Approximately 200*	Not stated	137 (19)
	USA	Boston	Population density	Not stated	*Approximately 200*	22.3% (1.8)	83 (8)
	USA	Chicago	Population density	Not stated	*Approximately 200*	11.0% (0.2)	68 (10)
	USA	Jersey City	Population density	Not stated	*Approximately 200*	Not stated	36 (6)
	USA	New York	Population density	Not stated	*Approximately 200*	20.9% (2.0)	65(9)
	USA	Oakland	Population density	Not stated	*Approximately 200*	21.0% (0.2)	120 (4)
	USA	Philadelphia	Population density	Not stated	*Approximately 200*	15.7% (1.3)	62 (6)
	USA	Sacramento	Population density	Not stated	*Approximately 200*	13%	73 (15)
Guan and Chen (2003a,b)	China	Guangzhou	Political	1443.6	10 plots for each landcover class	Vegetation cover 24.1% in built-up area; 66.7% in total area	Not stated
Yang et al. (2005)	China	Beijing	Political/ Physical	301.8	250	17%	Not stated
Golubiewski (2006)	USA	Denver-Boulder	Political	Not stated	53	NA	746 (73) stems ha^{-1}
Escobedo et al. (2010)	USA	Miami-Dade	Political	1273	229	14%	227
	USA	Gainsville	Political	122	93	51%	374
Zhao et al. (2010)	China	Hangzhou	Political	16 900	Not stated	Not stated	Not stated
Davies et al. (2011)	UK	Leicester	Political	73	347	19%	Between 0 and 1800 (400) depending on landcover
Hutyra et al. (2011)	USA	Seattle Region	Impervious surface cover	NA	154	44% of the study region	297 (95% CI : 51.3) stems/ha across study region
Ren et al. (2011)	China	Xiamen	Political	Not stated	Not stated	46%	Not stated
Liu and Li (2012)	China	Shenyang	Physical	455	213 (30 in *Ecological and Public Welfare Forests*; 46 in *Attached Forests*; 74 in *Landscape and Relaxation Forests*; 14 in *Production and Management Forests*; 49 in *Road Forests*)	22.28%	Not stated
Raciti et al. (2012b)	USA	Boston	Impervious surface cover in a 990 m^2 moving window	6231	139	41.7%	Not stated
	USA	Boston	Impervious surface cover in a 270 m^2 moving window	6231	139	41.7%	Not stated

Tree density: canopy cover (trees ha^{-1})	Overall carbon density (kg C m^{-2})	Canopy/forest cover carbon density (kgCm^{-2})	Urban carbon density (kg C m^{-2})	Landcover carbon density (kg C m^{-2})	Total carbon stored (t C)
990 (80) for *Natural* areas	No overall figure stated	2.60 (0.27) for *Natural* areas	0.47 (0.07) for *Urban* areas	No further detail given	63 000
1000 (60) for *Natural* areas	No overall figure stated	4.67 (0.39) for *Natural* areas	0.63 (0.08) for *Urban* areas		166 000
1290 (140) for *Natural* areas	No overall figure stated	6.01 (0.57) for *Natural* areas	0.66 (0.07) for *Urban* areas		60 000
1320 (140) for *Natural* areas	No overall figure stated	5.87 (0.56) for *Natural* areas	0.72 (0.13) for *Urban* areas		42 000
751	3.574 (0.269)	9.7 (0.7)	No further detail given	No further detail given	1 220 200 (91 900)
508	2.528 (0.316)	10.0 (1.3)			528 700 (66 100)
Not stated	2.282 (0.249)	9.4 (1.0)			148 300 (16 200)
372	2.030 (0.257)	9.1 (1.1)			289 800 (36 700)
618	1.419 (0.214)	12.9 (1.9)			854 800 (129 100)
Not stated	0.502 (0.068)	4.4 (0.6)			19 300 (2600)
312	1.533 (0.189)	7.3 (0.9)			1 225 200 (150 500)
570	1.101 (0.037)	5.2 (0.2)			145 800 (4900)
394	1.409 (0.142)	9.0 (0.9)			481 000 (48 400)
Not stated	4.691 (2.264)	36.1 (17.4)			1 107 300 (532 600)
Not stated	1.378	2.661	3.21 for *Built-up*; 3.955 for *Roadsides*; 4.273 for *Urban Parks*	0.288 for *Cultivated Areas*) to 4.273 for *Urban Parks*	1 330 000
79 (10)	Not stated	0.744	1.281 for *Old City* to 0.302 for *Third Ring Road Area*	No further detail given	224 200 (34 100)
NA	2.07 (0.294)	NA	NA	NA	NA
No *Forest* land use	0.93	No *Forest* land use	0.39 for *Transportation* to 14.1 for *Institution*	0.15 for *Wetland/ Water* to 3.01 for *Park*	467 728
767	3.84	7.47	0.92 for *Industrial* to 4.82 for *Institution*	0.22 for *Utility* to 7.47 for *Forest*	1 497 679
Not stated	3.025 (2.335)	1.822 (0.559) to 3.587 (1.695) depending on forest type	1.888 to 4.727 depending on urban district	Between 0.58 and 8.727 depending on forest type and stand age	11 740 000 (1 080 000)
1300 (300) to 1800 (400) depending on canopy cover type	3.16 (95% CI: 2.65–3.62)	6.66 (1.70) for publicly owned *Shrub* category to 28.86 (4.36) for publicly owned *Tree* category	No further detail given	From 0.14 (0.01) for mixed ownership *Herbaceous Vegetation* to 28.86 (4.36) for publicly owned *Tree*	232 521 (95% CI: 195 914 – 267 130)
297 (95% CI: 51.3) stems/ha across study region	8.9 (95% CI: 2.2)	14.0 (95% CI: 4.0)	All urban landcovers 1.8 (1.4). *Heavy Urban* 0.2 (0.19); *Medium Urban* 1.3 (1.2); *Low Urban* 3.8 (3.8).	*Mixed* forest 9.8 (4.1); *Conifer* forest 18.2 (6.0).	None given
527	1.514 (0.229)	Not stated	*Urban Core* 2.076 (0.1); *Suburbs* 1.705 (0.241); *Exurbs* 1.437 (0.234)	From 0.51 (sd: 0.04) to 9.34 (sd: 0.65) depending on forest type/age	1 139 528
569 (90)	3.322 (0.432)	3.322 (0.432)	No further detail given	1.317 (0.371) to 5.017 (0.550) depending on forest type	336 680 (43 820)
Not stated	7.2 (0.4)	*Non-Urban Forest*: 11.7 (0.5), *Low Population Density Urban Forest*: 10.4 (0.6), *High Population Density Urban Forest*: 9.2 (0.5)	From 0.6 (0.2) to 5.2 (0.6) depending on land use/degree of urbanisation	11.7 (0.5) for *Non-Urban Forest* to 0.6 (0.2) for *Other Developed Land* in High Population urban areas	4 200 000 (400 000)
Not stated	6.4 (0.5)	8.2 (0.7) for *Non-Urban* landcover	3.7 (0.7) for *Urban* landcover	3.7 (0.7) *Urban* to 8.2 (0.7) *Non-Urban*	4 600 000 (900 000)

