

Inconsistent definitions of “urban” result in different conclusions about the size of urban carbon and nitrogen stocks

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Abstract. There is conflicting evidence about the importance of urban soils and vegetation in regional C budgets that is caused, in part, by inconsistent definitions of “urban” land use. We quantified urban ecosystem contributions to C stocks in the Boston (Massachusetts, USA) Metropolitan Statistical Area (MSA) using several alternative urban definitions. Development altered aboveground and belowground C and N stocks, and the sign and magnitude of these changes varied by land use and development intensity. Aboveground biomass (live trees, dbh ≥ 5 cm) for the MSA was 7.2 ± 0.4 kg C/m² (mean \pm SE), reflecting a high proportion of forest cover. Vegetation C was highest in forest (11.6 ± 0.5 kg C/m²), followed by residential (4.6 ± 0.5 kg C/m²), and then other developed (2.0 ± 0.4 kg C/m²) land uses. Soil C (0–10 cm depth) followed the same pattern of decreasing C concentration from forest, to residential, to other developed land uses (4.1 ± 0.1 , 4.0 ± 0.2 , and 3.3 ± 0.2 kg C/m², respectively). Within a land use type, urban areas (which we defined as $>25\%$ impervious surface area [ISA] within a 1-km² moving window) generally contained less vegetation C, but slightly more soil C, than nonurban areas. Soil N concentrations were higher in urban areas than nonurban areas of the same land use type, except for residential areas, which had similarly high soil N concentrations. When we compared our definition of urban to other commonly used urban extents (U.S. Census Bureau, Global Rural–Urban Mapping Project [GRUMP], and the MSA itself), we found that urban soil (1 m depth) and vegetation C stocks spanned a wide range, from 14.4 ± 0.8 to 54.5 ± 3.4 Tg C and from 4.2 ± 0.4 to 27.3 ± 3.2 Tg C, respectively. Conclusions about the importance of urban soils and vegetation to regional C and N stocks are very sensitive to the definition of urban used by the investigators. Urban areas, regardless of definition, are rapidly expanding in their extent; a systematic understanding of how our development patterns influence ecosystems is necessary to inform future development choices.

Key words: biomass; Boston, Massachusetts, USA; carbon; development; forest; impervious surface area; nitrogen; residential land use; scale; soil; urban definition; urbanization.

INTRODUCTION

Urban areas are growing in population, land area, and ecological significance. For the first time in history, the world’s urban population is larger than the world’s rural population, and urban areas are expected to account for all population growth over the next four decades (United Nations Department of Economic and Social Affairs, Population Division 2008). In the United States, urban land area increased by almost 50% between 1982 and 1997 (Fulton et al. 2001), and almost 80% of the population now lives in urban areas (U.S. Census, available online).⁴ The most densely populated northeastern states of the USA (Rhode Island, New Jersey, Massachusetts, and Connecticut) may be 60% urban in land area by the year 2050 (Nowak and Walton

2005). The most obvious changes that accompany urbanization are increased impervious surface area (ISA) and replacement of natural vegetation or agricultural fields with lawns and gardens. In the United States, impervious surfaces cover an area nearly the size of Ohio (Elvidge et al. 2004), and the area in lawns is estimated to be even larger (Milesi et al. 2005). These changes in land use and land cover have major implications for regional and global C and N cycling, yet we are only beginning to understand how the process of urbanization influences ecosystem structure and dynamics (Kaye et al. 2006, Pickett et al. 2008).

A growing body of literature demonstrates that important environmental factors, such as temperature, growing-season length, CO₂ concentration, and N inputs can differ substantially across urban-to-rural gradients (see recent review by Pickett et al. 2011). For instance, a comparative study by Templer and McCann (2010) found that the rates of N deposition to a forested site within the Boston, Massachusetts, USA, urban core were four to five times greater than at a rural forested site. Lovett et al. (2000) found similar patterns in the

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New York City metropolitan area, where throughfall deposition of inorganic N was almost twice as high at urban sites compared to suburban and rural sites. Bettez (2009) found strong N deposition gradients along roadsides in Cape Cod, Massachusetts, with near-road areas experiencing two times greater deposition than areas only 150 m away. It is likely that major highways would be significantly greater sources of N deposition (Bettez 2009). Zhang et al. (2004) found that temperatures were significantly higher in urban areas in eastern North America, and that as a consequence, growing-season lengths were typically 15 days longer than at rural sites. Further, they found that the temperature footprint of urban areas was 2.4 times larger than the urban areas themselves. These factors, combined with well-documented increases in CO₂ concentrations in urban areas (Idso 2001, Coutts et al. 2007, George et al. 2007, Pataki et al. 2007), reflect an urban environment that foreshadows future climate change and suggests the potential for strong urban influences on plant productivity and soil processes (Carreiro and Tripler 2005).

There is evidence that urban soils and vegetation can provide important ecosystem services. Urban vegetation can provide environmental benefits that include carbon (C) storage (Nowak and Crane 2001, Pataki et al. 2006, Hutrya et al. 2011a), decreased stormwater runoff (Xiao and McPherson 2002), reduced airborne particulates (Nowak et al. 2006), ultraviolet (UV) protection, building energy savings, mitigation of urban heat island effects, buffering of wind and noise, and aesthetic value (McPherson et al. 2005). Urban soils may store carbon (Golubiewski 2006, Townsend-Small and Czimczik 2010, Raciti et al. 2011), act as a sink for atmospheric nitrogen (N) deposition (e.g., Raciti et al. 2008), and provide stormwater treatment (e.g., Zhu et al. 2004, Dietz and Clausen 2006). While soils in urban landscapes are generally thought of as low in fertility and highly disturbed (these findings are supported by research that has focused on highly compacted areas and human-constructed soils along streets; e.g., Craul and Klein 1980, Short et al. 1986), soils that are largely undisturbed or of high fertility have also been found (Pouyat et al. 2009, Raciti et al. 2011).

Research that addresses the influence of urbanization along an urban-to-rural continuum can help illuminate ecological patterns and processes (see review by McDonnell and Hahs 2008). Pouyat et al. (2008) found that the chemical composition of forest soils (including lead, copper, and calcium) varied with distance from the urban core of three major cities. Hutrya et al. (2011a) found that tree canopy cover and aboveground biomass increased with distance from the Seattle, Washington, urban core. Conversely, Ziska et al. (2005) found that emergent plant growth rates and productivity decreased with distance from the Baltimore, Maryland, urban core. Distance from urban land cover has also been shown to be a predictor of macroinvertebrate communities in streams (e.g., King et al. 2005) and species

richness of forests (e.g., Wolf and Gibbs 2004). Of course, linear distance alone cannot adequately explain ecological structure and function across the patchwork of land covers and land uses that characterize urban-to-rural transects (Medley et al. 1995, McDonnell and Hahs 2008, Hutrya et al. 2011a). Moreover, measures of urbanization (e.g., population density, ISA, patch size) can themselves vary dramatically and nonlinearly in space across urban areas (McDonnell and Hahs 2008, Pickett et al. 2011). To study heterogeneous human-natural ecosystems, ecologists must work at a diversity of sites, across a range of scales, and use more objective, indirect measures of urbanization itself to facilitate comparisons between cities (McDonnell and Hahs 2008).

If science is to inform urban development policies that minimize negative environmental impacts while maximizing ecosystem services, then a better understanding of urban ecosystems is of critical importance. Specifically, we must understand how the spatial structure of ecological, physical, and socioeconomic factors affect ecosystem function (Pickett et al. 2008). While a growing body of research has focused on differences between developed and natural areas, or urban and rural areas, human-dominated ecosystems cannot be understood in the context of simple dichotomies. The couplings between human and natural systems can have nonlinearities, thresholds, and ill-defined boundary conditions (Medley et al. 1995, Liu et al. 2006). Further, the definitions of urban, suburban, rural, and natural areas are themselves variable and ill defined. Thus, despite evidence that urban vegetation and soils can provide ecosystem services, there is conflicting evidence about the potential strength and extent of those services (Pataki et al. 2011), which may in part be caused by these inconsistent definitions of "urban" land use.

In this study we focused on urban ecosystem contributions to regional C and N stocks in the Boston Metropolitan Statistical Area (MSA). We quantified these contributions using several definitions of urban land use. We then combined our field-based measurements with remote sensing and demography to move beyond urban-rural dichotomies to assess how changes in human and environmental factors influence C and N stocks across urban-to-rural gradients. We used a transect-based, stratified random sampling design to compare our findings with traditional, distance based gradients, while also exploring the influence of indirect, distance-independent measures of urbanization. At the outset of our study we hypothesized that: (1) The urban classes we defined using land use, ISA, and population density, would be stronger predictors of C and N stocks than distance along our transect, due to the patchiness and small-scale heterogeneity that tend to characterize urban areas. (2) Aboveground biomass would be lower in urban areas compared to nonurban areas of the same land use. (3) Soil N would be higher in urban and residential areas due to local N deposition and fertilizer

application and that these inputs would influence patterns of soil C. (4) Estimates of urban C and N stocks in the study area would depend greatly on the definition of urban that is used and the scale at which it is applied.

METHODS

Study area

Boston, Massachusetts (see Plate 1, panel A), is the northernmost city of the largest megalopolis in the United States, the so-called “BosWash Corridor.” Its geographical position as an end member of this nationally important corridor makes this a key study area for an urban region that contains almost 20% of the U.S. population. Boston has a large population (4.4 million people within the MSA [12 105 km²]), substantial transportation infrastructure (e.g., toll highways, subway/trains, buses), and is a national leader in carbon emissions reduction plans and urban greening. Boston’s goal is an 80% reduction in net greenhouse gas emissions by 2050 (City of Boston, *available online*).⁵ In keeping with these sustainability initiatives, the City of Boston plans to increase tree canopy cover to 35% by 2030 by planting 100 000 trees (Grow Boston Greener, *available online*).⁶ The Boston regional landscape has a long history of human management starting with extensive agriculture (1630–1850), urban/industrial development (1800s–1950s), and currently, the region is experiencing urban renewal and has a service, technology, and financial services-based economy (1950s–present).