Table 2 (*continued*)

Study	Country	City	Definition of urban extent	Urban area (km^2)	Sample size	Canopy/tree cover (%)	Tree density: urban (trees ha^{-1})
	USA	Boston	Impervious surface cover in a 90 m^2 moving window	6231	139	41.7%	Not stated
	USA	Boston	Population density	6231	139	41.7%	Not stated
	USA	Boston	Night-time lighting data	6231	139	41.7%	Not stated
Strohbach and Haase (2012)	Germany	Leipzig	Political	297	190 (10 plots for each of 19 landcover classes)	19%	Not stated

[a] For cities where details were not given in Nowak and Crane (2002), tree density and canopy cover were taken from Nowak et al. (2001).

delimited according to descriptors as simple as the presence of generic 'canopy cover' (e.g., Yang et al., 2005; Escobedo et al., 2010), through to more resolved classifications distinguished by physical traits (e.g., age, Zhao et al., 2010; height, Davies et al., 2011) and forest types (e.g., coniferous/mixed/broad-leaved, Strohbach and Haase, 2012; species composition, Zhao et al., 2010; Ren et al., 2011; land-use, Liu and Li, 2012). Furthermore, some studies also explicitly disaggregate landcover estimates by land tenure, which has been recognised as another potential source of variability when calculating biological carbon stocks (Table 1).

To illustrate, Davies et al. (2011) evaluated carbon density estimates for sites that were either of public (i.e., areas such as roadside verges, copses, parks and recreation grounds maintained by the local authority) or mixed ownership (e.g., belonging to corporations, private individuals and abandoned industrial sites), as well as private domestic household gardens, with the aim of determining whether any disparities were apparent as a consequence of differences in vegetation structure, composition and/or management. This exercise demonstrated that trees within domestic gardens were notably smaller than those in other landcover-land ownership classes, resulting in significantly lower carbon densities. Not accounting for land tenure would have led to an overestimate for total carbon stored across Leicester of 27% (293 239 rather than 231 521 tonnes; Tables 2 and 4). Likewise, if all tree cover had been defined by a single homogenous 'canopy' class, comprising 19% of the Leicester area (Table 2), mean carbon density would have been

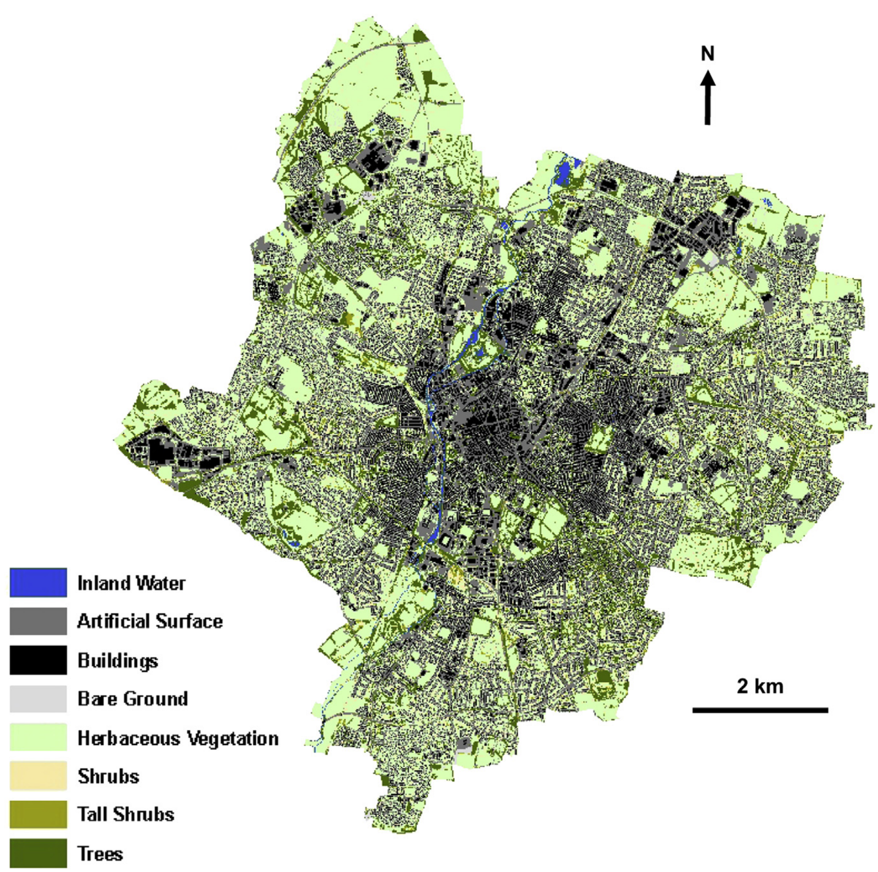

Fig. 1. Landcover across the city of Leicester, comprised of polygons classified by Infoterra in their LandBase digital cartographic dataset (www.infoterra.co.uk/landbase).

Tree density: canopy cover (trees ha^{-1})	Overall carbon density (kg C m^{-2})	Canopy/forest cover carbon density (kgCm^{-2})	Urban carbon density (kg C m^{-2})	Landcover carbon density (kg C m^{-2})	Total carbon stored (t C)
Not stated	7.1 (0.7)	9.7 (1.0) for *Non-Urban* landcover	3.5 (0.7) for *Urban* landcover	3.5 (0.7) *Urban* to 9.7 (1.0) *Non-Urban*	4 900 000 (1 000 000)
Not stated	6.4 (0.7)	8.8 (1.2) for *Non-Urban* landcover	6.6 (0.8) for *Urban* landcover	6.6 (0.8) *Urban* to 8.8 (1.2) *Non Urban*	26 500 000 (3 300 000)
Not stated	6.4 (0.7)	9.2 (1.7) for *Non-Urban* landcover	6.6 (0.8) for *Urban* landcover	6.6 (0.8) *Urban* to 9.2 (1.7) *Non Urban*	27 300 000 (32 000 000)
Not stated	1.181 (0.325)	6.82 (0.142)	From 0.1 to 4.9 depending on district	0.402 (0.113) for *Afforestation* to 9.826 (1.532) for *Riparian Forest*	316 000 (66 000)

estimated at 16.16 (S.E. = 1.14) kg C m^{-2}, rather than being recorded up to 28.86 (S.E. = 4.36) kg C m^{-2} for the more highly resolved landcover-land ownership category of *Trees* occurring on publicly owned/managed land (Davies et al., 2011). The implication of this would have been a 17% underestimate in the city-wide carbon stock. This mirrors the findings of Hutyra et al. (2011), who warned against using generic 'canopy cover' to estimate above-ground biomass, as it explained only 61% of the variation in the data. A balance needs to be struck, therefore, between capturing the heterogeneity associated with urban vegetation, and minimising the number of distinct classes used for categorisation, in order to allow the same methodological approach to be applied between cities. Moreover, this will have an important bearing on the size of plots used for sampling (Tables 1 and 2); quadrats need to be sufficiently large to be representative of the vegetation occurring in larger patches, but not excessively big to cover smaller discrete patches (e.g. using a 0.04 ha plot if average patch sizes are ~50 m^2), as well as small enough to maximise replication.

This situation can be further exacerbated by how authors choose to demarcate the boundaries of their urban area (Table 2). If the objective of the research is to inform local authority/municipality climate mitigation strategies, then the extent of the town or city is best defined by the borders of political entities, as this is the scale most germane for landscape planning, policy-making and management. However, this does not lend itself readily to comparisons between studies, because the distribution and total coverage of different land-uses (e.g., woodland/forests, agricultural land, industrial sites) are liable to vary substantially. To echo the call made