We established two transects across the study area (Fig. 1a). Both transects extend westward from the City of Boston, but demonstrate contrasting patterns of development. The northern transect begins in downtown Boston, continues through high-density suburbs, less dense suburbs, and then through rural areas beyond them. The southern transect, by contrast, follows a major transportation corridor from the City of Boston, through the smaller cities of Framingham and Worcester, Massachusetts. All of our analyses were performed using data within the State of Massachusetts. For data expressed on the scale of the Boston MSA, we are referring specifically to the Massachusetts portion of the Boston MSA, which includes five Massachusetts counties (Essex, Middlesex, Norfolk, Plymouth, and Suffolk) and includes >4.0 million of the 4.4 million inhabitants of the Boston MSA (Census 2010).

The Boston MSA has a temperate climate with cold winters (mean January temperature of -1.5°C in Boston) and hot summers (mean July temperature of 23.3°C in Boston; National Climatic Data Center [NCDC], *available online*).⁷ Average annual precipitation across the MSA is 105.4 cm/yr, spread relatively

evenly throughout the year (NCDC, see footnote 7). The natural vegetation in the Boston MSA is dominated by deciduous and mixed deciduous and evergreen forests, but also contains considerable areas of herbaceous and forested wetlands (Massachusetts Office of Geographic Information [MassGIS], *available online*).⁸ Forests in the study area are diverse. Dominant genera and species include oaks (*Quercus* spp.), maples (*Acer* spp.), hickories (*Carya* spp.), birches (*Betula* spp.), ashes (*Fraxinus* spp.), pine (*Pinus* spp.), American beech (*Fagus grandifolia*), and eastern hemlock (*Tsuga canadensis*; Forest Inventory and Analysis (FIA), *available online*).⁹

The surface geology has been heavily influenced by episodes of glacial erosion and deposition in the recent geologic past and includes glacial moraines, drumlins, and dissecting river valleys (NRCS 2009). The soils of the region are generally non-calcareous, acidic in pH, and glacial in origin, with much of the area characterized by glacial till and glaciofluvial soils (NRCS 2009).

Spatial data sets

Land use data was derived from the 2005 Massachusetts Land Cover data layer, a statewide, seamless digital data set created using semiautomated methods and based on 0.5-m resolution digital orthoimagery captured in April 2005 and enhanced with assessor parcel and other ancillary data (MassGIS, see footnote 8). This land cover data layer was converted from a high resolution polygon layer (0.1-ha minimum map unit) into a 30-m resolution raster layer that was aligned with the National Land Cover Dataset (NLCD; Homer et al. 2004). For grid cells that intersected more than one land use class, the land use class with the greatest total area was assigned to that cell. Once the raster data set was created, the 33 land use classes were aggregated to yield a single residential class, forest class, and “other developed” class (commercial, industrial, institutional, and developed open space). The remaining land use classes were assigned to an “unsampled” class, which included open water, wetlands, agricultural areas, and locations that would be potentially hazardous to visit (airports runways, train yards, active pit mines, and so on).

ISA was derived from the Massachusetts GIS impervious surface data layer (MassGIS, *available online*).¹⁰ This layer was created for Massachusetts GIS by Sanborn Map Company (Colorado Springs, Colorado, USA) using semiautomated techniques from 0.5-m resolution Vexcel (Boulder, Colorado, USA) UltraCam near-infrared orthoimagery that was acquired in April 2005. Impervious areas included all constructed surfaces, such as buildings, roads, asphalt, and man-made compacted soil. Non-impervious surfaces included

⁵ www.cityofboston.gov/climate/

⁶ www.growbostongreener.org

⁷ <http://lwf.ncdc.noaa.gov/oa/climate/climatedata.html>

⁸ <http://www.mass.gov/mgis/lus2005.htm>

⁹ <http://fia.fs.fed.us/>

¹⁰ www.mass.gov/mgis/impervious_surface.htm

all vegetated areas, water bodies and wetlands, and naturally occurring barren areas (i.e., sand, bare soil, rocky shores). For our analyses, the original 1-m resolution data layer was averaged across 30-m grid cells that were aligned to the NLCD data layer.

A population density map for Massachusetts was created using dasymetric population interpolation, a process by which ancillary data is used to intelligently distribute population across an area of known total population (Langford 2007). The total population of each census block was distributed across the residential area of that census block (using the aforementioned Massachusetts Land Cover data layer), instead of being assumed to be equally distributed across the census block. This population distribution method was particularly useful for rural census blocks, which covered large areas that were mostly undeveloped forest.

Urban-to-rural transects and site stratification

For this study, we defined two urban-to-rural transects that extended from downtown Boston, westward, to the Harvard Forest LTER and to Worcester, Massachusetts (the northern and southern transects, respectively; Fig. 1a). The two transects were delineated using geographic information system software (ArcGIS version 9.2; ESRI 2011). The sampling area of our transects was defined by overlaying 990×990 m grid boxes over the transect lines (Fig. 1a). These grid boxes were aligned so their edges would overlap with the 30-m NLCD to facilitate future comparisons with other geographic areas in the United States. Our overall goal was to take measurements from at least 135 plots along the transects, with an equal number of plots spread across three land use classes (forest, residential, other developed) and three urban classes (high population urban, lower population urban, and rural, as defined in the next paragraph). This would yield at least 15 plots for any given combination of land use and urban class. We used this stratified random sampling scheme to ensure that all land use classes and intensities of urban development would be adequately sampled, particularly those with relatively small land areas, such as the urban cores of Boston and Worcester, Massachusetts.

Development of a new urban classification

Individual 30-m cells were defined as “urban” if the 990×990 m neighborhood around that cell had greater than 25% ISA. Other cells were defined as nonurban. The 25% threshold for ISA was based on the steep drop in ISA when crossing the Interstate 95 (I-95) corridor around Boston (Fig. 2), and represented what one might subjectively define as the most obviously “urban” portion of the Boston MSA. To ensure that we adequately sampled high population density urban areas, which represent a small total land area, we divided our urban land cover category into high and lower population density classes (greater or less than 2500 persons/km², respectively). This yielded three

classes: high population urban, lower population urban, and nonurban. The analysis was repeated at 270-m and 90-m grid sizes, so we could study the influence of scale on our definition of urban. The calculations to determine urban class were performed in ArcGIS using a moving window function to calculate neighborhood statistics around each 30-m cell.

Comparison of urban definitions and spatial scales

We extrapolated plot-level data (biomass and soil C and N; details in the following sections of the *Methods: Aboveground live biomass, Coarse woody debris, Soils*) to the Boston MSA and the state of Massachusetts using five different urban extents to test the sensitivity of our results to the definition of urban. We used urban extents from the U.S. Census Bureau, the Global Rural–Urban Mapping Project (GRUMP, *available online*),¹¹ and our own three-class definition of urban (as described in *Methods: Development of a new urban classification*) applied at three neighborhood scales (90 m, 270 m, and 990 m). The U.S. Census Bureau delineates urban areas using population density, with a minimum threshold of 2590 people/km² (1000 people/mile²) for the core of urban areas and a threshold of 1295 people/km² (500 people/mile²) in surrounding census blocks (U.S. Census Bureau, see footnote 5). By contrast, GRUMP delineates urban areas starting with global nighttime lights data, which is converted to polygons that represent urban areas, to which urban areas delineated in the Digital Chart of the World are then added (GRUMP, *available online*, see footnote 11). We systematically reclassified each of our 139 research plots (as urban and nonurban) using these five different urban extents and calculated mean and total aboveground biomass, soil C, and soil N for each. This definition analysis resulted in five estimates of urban C and N stocks on a per-unit-area basis and on an MSA-wide basis.

Study plots

With the goal of sampling 135 plots across both public and private lands, we started by selecting 45 random points in each of our nine sampling classes (three urban classes by three land use classes; Table 1). Digital parcel data were used to get address and ownership information for most properties. For properties where this information was unavailable, we had to visit the plots in person to collect initial address information. To gain permission to visit potential study plots, we mailed letters with self-addressed reply cards to each of the property owners. If accessing a plot would have required permission from three or more land owners, that plot was removed from consideration due to the low probability of receiving permission from so many different property owners. In general, properties that were picked first by our random point generator were