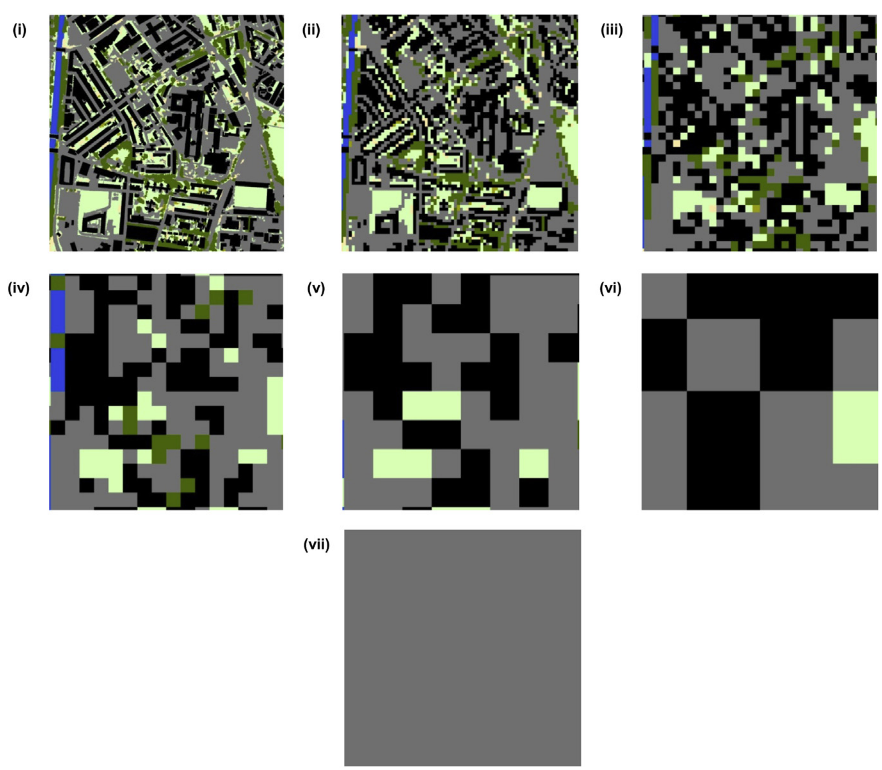

Fig. 2. An illustrative 1000 × 1000 m area from the LandBase digital cartographic dataset for Leicester which demonstrates how landcover would be assigned, based on the predominant category per pixel, at the following grid square resolutions: (i) original vector dataset (accurate to 0.25 m^2); (ii) 10 × 10 m; (iii) 25 × 25 m; (iv) 50 × 50 m; (v) 100 × 100 m; (vi) 250 × 250 m, and; (vii) 1000 × 1000 m. Despite the 1000 × 1000 m area being a mosaic of different landcovers, more of it is assigned to *Artificial Surface* as pixel size increases.

Table 3
The change (i) areal extent (ha) and (ii) proportional coverage (benchmarked against the original vector dataset) of each landcover class across the city of Leicester with progressively coarse spatial resolution maps. Each grid square is classified according to the predominant landcover. The original vector dataset polygons are accurate to 0.25 m^2.

	Resolution (m)	Inland Water	Artificial Surface	Buildings	Bare Ground	Herbaceous Vegetation	Shrubs	Tall Shrubs	Trees
(i)	Vector	41.58	2007.59	1115.61	19.61	2747.18	524.17	96.12	779.62
	10	42.19	2073.80	1143.24	16.65	2762.92	406.66	93.68	792.35
	25	42.42	2364.24	892.48	12.95	3003.90	179.11	67.65	768.74
	50	37.92	2537.37	770.10	10.63	3233.18	39.52	26.52	676.25
	100	24.19	2551.26	706.73	12.67	3484.46	12.15	10.10	529.93
	250	9.35	2464.74	643.75	1.01	3977.04	6.25	0.41	228.95
	1000	0.00	2225.47	300.00	1.01	4805.00	0.00	0.00	0.00
(ii)	10	0.01	0.03	0.02	−0.15	0.01	−0.22	−0.03	0.02
	25	0.02	0.18	−0.20	−0.34	0.09	−0.66	−0.30	−0.01
	50	−0.09	0.26	−0.31	−0.46	0.18	−0.92	−0.72	−0.13
	100	−0.42	0.27	−0.37	−0.35	0.27	−0.98	−0.89	−0.32
	250	−0.78	0.23	−0.42	−0.95	0.45	−0.99	−1.00	−0.71
	1000	−1.00	0.11	−0.73	−0.95	0.75	−1.00	−1.00	−1.00

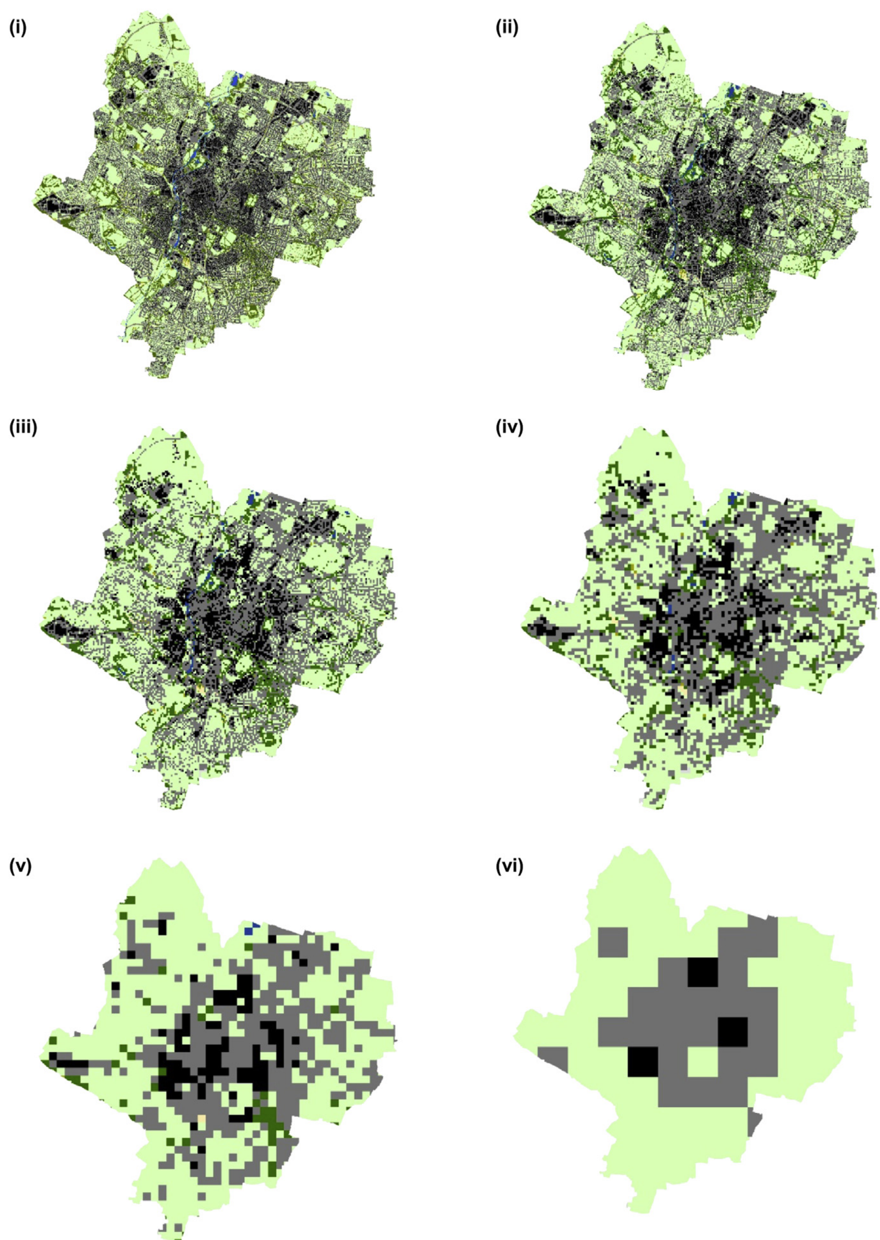

Fig. 3. The landcover across the city of Leicester, where each grid square is classified according to the most predominant landcover, for six progressively coarse spatial scale maps of the following resolutions: (i) 10 × 10 m; (ii) 25 × 25 m; (iii) 50 × 50 m; (iv) 100 × 100 m; (v) 250 × 250 m, and; (vi) 1000 × 1000 m.

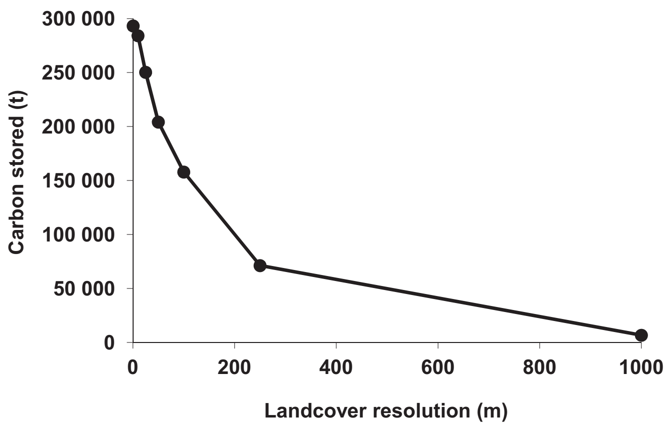

Fig. 4. Change in the estimated quantity of above-ground carbon stored in vegetation across Leicester, as a consequence of using progressively coarse spatial resolution maps to classify landcover per grid square and extrapolate field-derived carbon densities up to a city-wide value. Note that the total value for stored carbon differs from the more accurate estimate in Davies et al. (2011) as it neglects to account for land ownership, in order to be comparable with the majority of other published studies (Table 1).

by Raciti et al. (2012b), ecologists need to be less equivocal when using terminology such as 'urban', 'urbanised', 'suburban', 'developed' and 'rural' to describe sampling locations within the boundary of their study area (Table 2), ensuring they are clearly defined and quantified via easily obtainable metrics that are directly related to their biophysical and/or socio-economic state (e.g., proportion of hard surface per unit area, human population density), in order to facilitate analyses both within and among cities.