¹¹ www.ifpri.org/dataset/global-rural-urban-mapping-project-grump

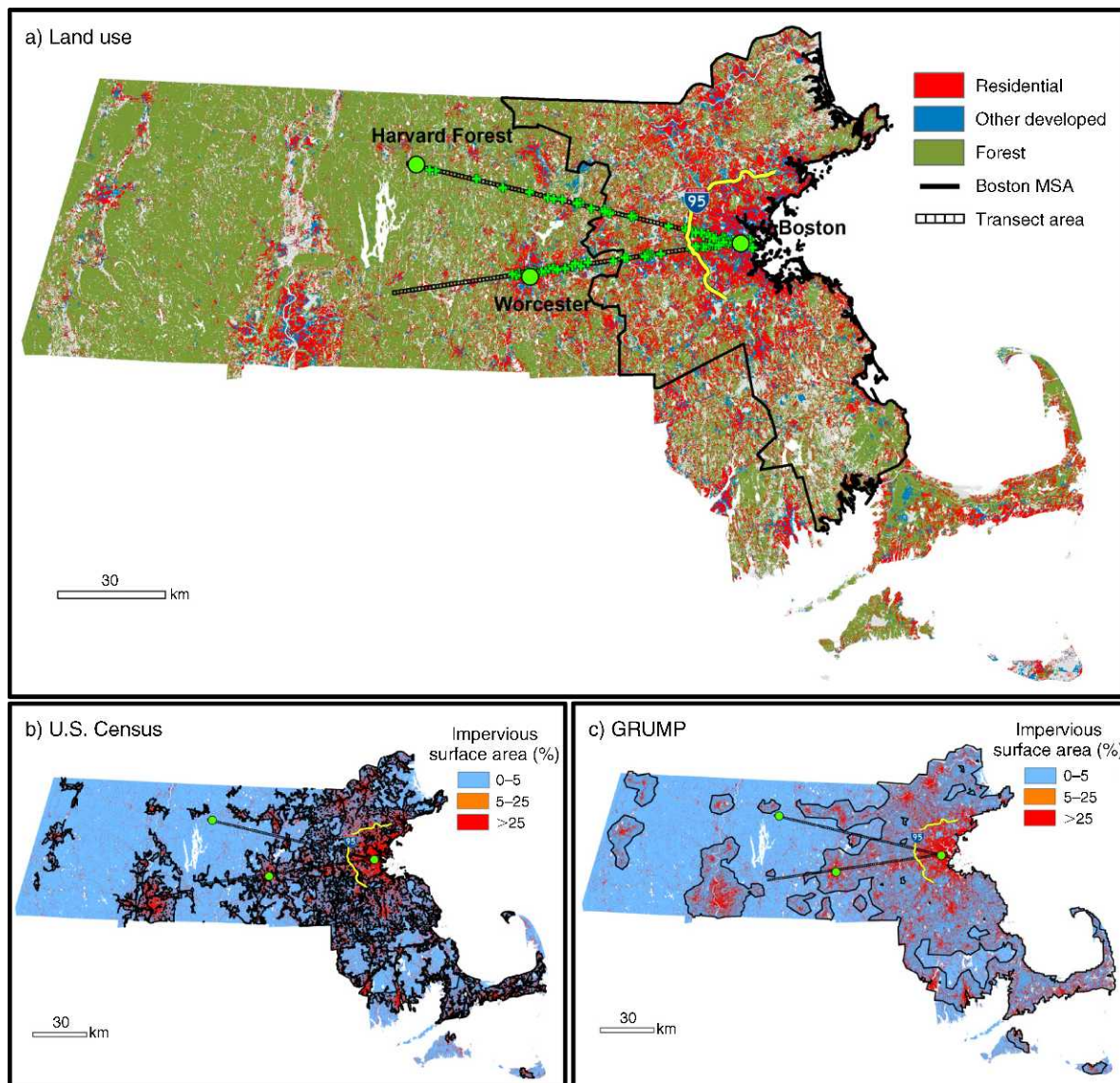


FIG. 1. (a) Map showing land use in the study area, the location of the northern and southern transect lines, the boundary for the Boston Metropolitan Statistical Area (MSA), Massachusetts, USA, and individual sampling locations (small green “+” symbols). The study area was divided into a single residential class, a forest class, and an “other developed” class (commercial, industrial, institutional, and developed open space). Lower maps show the (b) U.S. Census Bureau and (c) Global Rural–Urban Mapping Project (GRUMP) urban extents superimposed over impervious surface area (ISA). The U.S. Census delineates urban areas using the population density of census blocks (*available online*, see footnote 4). By contrast, GRUMP delineates urban areas using global nighttime lights data and urban areas from the Digital Chart of the World (*available online*, see footnote 11).

chosen over properties picked later; however, to prevent a potential bias towards public lands, for which permissions were easier to obtain, we chose to sample public and private plots in the same proportion as found among the first 15 randomly chosen plots in each category.

Field sampling occurred between June and August 2010. At the end of the field season we had surveyed 139 plots distributed relatively equally across the land use and urban classes (Fig. 1a). Sample plots were fixed radius, circular with a 15 m radius (707 m²). We used a

Garmin Dakota GPS with Wide Area Augmentation System (WAAS) averaging enabled to locate plot centers (Garmin International, Olathe, Kansas, USA). Typical GPS errors were 3 m, for relatively open plots, and up to 10 m for closed-canopy forest locations. We attempted to match the center of the land cover pixels (30-m resolution) with on-the-ground plot centers (30-m plot diameter), but errors in the GPS locations meant that exact co-location was not possible. Depending on the local topographic and vegetation circumstances, we used a combination of a TruPulse 200 Professional

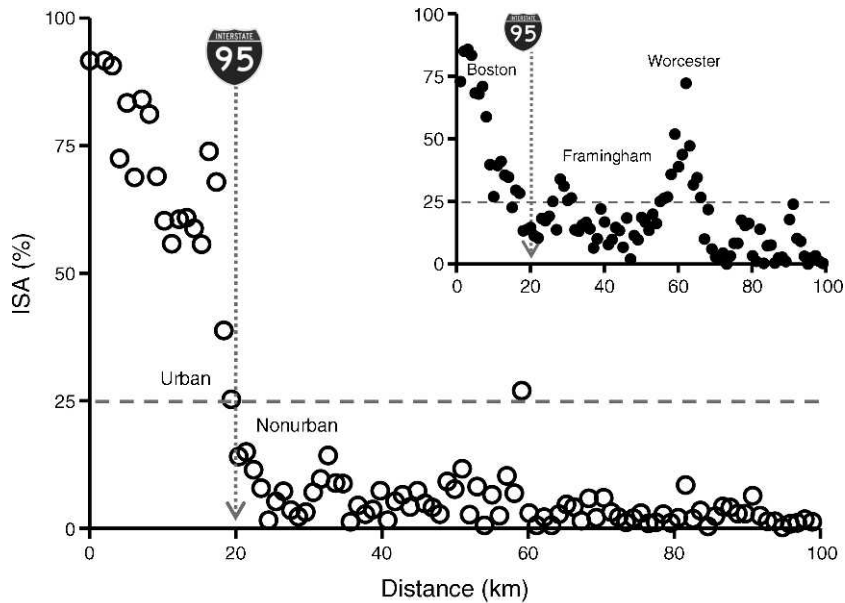


FIG. 2. Percentage of impervious surface area (ISA) as a function of distance from downtown Boston for the northern and southern (inset) transects of the study area. Each point represents the mean impervious surface area (ISA) across a 990×990 m grid box along the transect. The vertical dotted line indicates the location of Interstate 95 (I-95), a major highway in the Boston Metropolitan Statistical Area (see Fig. 1a). The horizontal dashed line indicates the threshold of ISA (25%) that was used to delineate urban areas in this study. Individual 30-m cells were defined as "urban" if the 990×990 m neighborhood around that cell had greater than 25% ISA. Other cells were defined as nonurban.

Range Finder and Hypsometer (Laser Technology, Centennial, Colorado, USA) and meter tapes to determine exact plot boundaries. All slope distances were corrected to horizontal distance. For each plot, on-site estimates of ground cover were made, including percent impervious surface, lawn, garden, weedy (unmanaged fine vegetation), degraded forest, and forest.

Aboveground live biomass

All live trees larger than 5 cm in diameter at breast height (dbh) were surveyed (see Plate 1, panel B). The dbh was measured at 1.37 m unless slope or tree form abnormalities required adjustments; measurements followed the protocols outlined in Fahey and Knapp (2007). Tree diameters were measured with dbh tapes to the nearest 0.1 cm. Where possible, trees were identified to species or genus (if species could not be determined), but due to the large number of exotic species present within urban areas, 2% of stems were identified as miscellaneous hardwood species.

Biomass of live trees was estimated using published allometric equations relating plant diameter to dry mass. Species-specific equations were used where possible including *Acer rubrum* (Crow 1983), *Acer saccharinum* (McHale et al. 2009), *Acer saccharum* (Brenneman et al. 1978), *Ailanthus* (T. G. Siccama and K. A. Vogt, unpublished data), *Betula alleghaniensis* (Freedman et al. 1982), *Betula lenta* (Martin et al. 1998), *Betula populifolia* (Freedman et al. 1982), *Carya* spp. (Martin et al. 1998), *Fagus grandifolia* (Siccama et al. 1994),

Fraxinus americana (Brenneman et al. 1978), *Hamamelis virginiana* (Telfer 1969), *Picea glauca* (Ker 1980), *Pinus resinosa* (Ker 1980), *Pinus strobus* (Pastor et al. 1984), *Prunus serotina* (Brenneman et al. 1978), *Quercus alba* (Bridge 1979), *Quercus rubra* (Brenneman et al. 1978), *Quercus velutina* (Ter-Mikaelian and Korzukhin 1997), and *Tsuga canadensis* (Young et al. 1980). The most specific equation possible was applied in all cases; where species or genus level equations were unavailable, we applied the Jenkins et al. (2003) miscellaneous hardwood or softwood equations. One half of live plant biomass was assumed to be carbon, and biomass is reported in units of dry mass carbon, kg C/m^2 . Note that recent

TABLE 1. Initial stratification scheme for the 139 study plots in the Boston Metropolitan Statistical Area (MSA), Massachusetts, USA.

Urban class	Number of plots		
	Forest	Residential	Other developed
Urban, high population density	15	16	16
Urban, lower population density	14	15	18
Nonurban	15	15	15

Notes: The three urban classes are based on population and impervious surface area statistics for a 990×990 m neighborhood around each plot. If the neighborhood impervious surface area (ISA) was $>25\%$, the plot was classified as urban. Among urban plots, those with a neighborhood population density $>2500/\text{km}^2$ were classified as high population density. The "other developed" class includes commercial, industrial, institutional, and developed open space.

results from McHale et al. (2009) suggest problems with the application of forest-derived allometries to urban trees; however, a more complete allometric analysis was not possible within the scope of this study due to the lack of availability of urban tree allometries for the regional species assemblage. The use of forest-derived allometric equations also afforded the analysis a methodological consistency across the urban-to-rural gradient, which included a range from heavily urbanized street trees to rural forest trees. It is unclear if the use of urban-specific allometric equations would have increased or decreased the estimated carbon stocks in the most urban plots; this is an area that requires additional research.