As the vast majority of urban vegetation carbon stocks are attributable to trees, rather than herbaceous and woody material (e.g., Golubiewski, 2006; Davies et al., 2011), this is another possible origin of variability in the estimates generated in different studies. Carbon densities are a function of tree density and size. While both metrics are straightforward to ascertain, comprising of the number of stems per unit area and DBH (diameter at breast height, measured at 1.30 m from ground level) respectively, there is no constant threshold above which a tree is deemed sufficiently large to be sampled (Table 1). Whilst small individuals only store relatively limited quantities of carbon, they make up a large proportion of the total number of trees in urban areas (e.g., Nowak, 1993, 1994; Britt and Johnson, 2008; Davies et al., 2011; Hutyra et al., 2011), so excluding them might have repercussions on estimated tree densities (Table 2) and scaling up of carbon to a city-wide scale. Thus, for example, the disparities evident between tree densities recorded in the UK ($\sim$0.15 trees m^{-2}; Davies et al., 2011) and US ($\sim$0.05 trees m^{-2}; Nowak and Crane, 2002) may represent genuine variation in the study systems, or simply be an artefact of the sampling protocol followed.

Table 4
Proportional change (benchmarked against the original vector dataset; all differences are statistically significant at $p < 0.001$) in the amount of above-ground carbon stored in vegetation across Leicester, as a consequence of using progressively coarse spatial resolution maps to classify landcover per grid square and extrapolate field-derived carbon densities up to a city-wide value. The original vector dataset polygons are accurate to 0.25 m^2. Note that the total value for stored carbon differs from the more accurate estimate of 231 521 tonnes in Davies et al. (2011) as it neglects to account for land ownership, in order to be comparable with the majority of other published studies (Table 3).

Resolution (m)	Carbon stored (t)	Proportional change
Vector	293 239	–
10	284 161	−0.03
25	250 324	−0.15
50	204 168	−0.30
100	157 907	−0.46
250	71 256	−0.76
1000	6727	−0.98

Most urban vegetation carbon estimates rely on allometric equations obtained from the literature to convert survey measurements of tree size to biomass (Table 1). Currently, very few of these predictors exist specifically for urban trees, being derived principally from forested areas in Europe and North America (e.g., Ter-Mikaelian and Korzukhin, 1997; Zianis et al., 2005; Snorrason and Einarsson, 2006). Furthermore, the range of species for which allometric equations are available tends to be restricted to those of commercial value in the forestry industry. The findings of McHale et al. (2009) suggest that bias may be integrated into carbon accounting as an outcome of applying such equations, but that it is difficult to forecast; for particular species, the allometries determined from trees grown in woodland stands underestimate the biomass for urban specimens, while for others they overestimate it. Until a suite of explicit urban tree predictors are developed, which would be a major, and probably unrealistic, undertaking (given the extreme biophysical/socio-economic diversity of cities worldwide and tree species therein), little can be done to lessen this problem. However, wherever possible, McHale et al. (2009) recommend that the likelihood of incorporating any systematic bias in biomass estimates for each species should be decreased by using generalised results from a group of equations derived from different studies. Additionally, a more consistent approach needs to be adopted in relation to the use, or not, of urban tree biomass correction factors, and the inclusion/exclusion of herbaceous vegetation and tree root systems (Table 1).

Despite urbanisation being a major agent of land-use change worldwide, towns and cities remain relatively understudied ecosystems from an ecological perspective (Gaston, 2010a). Our investigation of the peer-reviewed literature over the past decade demonstrates that the number of independent studies focussed on quantifying vegetation carbon storage is still small (Tables 1 and 2), even though appraisal of such stocks is a key first step in the development of local management recommendations/policies that will secure provision of this important ecosystem service. Nonetheless, the research conducted thus far has challenged the perception that urban areas are devoid of biological carbon to such an extent that it is not worth accounting for (Pataki et al., 2011). In order to garner a more mechanistic understanding of how and why carbon densities vary within and between cities, researchers need to take a more uniform approach to estimating stores and relate this quantitatively to standardised metrics that can be used to characterise 'urbanisation'. Only then will we be able to elucidate the drivers that underpin variation and compare/contrast these among regions.

Acknowledgements

This work was supported by EPSRC grant EP/F007604/1 to the 4M consortium: Measurement, Modelling, Mapping and Management: an Evidence Based Methodology for Understanding and Shrinking the Urban Carbon Footprint. The 4M consortium has five UK partners: Loughborough University, De Montfort University, Newcastle University, University of Sheffield and University of Exeter. M. Dallimer holds an EU-FP7 Marie-Curie Fellowship (Grant Number 273547). Infoterra kindly provided access to LandBase. We thank L. Hutyra and D. Nowak for useful discussions on this topic.

References

As-syakur, A.R., Osawa, T., Adnyana, I.W.S., 2010. Medium spatial resolution satellite imagery to estimate gross primary production in an urban area. Remote Sensing 2, 1496–1507.
Britt, C., Johnson, M.D., 2008. Trees in Towns II: a New Survey of Urban Trees in England and Their Condition and Management. Department for Communities and Local Government, London.

Cadenasso, M.L., Pickett, S.T.A., Schwarz, K., 2007. Spatial heterogeneity in urban ecosystems: reconceptualizing land cover and a framework for classification. Frontiers in Ecology and the Environment 5, 80–88.
Churkina, G., Brown, D.G., Keoleian, G., 2010. Carbon stored in human settlements: the conterminous United States. Global Change Biology 16, 135–143.
Dallimer, M., Tang, Z.Y., Bibby, P.R., Brindley, P., Gaston, K.J., Davies, Z.G., 2011. Temporal changes in greenspace in a highly urbanized region. Biology Letters 7, 763–766.
Davies, Z.G., Edmondson, J.L., Heinemeyer, A., Leake, J.R., Gaston, K.J., 2011. Mapping an urban ecosystem service: quantifying above-ground carbon storage at a city-wide scale. Journal of Applied Ecology 48, 1125–1134.
Dyson, K.E., Mobbs, D.C., Milne, R., 2009. Annual Inventory Estimates for the UK (WP 1.1) Inventory and Projections of UK Emissions by Sources and Removals by Sinks Due to Land Use, Land Use Change and Forestry. Department for the Environment, Food and Rural Affairs, London, pp. 13–49.
Escobedo, F.J., Kroeger, T., Wagner, J.E., 2011. Urban forests and pollution mitigation: analyzing ecosystem services and disservices. Environmental Pollution 159, 2078–2087.
Escobedo, F., Varela, S., Zhao, M., Wagner, J.E., Zipperer, W., 2010. Analyzing the efficacy of subtropical urban forests in offsetting carbon emissions from cities. Environmental Science and Policy 13, 362–372.
Gaston, K.J., 2010a. Urban ecology. In: Gaston, K.J. (Ed.), Urban Ecology. Cambridge University Press, Cambridge, pp. 1–9.
Gaston, K.J., 2010b. Urbanisation. In: Gaston, K.J. (Ed.), Urban Ecology. Cambridge University Press, Cambridge, pp. 10–34.
Gill, S.E., Handley, J.F., Ennos, A.R., Pauleit, S., Theuray, N., Lindley, S.J., 2008. Characterising the urban environment of UK cities and towns: a template for landscape planning. Landscape and Urban Planning 87, 210–222.
Golubiewski, N.E., 2006. Urbanization increases grassland carbon pools: effects of landscaping in Colorado's front range. Ecological Applications 16, 555–571.
Goode, D., 2006. Green Infrastructure: Report to the Royal Commission on Environmental Pollution.
Grimm, N.B., Faeth, S.H., Golubiewski, N.E., Redman, C.L., Wu, J.G., Bai, X.M., Briggs, J.M., 2008. Global change and the ecology of cities. Science 319, 756–760.
Guan, D.S., Chen, Y.J., 2003a. Status of urban vegetation in Guangzhou city. Journal of Forestry Research (Harbin) 14, 249–252.
Guan, D.S., Chen, Y.J., 2003b. Roles of urban vegetation on balance of carbon and oxygen in Guangzhou, China. Journal of Environmental Sciences – China 15, 155–159.
Hansen, A.J., Knight, R.L., Marzluff, J.M., Powell, S., Brown, K., Gude, P.H., Jones, A., 2005. Effects of exurban development on biodiversity: patterns, mechanisms, and research needs. Ecological Applications 15, 1893–1905.
Hutyra, L.R., Yoon, B., Alberti, M., 2011. Terrestrial carbon stocks across a gradient of urbanization: a study of the Seattle, WA region. Global Change Biology 17, 783–797.
International Energy Agency, 2008. World Energy Outlook 2008. OECD/IEA, Paris.
Jo, H.K., 2002. Impacts of urban greenspace on offsetting carbon emissions for middle Korea. Journal of Environmental Management 64, 115–126.
Kennedy, C., Steinberger, J., Gasson, B., Hansen, Y., Hillman, T., Havránek, M., Pataki, D., Phdungsilp, A., Ramaswami, A., Mendez, G.V., 2010. Methodology for inventorying greenhouse gas emissions from global cities. Energy Policy 38, 4828–4837.
Leicester City Council, 2012. http://www.leicester.gov.uk/your-council-services/council-and-democracy/city-statistics/ (last accessed Oct 2012).
Liu, C.F., Li, X.M., 2012. Carbon storage and sequestration by urban forests in Shenyang, China. Urban Forestry & Urban Greening 11, 121–128.
McHale, M.R., Burke, I.C., Lefsky, M.A., Peper, P.J., McPherson, E.G., 2009. Urban forest biomass estimates: is it important to use allometric relationships developed specifically for urban trees? Urban Ecosystems 12, 95–113.
Mette, T., Hajnsek, I., Papathanassiou, K., 2003. Height-biomass allometry in temperate forests. International Geoscience and Remote Sensing Symposium 2003. Institute of Electrical and Electronics Engineers, New York, pp. 1942–1944.
Nichol, J., Lee, C.M., 2005. Urban vegetation monitoring in Hong Kong using high resolution multispectral images. International Journal Of Remote Sensing 26, 903–918.
Nowak, D.J., 1993. Atmospheric carbon-reduction by urban trees. Journal of Environmental Management 37, 207–217.