Coarse woody debris

Downed and standing coarse woody debris (CWD) with a diameter greater than 10 cm and a minimum length of 1 m were surveyed within the full sample plot areas (see Plate 1, panel C). Logs were identified as hardwood or softwood (where possible) and assigned decay class values, following the conventions defined by Harmon and Sexton (1996) and Barford et al. (2001). Dimensional measurements were converted to volumes, using Newton's formula for a cylinder (Harmon and Sexton 1996). CWD wood density values from Liu et al. (2006) were applied to the calculated volumes to estimate biomass. One half of the CWD biomass was assumed to be carbon. All CWD biomass reported here is in units of dry mass carbon, kg C/m².

For recently dead and standing trees, where branches and twigs were still present, we estimated biomass by reducing the allometrically estimated biomass for a live tree by one-third to account for biomass losses associated with the recent mortality (Liu et al. 2006). For more decomposed standing dead trees (no branches), volume was estimated by measuring tree height, base diameter, and decay class. Top diameter was measured (where possible) or visually estimated. Biomass for standing logs was estimated using the same five-class decay method.

Soils

Two soil cores were collected from each plot for which there was exposed (e.g., non-impervious) ground (see Plate 1, panel D). Soil cores were taken from random locations within the field plot. For plots that were largely impervious, we chose locations that were representative of the majority of non-impervious cover. For instance, if lawn was the dominant non-impervious cover at a plot, we took soil cores from lawn areas. If there was a relatively even mix of lawn and garden, we took a soil core from each area. For urban areas where the only non-impervious cover was inside small tree pits, we did not collect soil samples to avoid damaging street trees. Soil cores were collected to a depth of 10 cm using a 5 cm diameter slide-hammer corer (AMS Equipment, American Falls, Idaho, USA) and taken back to the laboratory where they were refrigerated

until they could be processed to determine bulk density, soil moisture, and total C and N content.

Intact soil cores were weighed and then sieved to remove rocks, coarse roots, and organic material >2 mm in size. Rocks and large organic material were weighed and set aside. Approximately 50 g of the remaining, homogenized soil was dried (60°C for 48 h or until no further change in mass was detectable) and then ground into a fine powder with a mortar and pestle. For each soil, a 20-mg subsample of this well-mixed, finely ground material was loaded into a 9 × 5 mm tin capsule, placed in a sealed microtiter plate and stored until it could be analyzed for total C and N content by flash-combustion/oxidation using a Thermo Finnigan Flash EA 1112 elemental analyzer (0.06% C and 0.01% N detection limits; Thermo Fisher Scientific, Waltham, Massachusetts, USA). Note that this method may overestimate organic carbon for soils that contain significant inorganic carbon pools (e.g., those with carbonate-rich parent material). The soils in our study area are generally acidic, so total carbon should be a fairly accurate reflection of organic carbon. For all data, the density of C per unit area (1 m²) was calculated as $C = C_f B_D (1 - \delta_{2mm}) V$, where C is carbon density, δ_{2mm} is the fraction of material larger than 2 mm diameter, B_D is bulk density, C_f is the fraction by mass of organic C, and V is the volume of the soil core (Post et al. 1982).

Logistical constraints did not allow us to collect deep soil cores from our study plots. To estimate total soil C and N stocks, we extrapolated our findings from 0–10 cm to a depth of 1 m using methods from Pouyat et al. (2008) and deeper soils data from forests in Massachusetts (P. C. Lemos and A. C. Finzi, *unpublished data*) and developed sites in Baltimore (Raciti et al. 2011). Raciti et al. (2011) found that the top 10 cm of their developed soils contained 32.0% ± 2.4% and 30.8% ± 2.3% (mean ± SE) of total soil C and N, respectively, down to one meter. Lemos and Finzi (*unpublished data*) found similar proportions of total soil C and N in the upper 10 cm of forested soils (31.2% ± 3.4% and 26.2% ± 3.5%, respectively). Based on these data, we assumed that our 10-cm measurements represented 30% of 1 m deep soil C and N pools and used that proportion to estimate soil C and N to a depth of 1 m. We note that this method of extrapolation will not account for the presence of buried A horizons.

Unsampled land use classes

Approximately 81% of the State and 78% of the Massachusetts portion of the Boston MSA (Table 2) is covered by our sampled land use classes. Most of the unsampled area was wooded wetlands, herbaceous wetlands, and agricultural areas. We used a mix of literature and field measurements to estimate the vegetation C, soil C, and soil N stocks of these unsampled land use/land cover classes. Wooded wetlands, which cover more than 1000 km² of Massachusetts, were the only land use class that contained

TABLE 2. Distribution of land use classes and their absolute (km²) and relative (%) areas within the Boston MSA.

Land use	Area (km ²)	Area (%)
Forest	2599	41.7
Residential	1596	25.6
Other developed	637	10.2
Unsampled	1398	22.4
Forested wetlands	532	8.5
Herbaceous wetlands	269	4.3
Agriculture	187	3.0
Miscellaneous	411	6.6
Total	6231	100.0

potentially significant stocks of aboveground tree biomass. The other unsampled land use/land cover classes contain little tree canopy. For scaling purposes, we assumed that aboveground tree biomass in wooded wetlands was similar to forests in the region (see Bridgman et al. 2006) and therefore assigned the mean (non-area-weighted) biomass for forests in our study (10.4 ± 5.4 kg C/m²). For wetland soils, we used an estimate of 16.2 kg C/m² (Bridgman et al. 2006) and assumed a C:N ratio of 21.7:1 (mean of Bedford et al. 1999; Appendix B) to estimate soil N concentration. Note that the 16.2 kg C/m² (for 1 m depth) is a conservative estimate, as peatlands can contain considerably greater carbon stores (Bridgman et al. 2006). For agricultural soil C, we used a value of 6.0 kg C/m² (Birdsey 1992). For the small remaining area (airports, train yards, mining areas, junkyards, and so on), we assumed that biomass and soil C and N concentrations were the same as our “other developed” land use category that included commercial, industrial, and developed open space areas.

Comparison to U.S. Forest Service reported urban carbon estimates for Massachusetts

To put our results into perspective, we compared our urban tree biomass estimates to U.S. Forest Service estimates of urban tree biomass for Massachusetts (Nowak and Greenfield 2008). The U.S. Forest Service analysis starts with NLCD total tree canopy area within the U.S. Census urban extent and then multiplies that canopy area by 9.1 kg C/m²-of-tree-canopy. This value (9.1 kg C/m²) is the mean per-canopy-area biomass for 17 cities and towns in the United States, from Boston, in the Northeast (9.1 kg C/m²), to Atlanta, Georgia, in the Southeast (9.7 kg C/m²), to Minneapolis, Minnesota, in the Midwest (5.7 kg C/m²), to San Francisco and Oakland, California (12.3 and 5.2 kg C/m², respectively), on the West coast.

Accounting for soils under impervious surfaces

We were unable to sample soils from underneath impervious surfaces, which covered a significant proportion of many residential and other developed plots; however, we used a worst-case scenario to evaluate the

potential impact of impervious areas on soil C and N stocks. We assumed a carbon density value of 3.38 ± 0.99 kg C/m² (upper 1 m) for soils under impervious surfaces, which is the mean of values reported by Pouyat et al. (2002) for clean fill. We then assigned this C density to the impervious portion of each field plot (e.g., if the area of the plot was 25% impervious, one-quarter of the plot was estimated to have a C density of 3.38 kg C/m²). To estimate soil N for the impervious portion of the plot, we assumed that the C:N ratio of the impervious area soil would be the same as the uncovered area soil. We made this assumption because no data were available that describe the N composition of impervious-covered soils and Pouyat et al. (2002) do not report the N content of clean fill. Some of these impervious covered soils are likely to be richer in C and N than clean fill, but it was our intent to assess the maximum likely influence of impervious surfaces on soil C and N stocks.

Statistics

All error values shown in the text are standard error of the mean for plot-level C and N stocks. Where necessary, the data were transformed to meet assumptions of normality. Error bars in bar graphs represent 95% confidence intervals. Comparisons between two means were performed using a Student's *t* test. Comparison of means among multiple treatments were performed using ANOVA and Tukey's honestly significant difference post hoc test. The Massachusetts-wide and MSA-wide estimates of C and N stocks were generated by multiplying the mean C and N stocks for a given cover class (e.g., urban forest) by the land area in that cover class. The standard error reported for these Massachusetts- and MSA-level C and N stocks were calculated by multiplying the per-area standard error by the land area of the cover class. We did not attempt to propagate map and classification errors from the data layers used in our spatial analyses, so the uncertainty of biomass and soil C and N estimates may be larger than reflected in the standard error values reported here. All statistics were performed using a combination of SAS JMP 9.01 (SAS Institute 2009) and R (R Development Core Team 2011) software.

RESULTS

Note that, except where explicitly stated, all vegetation and soil analyses are based on the urban definition that we developed (>25% ISA over 990 m²) and our original site stratification scheme (Table 3).

Vegetation and soil C and N stocks for the Boston MSA

Aboveground biomass (trees ≥ 5 cm dbh) for the Massachusetts portion of the Boston MSA was 44.9 ± 2.4 Tg C, or 7.2 ± 0.4 kg C/m², (mean ± SE) using our original urban definition and stratification scheme. These estimates were based on a land area of 6231 km² that included 41.7% forest, 25.6% residential, and

TABLE 3. Aboveground biomass and soil C and N stocks (means with SE in parentheses) for the Boston MSA using several urban definitions and after adjusting for soils under impervious surface areas (990-m ISA-adjusted, only).

Measurement and scale (grid size)	Urban	Nonurban	Total (includes unsampled)
Biomass C (kg/m²)			
990 m	3.9 (0.4)	9.3 (0.5)	7.2 (0.4)
990 m, ISA-adjusted	3.9 (0.4)	9.3 (0.5)	7.2 (0.4)
270 m	3.7 (0.7)	8.2 (0.7)	6.4 (0.5)
90 m	3.5 (0.7)	9.7 (1.0)	7.1 (0.7)
U.S. Census	6.6 (0.8)	8.8 (1.2)	6.4 (0.7)
GRUMP	6.6 (0.8)	9.2 (1.7)	6.4 (0.7)
Soil C (kg/m²)			
990 m	13.0 (0.7)	12.9 (0.3)	13.6 (0.3)
990 m, ISA-adjusted	8.3 (0.6)	11.5 (0.3)	11.9 (0.3)
270 m	12.5 (1.1)	13.3 (1.6)	13.7 (1.2)
90 m	12.5 (1.1)	14.6 (1.1)	14.4 (0.9)
U.S. Census	13.3 (0.9)	13.4 (1.8)	13.9 (0.8)
GRUMP	13.2 (0.8)	12.3 (2.6)	13.6 (0.8)
Soil N (kg/m²)			
990 m	0.79 (0.07)	0.68 (0.06)	0.76 (0.05)
990 m, ISA-adjusted	0.50 (0.05)	0.61 (0.05)	0.67 (0.04)
270 m	0.76 (0.05)	0.66 (0.07)	0.74 (0.05)
90 m	0.77 (0.05)	0.77 (0.06)	0.81 (0.04)
U.S. Census	0.77 (0.04)	0.67 (0.07)	0.80 (0.04)
GRUMP	0.76 (0.04)	0.72 (0.09)	0.79 (0.04)
Area (km²)			
990 m	1103	3729	6231
990 m, ISA-adjusted	1103	3729	6231
270 m	1246	3586	6231
90 m	1418	3414	6231
U.S. Census	4020	812	6231
GRUMP	4132	701	6231

Note: Soil C and N stocks are extrapolated to a depth of 1 m (see *Methods: Soils*). The U.S. Census delineates urban areas using the population density of census blocks (*available online*, see footnote 4). The Global Rural–Urban Mapping Project (GRUMP) delineates urban areas using global nighttime lights data and urban areas from the Digital Chart of the World (*available online*, see footnote 11). The urban definitions developed as part of this study (990 m, 270 m, and 90 m) are based on the impervious surface area (ISA). Individual 30-m cells were defined as “urban” if the 990 × 990, 270 × 270, or 90 × 90 m neighborhood around that cell had greater than 25% ISA. The 990-m ISA-adjusted urban definition is the same as the 990-m definition, but soil carbon (C) and nitrogen (N) stocks were adjusted to reflect the C and N losses that are likely to have occurred beneath impervious surfaces (see *Methods: Accounting for soils under impervious surface*).

10.2% other developed areas (Table 2). Forests held the largest proportion of aboveground biomass (11.6 ± 0.5 kg C/m²), but trees in residential (4.6 ± 0.5 kg C/m²) and other developed areas (2.0 ± 0.4 kg C/m²) still contained significant C stocks. Forested wetlands covered 8.5% of the land area of the MSA and contained an estimated 6.2 Tg C. Overall, developed areas contained 18.5% of the total aboveground C stocks in the MSA and covered 35.8% of the land area.

Urban areas showed consistently lower aboveground biomass than nonurban areas of the same land use, though these differences were only statistically significant for the other developed land use category ($P=0.01$; Fig. 3a). Urban forest, residential, and other developed land uses contained 10.3 ± 0.6 , 3.4 ± 0.4 , and 1.3 ± 0.3 kg C/m², respectively, compared to 11.7 ± 0.5 , 5.2 ± 0.6 , and 2.8 ± 0.5 kg C/m², for nonurban areas of the same land use type, respectively (Table 4).

Coarse woody debris (standing or fallen dead trees) contributed an estimated 2.0 ± 0.8 Tg C or 0.42 ± 0.15 kg C/m² to C stocks in the Boston MSA. Most of the CWD was concentrated in nonurban forests (0.78 ± 0.27 kg C/m²). Urban forests (0.46 ± 0.17 kg C/m²), urban residential (0.04 ± 0.06 kg C/m²), nonurban residential (0.03 ± 0.03 kg C/m²), and urban and nonurban other land uses (0.0002 ± 0.0002 kg C/m²) contained negligible CWD stocks.

Soil C stocks (extrapolated to 1 m depth) for the Massachusetts portion of the Boston MSA were 84.5 ± 2.1 Tg C using our original stratification scheme (Tables 1 and 3), which equates to an area-weighted mean of 13.6 ± 0.3 kg C/m² (Table 4). Forest soils held 40.9% of total soil C, followed by residential soils with 25.0%, and other developed areas with 8.2%. The remaining 25.9% of soil C was held in unsampled land use classes.

Soil N stocks (extrapolated to 1 m depth) for the Boston MSA were 4.7 ± 0.3 Tg C using our original

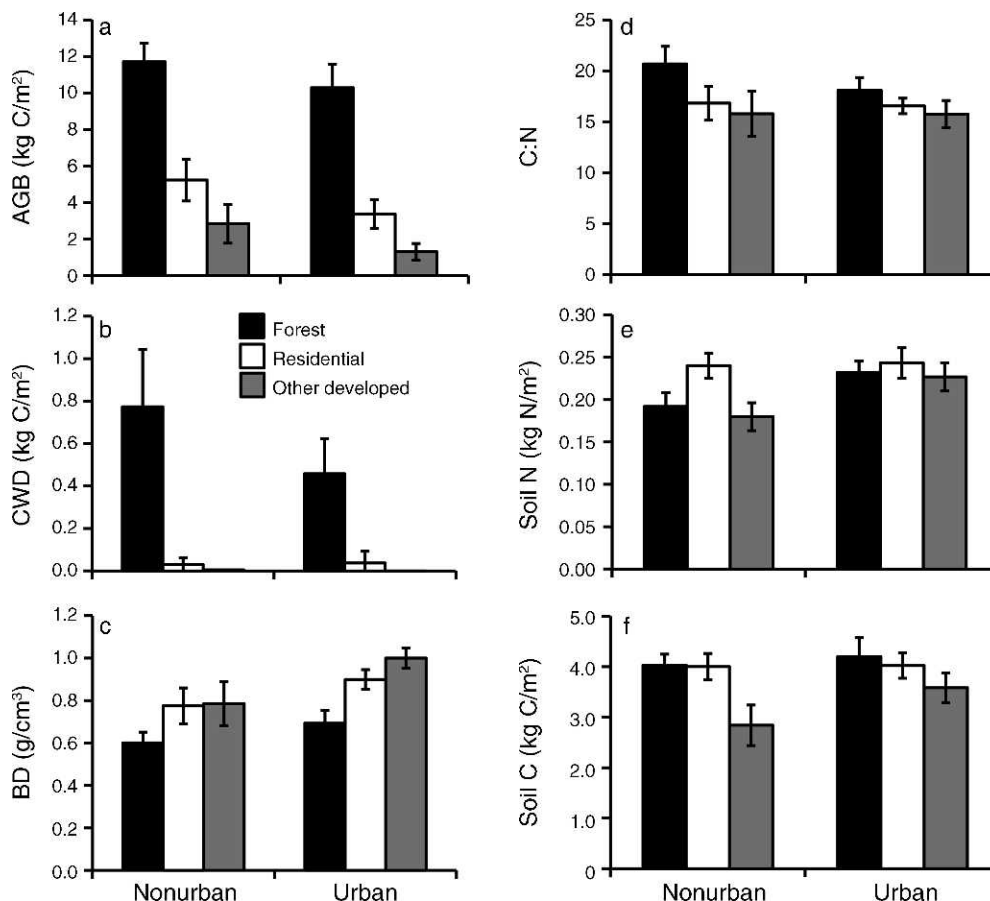


FIG. 3. (a) Mean aboveground biomass (AGB), (b) coarse woody debris (CWD), (c) soil bulk density (BD), (d) soil carbon to nitrogen ratio, and (e) soil nitrogen and (f) carbon concentrations for 10 cm deep soil cores from urban and nonurban areas (using our 990-m definition) across three land use types. Error bars are 95% confidence intervals.

stratification scheme (Table 1). Forest soils contained 35.7%, residential soils 27.2%, and other developed soils 9.3% of the total. We calculated that unsampled land use classes held 27.8% of total soil N.

Soil C, N, and bulk density (0–10 cm)

We found that the greatest concentrations of soil C (0–10 cm, original 990-m urban definition) were held in forest (4.2 ± 0.2 and 4.0 ± 0.1 kg C/m² for urban and

TABLE 4. Aboveground biomass (AGB) and soil C and N stocks (means with SE in parentheses) by land use and urban class (using our 990-m urban definition) for the Massachusetts portion of the Boston MSA.

Land use	Area (km ²)	AGB (kg C/m ²)	Soil C (kg C/m ²)	Soil N (kg N/m ²)
Urban, high population				
Forest	11	9.2 (0.5)	4.4 (0.2)	0.24 (0.02)
Residential	198	1.9 (0.4)	4.3 (0.2)	0.23 (0.01)
Other developed	88	0.6 (0.2)	4.0 (0.2)	0.26 (0.02)
Urban, lower population				
Forest	172	10.4 (0.6)	4.2 (0.2)	0.23 (0.02)
Residential	365	4.2 (0.4)	3.9 (0.2)	0.25 (0.01)
Other developed	270	1.5 (0.3)	3.4 (0.3)	0.22 (0.02)
Nonurban				
Forest	2416	11.7 (0.5)	4.0 (0.1)	0.19 (0.02)
Residential	1033	5.2 (0.6)	3.9 (0.1)	0.24 (0.02)
Other developed	280	2.8 (0.5)	2.8 (0.2)	0.18 (0.01)

Note: Soil C and N stocks are measured values for 0–10 cm depth.

nonurban, respectively), followed by residential (4.0 ± 0.2 and 4.0 ± 0.1 kg C/m²), and then other developed soils (3.6 ± 0.2 and 2.8 ± 0.2 kg C/m²; Fig. 3f). While the pattern of decreasing soil C from forest to residential to other developed land uses was consistent for urban and nonurban areas, the differences were not statistically significant. The exception was for other developed land uses, where soil C concentrations (3.6 ± 0.3 and 2.8 ± 0.2 kg C/m², urban and nonurban, respectively) were significantly lower than residential (4.0 ± 0.2 and 3.9 ± 0.1 kg C/m²; $P=0.3$ and $P=0.4$, respectively) and forest (4.2 ± 0.2 and 4.0 ± 0.1 kg C/m²; $P=0.3$ and $P < 0.1$, respectively) land uses within a given urban class. In the other developed land use class, soil C concentrations were higher in urban than nonurban areas ($P=0.03$).