Nowak, D.J., 1994. Atmospheric carbon dioxide reduction by Chicago's urban forest. In: McPherson, E.G., Nowak, D.J., Rowntree, R.A. (Eds.), Chicago's Urban Forest Ecosystem: Results of the Chicago Urban Forest Climate Project. USDA Forest Service General Technical Report NE-186, Radnor, PA, pp. 83–94.
Nowak, D.J., Crane, D.E., 2000. The Urban Forest Effects (UFORE) model: quantifying urban forest structure and functions. In: Hansen, M., Burk, T. (Eds.), Integrated Tools for Natural Resources Inventories in the 21st Century. U.S. Dept. of Agriculture, Forest Service, North Central Forest Experiment Station, St. Paul, MN, pp. 714–720. Gen. Tech. Rep. NC-212.
Nowak, D.J., Noble, M.H., Sisinni, S.M., Dwyer, J.F., 2001. People and trees – assessing the US urban forest resource. Journal of Forestry 99, 37–42.
Nowak, D.J., Crane, D.E., 2002. Carbon storage and sequestration by urban trees in the USA. Environmental Pollution 116, 381–389.
Pataki, D.E., Alig, R.J., Fung, A.S., Golubiewski, N.E., Kennedy, C.A., McPherson, E.G., Nowak, D.J., Pouyat, R.V., Lankao, P.R., 2006. Urban ecosystems and the North American carbon cycle. Global Change Biology 12, 2092–2102.
Pataki, D.E., Carreiro, M.M., Cherrier, J., Grulke, N.E., Jennings, V., Pincetl, S., Pouyat, R.V., Whitlow, T.H., Zipperer, W.C., 2011. Coupling biogeochemical cycles in urban environments: ecosystem services, green solutions, and misconceptions. Frontiers in Ecology and the Environment 9, 27–36.
Pullin, A.S., Stewart, G.B., 2006. Guidelines for systematic review in conservation and environmental management. Conservation Biology 20, 1647–1656.
Raciti, S.M., Fahey, T.J., Thomas, R.Q., Woodbury, P.B., Driscoll, C.T., Carranti, F.J., Foster, D.R., Gwyther, P.S., Hall, B.R., Hamburg, S.P., Jenkins, J.C., Neill, C., Peery, B.W., Quigley, E.E., Sherman, R., Vadeboncoeur, M.A., Weinstein, D.A., Wilson, G., 2012a. Local-scale carbon budgets and mitigation opportunities for the northeastern United States. Bioscience 62, 23–38.
Raciti, S.M., Hutyra, L.R., Rao, P., Finzi, A.C., 2012b. Inconsistent definitions of "urban" result in different conclusions about the size of urban carbon and nitrogen stocks. Ecological Applications 22, 1015–1035.
Ren, Y., Wei, X., Wei, X.H., Pan, J.Z., Xie, P.P., Song, X.D., Peng, D., Zhao, J.Z., 2011. Relationship between vegetation carbon storage and urbanization: a case study of Xiamen, China. Forest Ecology and Management 261, 1214–1223.
Satterthwaite, D., 2008. Cities' contribution to global warming: notes on the allocation of greenhouse gas emissions. Environment and Urbanization 20, 539–549.
Schimel, D.S., 1995. Terrestrial ecosystems and the carbon-cycle. Global Change Biology 1, 77–91.
Seto, K.C., Güneralp, B., Hutyra, L.R., 2012. Global forecasts of urban expansion to 2030 and direct impacts on biodiversity and carbon pools. Proceedings of the National Academy of Sciences USA 109, 16083–16088.
Siedentop, S., Fina, S., 2012. Who sprawls most? Exploring the patterns of urban growth across 26 European countries. Environment and Planning A 44, 2765–2784.
Snorrason, A., Einarsson, S.F., 2006. Single-tree biomass and stem volume functions for eleven tree species used in Icelandic forestry. Icelandic Agricultural Science 19, 15–24.
Strohbach, M.W., Haase, D., 2012. Above-ground carbon storage by urban trees in Leipzig, Germany: analysis of patterns in a European city. Landscape and Urban Planning 104, 95–104.
Ter-Mikaelian, M.T., Korzukhin, M.D., 1997. Biomass equations for sixty-five north American tree species. Forest Ecology and Management 97, 1–24.
United Nations, 2012. World Urbanization Prospects, the 2011 Revision. United Nations, Department of Economic and Social Affairs, Population Division, New York.
White, E.M., Morzillo, A.T., Alig, R.J., 2009. Past and projected rural land conversion in the US at state, regional, and national levels. Landscape and Urban Planning 89, 37–48.
Yang, J., McBride, J., Zhou, J., Sun, Z., 2005. The urban forest in Beijing and its role in air pollution reduction. Urban Forestry & Urban Greening 3, 65–78.
Zhao, M., Kong, Z.H., Escobedo, F.J., Gao, J., 2010. Impacts of urban forests on offsetting carbon emissions from industrial energy use in Hangzhou, China. Journal of Environmental Management 91, 807–813.
Zheng, D.L., Heath, L.S., Ducey, M.J., 2012. Potential overestimation of carbon sequestration in the forested wild land-urban interface in northern New England. Journal of Forestry 110, 105–111.
Zianis, D., Muukkonen, P., Mäkipää, R., Mencuccini, M., 2005. Biomass and stem volume equations for tree species in Europe. Silva Fennica Monographs 4, 1–63.