Soil N concentrations (0–10 cm, original 990-m urban definition) in forested and other developed land uses (0.23 ± 0.01 and 0.23 ± 0.01 , respectively) tended to be higher in urban areas than nonurban areas of the same land use (0.19 ± 0.01 and 0.18 ± 0.01 ; $P=0.03$ and $P=0.04$, respectively; Fig. 3e). There was no difference in soil N between urban and nonurban residential land uses (0.24 ± 0.01 kg N/m² and 0.24 ± 0.01 kg N/m², respectively, $P=0.39$).

Soil C and N concentrations (0–10 cm, original 990-m urban definition) were correlated for all land use classes ($r^2=0.37$, $P < 0.001$) and within each land use class ($P < 0.01$ for all). The ratio of C to N concentration (C:N) was similar for urban and nonurban residential and other urban land uses (between 15.8 ± 0.7 and 16.8 ± 0.9). Forests tended to have higher C:N (18.1 ± 0.6 and 20.7 ± 0.9 for urban and nonurban, respectively) than other land uses, but this difference was only statistically significant for nonurban forests ($P < 0.05$ for all comparisons; Fig. 3d).

Soil bulk density (0–10 cm, original 990-m urban definition) was higher in residential (0.90 ± 0.05 and 0.78 ± 0.09 g/cm³ for urban and nonurban, respectively) and other developed land uses (1.00 ± 0.05 and 0.79 ± 0.13 g/cm³) than in forests (0.69 ± 0.06 and 0.60 ± 0.05 g/cm³; $P < 0.05$ for all; Fig. 3c). Within land use classes, bulk density was significantly higher in urban than nonurban areas for residential and other developed ($P=0.03$ and $P=0.04$, respectively) and nearly significantly higher for forest land use ($P=0.051$).

Accounting for impervious surface area in soil C and N estimates

To estimate the maximum potential impact of covered soils on C and N stocks, we assigned the impervious portion of each field plot a C and N density equivalent to clean fill soils (Pouyat et al. 2002). ISA in urban forest (6.4%), residential (40.6%), and other developed (58.1%) field plots was greater than in nonurban counterparts (4.0%, 31.3%, 28.9%, respectively). After accounting for impervious areas, total C stocks decreased by 36% in urban areas (from 13.0 ± 0.7 to 8.3 ± 0.6 kg C/m²), 11% in nonurban areas (from 12.9 ± 0.3 to 11.5 ± 0.3 kg C/

m²), and 12% across the MSA (from 13.6 ± 0.3 to 11.9 ± 0.3 kg C/m²; Table 3).

Trends with distance, impervious surface area, and population

Along the northern transect of our study, we observed a steep drop in ISA around our field plots (990×990 m neighborhood) when crossing from within the I-95 corridor outward from Boston (Fig. 2). The northern transect starts in downtown Boston, travels through high population density suburbs, and then through less dense suburban and rural areas outside of I-95 (Fig. 1). The southern transect exhibits a more complex pattern of development as it travels alongside two major highways (Interstate 90 and Route 9) and passes through two secondary urban centers (Framingham and Worcester, Massachusetts). Along this southern transect, ISA decreases just outside the I-95 corridor, increases as it passes through Framingham, decreases between Framingham and Worcester, then increases once again as it approaches the city of Worcester (Fig. 2, inset). Neighborhood-level ISA (990×990 m around each plot) was strongly and positively correlated with neighborhood population density (Fig. 4a). Population density appears to rise exponentially with ISA for both the northern and southern transects in our study area ($r^2=0.84$, $r^2=0.69$, respectively).

Despite the strong relationship between distance and ISA, among field plots, we did not observe any relationship between distance from the city center and measures of aboveground biomass, soil C, and soil N. This was true when the data were aggregated (i.e., all land uses) and when the data were analyzed by individual land use (forest, residential, and other developed).

We extrapolated our plot-level observations to the neighborhood scale (990×990 m), by multiplying each urban-by-cover class (see Table 4 for means and SE) by its mean C or N stock, and found a strong relationship ($r^2=0.64$, $P < 0.001$) between aboveground biomass and distance along the northern transect (Fig. 5a). For the southern transect, where the intensity of development is higher and varies nonlinearly with distance from Boston, this relationship was weaker, but still significant ($r^2=0.40$, $P < 0.01$; Fig. 5b). Neighborhood-scale relationships between distance and soil N concentrations were strong for the northern and southern transects, with soil N concentrations decreasing with distance from the downtown Boston ($r^2=0.62$ and $r^2=0.41$, respectively; Fig. 5c). Neighborhood scale soil C concentrations showed weak, but still statistically significant relationships with distance along the northern and southern transects ($r^2=0.20$, $P < 0.001$ and $r^2=0.16$, $P < 0.01$, respectively).

Neighborhood level (990×990 m) ISA was strongly and negatively correlated with neighborhood level biomass for the northern and southern transects ($r^2=0.81$, $P < 0.001$; Fig. 4b). Neighborhood level

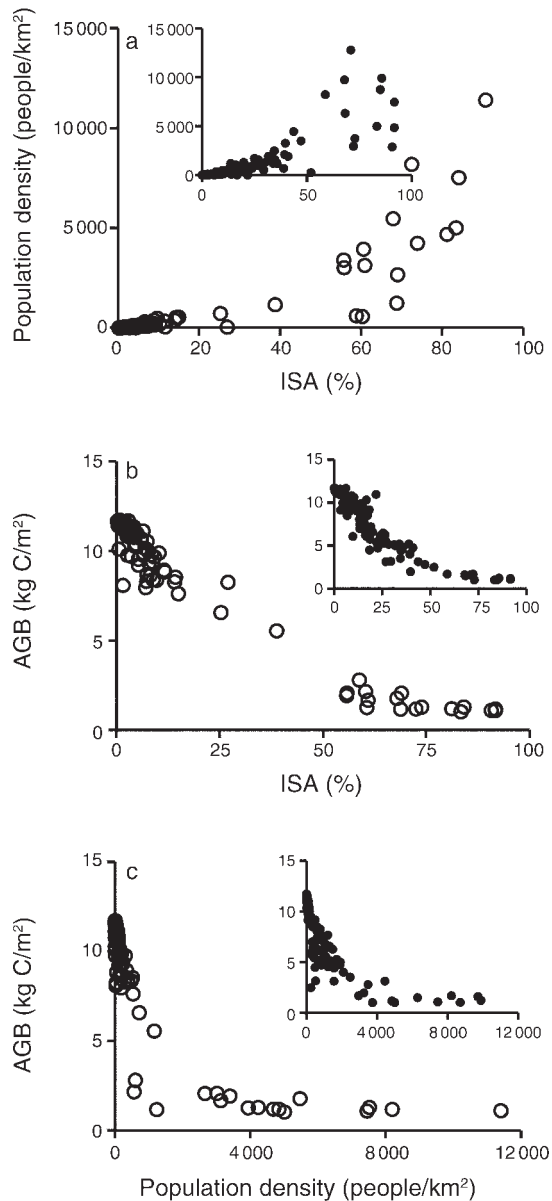


FIG. 4. (a) Population density vs. impervious surface area (ISA), (b) aboveground biomass (AGB) as a function of ISA, and (c) AGB as a function of population density. All data are at the neighborhood level (990×990 m around each plot). The northern (main graphs) and southern (inset graphs) transects are shown, with the same axes.

population density was also a strong predictor of neighborhood level biomass ($r^2 = 0.68$, $P < 0.001$; Fig. 4c). While ISA and population density are strongly correlated for the northern and southern transects (Fig. 4a), ISA appears to be a better predictor of neighborhood level aboveground biomass ($r^2 = 0.90$ vs. 0.55 , respectively), soil C ($r^2 = 0.36$ vs. 0.21), and soil N ($r^2 = 0.71$ vs. 0.48).

Comparison to U.S. Forest Service estimates of urban biomass

For comparison with U.S. Forest Service reported estimates of urban tree biomass (Nowak and Greenfield 2008), we extrapolated our aboveground biomass estimates to the areas that the U.S. Census defines as urban within the state of Massachusetts. This resulted in an urban tree biomass estimate of 42.8 ± 6.8 Tg C (6.9 ± 0.9 kg C/m²), which is much higher than the U.S.

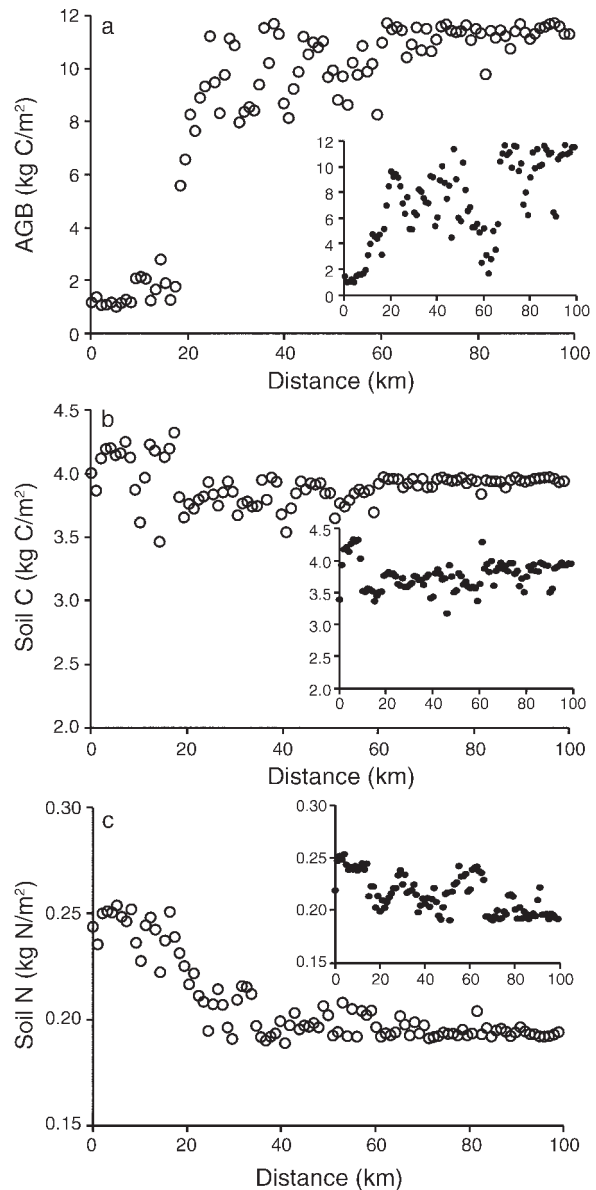


FIG. 5. (a) Aboveground biomass (AGB), (b) soil carbon, and (c) soil nitrogen extrapolated to the neighborhood scale (990×990 m) as a function of distance from downtown Boston. Extrapolations were performed by applying mean measured values to each of nine urban \times land use classes (Table 4). Soil data are to 10 cm depth. The northern (main graphs) and southern (inset graphs) transects are shown, with the same axes.

Forest Service estimate of 28.7 Tg C (4.0 kg C/m²) for the same area. The U.S. Forest Service applied a constant biomass of 9.1 kg C/m²-of-tree-canopy to land that the NLCD predicts as having tree canopy within the census-defined urban area. The large difference between our estimate and the U.S. Forest Service estimate may be caused by (1) systematic underestimation of tree canopy by the NLCD (Greenfield et al. 2009, Nowak and Greenfield 2010, Smith et al. 2010), and (2) application of biomass values derived from major cities to the census-defined urban area, which is more sparsely populated (on average) and contains large areas of forest (42% of the census-defined urban area). To test this hypothesis, we adjusted the NLCD canopy area based on a recent sensitivity analysis by the same authors (Nowak and Greenfield 2010). Their analysis demonstrated that the NLCD underestimated canopy by 3.4% for forested areas, 31.5% for developed areas, 98.2% for agricultural areas, and 57.7% for other land uses within the NLCD mapping zone that covers Massachusetts (zone 65). The application of Nowak and Greenfield's (2010) adjustments resulted in a canopy area estimate of 4017 km² compared to 3149 km² prior to the adjustments. We multiplied the adjusted canopy area by 10.6 ± 1.2 kg C/m²-of-tree-canopy, which was the mean biomass per area of tree canopy for the census-defined urban area (as we found). This resulted in an adjusted urban biomass estimate of 42.6 ± 4.7 Tg C, which is similar to our original land use-based estimate of 42.8 ± 6.8 Tg C. Note that the U.S. Forest Service biomass estimates are based on trees with >15 cm dbh compared to >5 cm dbh for the current study. This methodological difference does not substantially alter the outcomes presented here, as trees in the 5–15 cm size class contained just 3.4% ± 0.4% of the biomass in our field plots.

Reinterpretation of results using different urban definitions and scales

We used five different estimates of urban land cover (U.S. Census Bureau, GRUMP, and our own definition at three spatial scales) to calculate urban aboveground biomass and soil C and N stocks for the Boston MSA (Table 3). The definitions that we developed (>25% ISA over three different neighborhood sizes) were based on a steep decline in ISA when traveling from the urban core of the Boston MSA to just outside the I-95 corridor (see *Urban-to-rural transects and site stratification* under the *Methods* section). The ISA-based definition resulted in urban extents that covered 22.8%, 25.8%, or 29.3% of the sampled land area (Table 2) in the Boston MSA (990-, 270-, and 90-m neighborhood sizes, respectively). By contrast, the U.S. Census and GRUMP urban extents covered 83.2% and 85.5% of the sampled land area within the MSA, respectively (Fig. 1b, c). The land use incorporated by these urban definitions also varied greatly. While the ISA-based urban extents contained 5.8% to 6.8% forest land use by area, the Census and

GRUMP urban extents contained 39.7% and 42.6% forest land area, respectively.

When we extrapolated plot-level data to the Boston MSA using the five different urban extents we found that urban tree biomass varied with the definition of urban. Total urban tree biomass for the MSA was estimated at 4.2 ± 0.4, 4.6 ± 0.9, and 4.9 ± 1.0 Tg C using our ISA-based definitions of urban (990, 270, and 90 m, respectively). By contrast, the U.S. Census and GRUMP urban definitions resulted in urban biomass estimates of 26.5 ± 3.3 and 27.3 ± 3.2 Tg C, respectively. Collectively, the estimated urban biomass ranged from 3.5 ± 0.7 kg C/m² (using our 90-m definition) to 6.6 ± 0.8 kg C/m² (GRUMP definition; Table 3).

Urban soil C and N stocks varied little with urban definition on a per-unit-area basis (12.5 ± 1.1 to 13.0 ± 0.7 kg C/m² and 0.76 ± 0.05 to 0.79 ± 0.07 kg N/m²; extrapolated to 1 m depth; Table 3), but the land area categorized as urban varied dramatically, leading to large differences in urban soil C and N stocks among the urban definitions. The ISA-based urban definitions resulted in total urban soil C stocks of 14.4 ± 0.8, 15.5 ± 1.4, and 17.8 ± 1.6 Tg C (990, 270, and 90 m; Table 3).

Total C and N stocks for the entire study region (urban and nonurban, with soils extrapolated to 1 m depth) did not differ greatly as a result of the urban definition used to scale the data. Estimates of aboveground biomass ranged from 39.8 ± 4.4 Tg C to 44.8 ± 2.4 Tg C (GRUMP and our 990-m definition, respectively) depending on which urban definition was used to scale the data. Soil C ranged from 84.9 ± 5.2 to 89.5 ± 5.3 Tg C (GRUMP and our 90-m definition, respectively). Finally, soil N ranged from 4.6 ± 0.3 to 5.0 ± 0.3 Tg N for the MSA (our 270-m and 90-m definition, respectively), depending on the urban definition.

For the ISA-based urban definitions, decreasing the neighborhood size from 990 m to 270 m, and then to 90 m, resulted in a larger urban footprint. The two smaller neighborhood sizes caused an abundance of isolated, small-scale impervious features (buildings, parking lots, highways) to be classified as urban features (Fig. 6a). At the 990-m neighborhood scale, fewer isolated areas were included in the urban extent, resulting in relatively cohesive clusters of urban land cover around town and city centers (Fig. 6b). Unlike the Census and GRUMP urban definitions, the 990-m neighborhood scale was fine enough to exclude large tracts of forest land from the urban extent (c.f. land use in Fig. 1a and the urban extent in Fig. 1b, c).

DISCUSSION

Is urban vegetation important to regional C balance?

We found that, depending on the definition used, urban areas in the Boston MSA can contain large stocks of aboveground biomass, which have the potential to alter the C balance of the region due to trajectories of

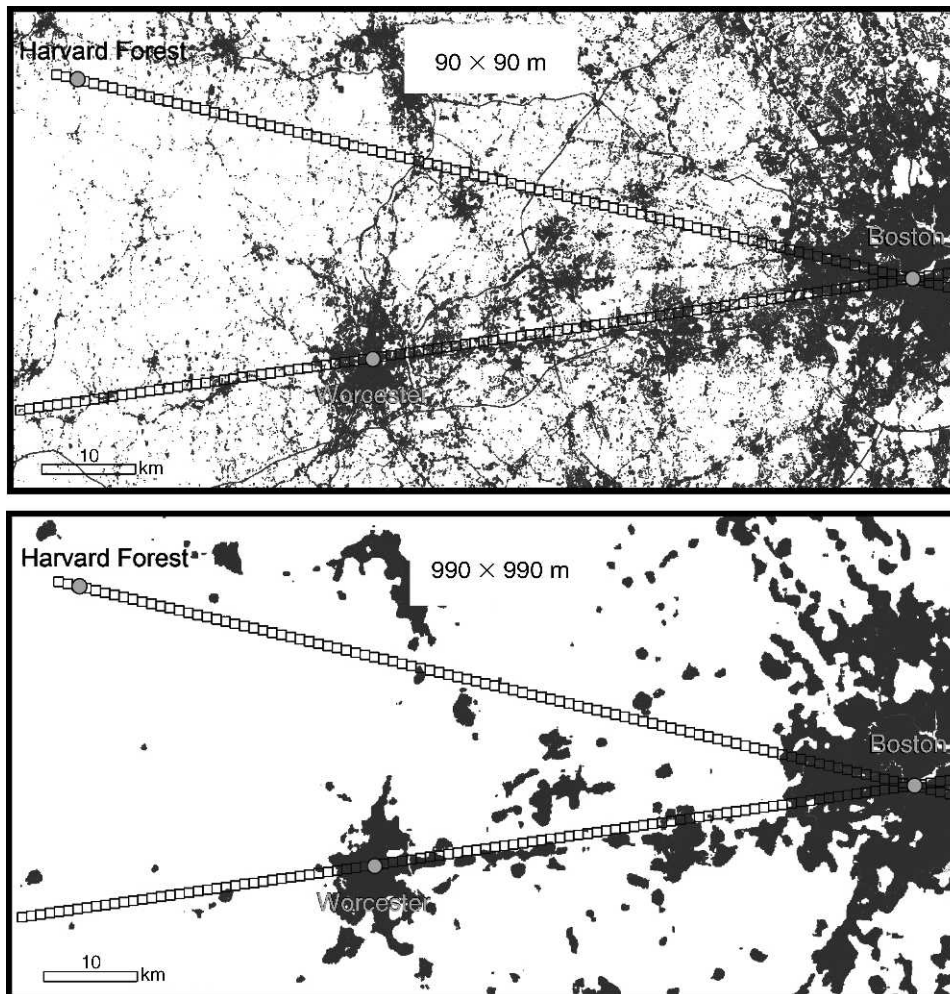


FIG. 6. Distribution of urban areas based on a definition of $>25\%$ impervious surface area (ISA) over (top) 90×90 m and (bottom) 990×990 m neighborhood sizes.

forest loss to development and other threats. While some studies have concluded that the influence of urban vegetation on regional C balance is negligible (see review by Pataki et al. 2011), these studies used different urban definitions and only considered the direct C sequestration (rather than potential losses) of vegetation. These results highlight why the field of urban ecology requires more explicitly stated definitions of what constitutes an urban ecosystem and also why factors beyond active C sequestration must be considered when evaluating the relative importance of urban vegetation. We found that, on a per-area basis, residential land use contains large stocks of aboveground tree biomass (39% as much as forests in the Boston MSA). Commercial, industrial, and developed open spaces (“other developed”) contained smaller, but still considerable biomass stocks (17% as much as forests). However, the extent to which “urban” areas contribute to total biomass is a more complex question that varies strongly based on the definition used by the investigators.

The urban definition that we developed (990 m; Table 1) sets a relatively high threshold ($>25\%$ ISA over 1 km^2) that is based on the physical patterns of land cover found within the Boston area (Fig. 2). Using this definition we might conclude that 4.2 Tg C, or 9.5% of aboveground biomass within the Boston MSA, is contained in urban areas. However, if the U.S. Census definition of urban is used, we might conclude that 26.5 Tg C, or fully 68.4% of the MSA’s woody biomass, is contained within urban areas. Changes to this large C stock could result in significant carbon emissions with future land use change and urban development (Hutyra et al. 2011b); Gurney et al. (2009) estimated fossil fuel emissions for the Massachusetts portion of the MSA to be 12.6 Tg C/yr. The aforementioned review by Pataki et al. (2011) used two case studies to show that direct C sequestration by urban vegetation is negligible when compared to emissions. One case study focused on tree planting in developed areas, while the other focused specifically on vegetation productivity within the census-

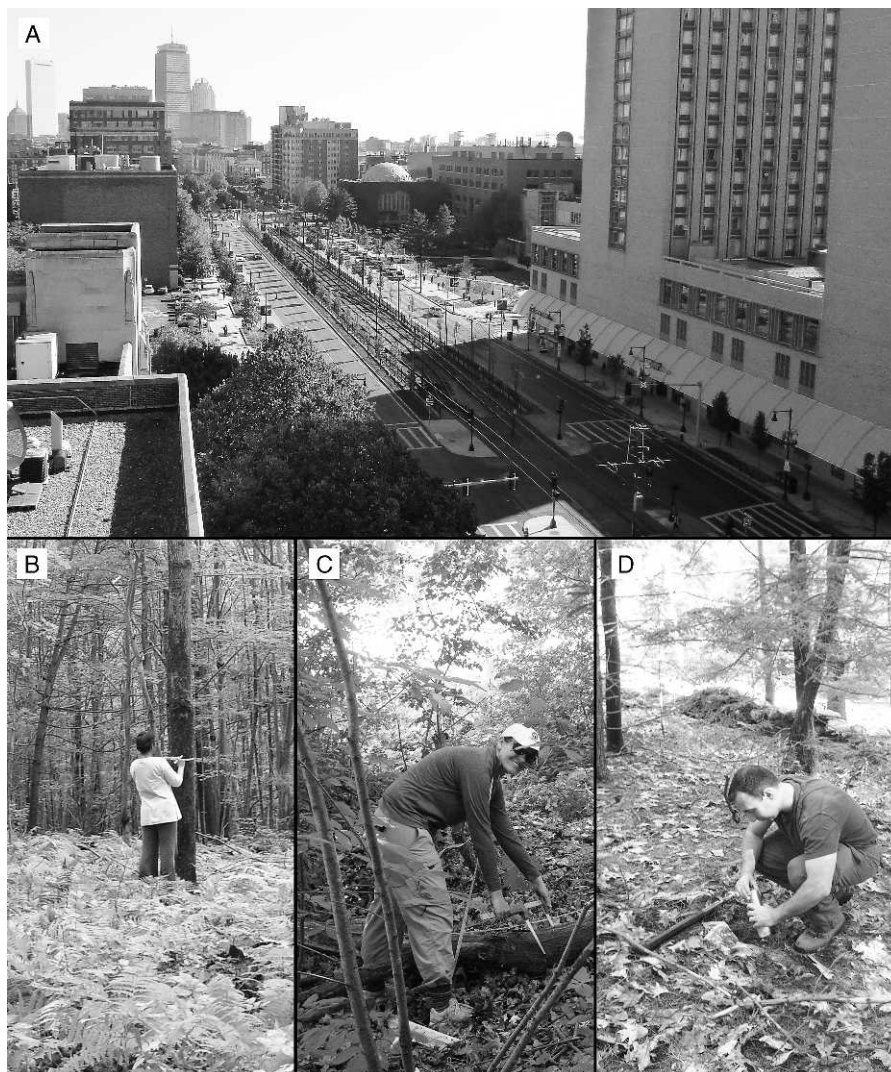


PLATE 1. (A) A view of downtown Boston from a Boston University rooftop. We measured (B) tree species composition and diameter at breast height, (C) coarse woody debris, and (D) soil carbon and nitrogen concentrations at 139 research plots across direct (distance-based) and indirect (distance-independent) urban-to-rural gradients. Photo credits: (A, B) S. M. Raciti, (C) P. Rao, and (D) L. R. Hutya.

defined urban area of Los Angeles County, California, USA, which is densely populated and does not support much naturally occurring, closed-canopy forest vegetation due to the semiarid climate. We agree that direct C sequestration from urban vegetation cannot offset the majority of emissions from urban areas; however, our results suggest that, depending on the location and the urban definition used by the investigator, vegetation may significantly influence the C balance of urban areas in other ways.

With respect to C emissions, the large biomass stocks of the Boston MSA represent a double-edged sword; depending on trajectories of forest growth and patterns of development in the region, they may become C sources or sinks. For the first time in the past 200 years, new development may be outpacing forest recovery in

New England, leading to net loss of forest cover (Foster et al. 2010). As development expands, particularly in heavily wooded suburban and rural areas, some of the region's carbon stocks (and active C sinks) will become sources of C emissions (Stein et al. 2005). The loss of forests to development (Hutya et al. 2011b) is compounded by other threats, including outbreaks of native and invasive pests and pathogens (Lovett and Mitchell 2004), changes in climate, and associated shifts in the frequency of fires, storms, droughts, and other disturbances (IPCC 2007). The question of how long the region's maturing forests will continue to serve as sinks for atmospheric CO₂ is another source of uncertainty that complicates predictions of future forest C storage in the region. There is evidence that net C storage in some forests has plateaued (Fahey et al. 2005), while others

continue to be a strong sinks (Urbanski et al. 2007). Regardless of what the future holds, forests in the Boston MSA are likely to play an important role in regional carbon balance.

Similarly, urban forest biomass may be underestimated at regional scales due to inconsistent definitions of urban land use and the coarse-resolution (relative to urban features) of the remotely sensed data that is typically used for regional-scale biomass assessments. For instance, urban forest biomass in the state of Massachusetts (42.8 ± 6.8 Tg C by our estimates, using the census-defined urban area) may be 50% higher than reported elsewhere (28.7 Tg C; Nowak and Greenfield 2008). Urban areas are characterized by fine-scale spatial heterogeneity, with major changes in land cover occurring over short distances (Cadenasso et al. 2007, McDonnell and Hahs 2008, Pickett et al. 2011). When the features of interest (e.g., urban trees) are smaller than can be resolved by the resolution of the sensor used to measure them, those features may be underestimated (Woodcock and Strahler 1987). This is particularly problematic in cases where the 30-m resolution NLCD canopy cover data set has been applied to highly developed areas. While this data set is reasonably accurate in estimating canopy in rural, forested areas, it severely underestimates canopy in developed land uses (Greenfield et al. 2009, Nowak and Greenfield 2010, Smith et al. 2010). Inconsistent definitions of urban land use can also contribute to underestimation or overestimation of urban tree canopy. For instance, the urban forest biomass estimate reported by the U.S. Forest Service (Nowak and Greenfield 2008) uses field measurements from major cities around the United States. This is not, of itself, problematic, but these biomass estimates were then applied to U.S. Census-defined urban areas, which extend far beyond city limits. In the case of Massachusetts, the U.S. Census urban extent includes areas that would be considered suburban, or even rural in other contexts, and encompasses large tracts of forested land use (42% of the total area). The biomass estimates in this study were obtained using a stratified random sampling design that stretched across a wide gradient of development intensities (Fig. 1b), and should therefore be more representative of the census urban extent than the data used by Nowak and Greenfield (2008).

Does urbanization lead to enhanced C and N sequestration in soils?

Soil C and N concentrations tended to be higher in urban areas compared to nonurban areas (though these differences were not always statistically significant), particularly for areas of the same land use type (Fig. 3f), which is consistent with findings from other metropolitan areas. In the Baltimore, Maryland, region, Pouyat et al. (2002, 2009) found that soil C concentrations were higher in urban forest and residential areas than rural forests. Golubiewski (2006) found similar

trends in the Denver–Boulder, Colorado, region, where urban green spaces had greater concentrations of soil C and N than native grasslands. Among developed land uses, residential areas have shown evidence of greater soil C concentrations than other developed land uses (as we found; Pouyat et al. 2002) and sometimes even greater concentrations than forest land use (Pouyat et al. 2002, Raciti et al. 2011). We expect the main drivers of these differences (e.g., water, N inputs, vegetation shifts) and their magnitudes to vary with climate, with the largest potential differences occurring in water-limited systems that receive irrigation under developed land uses (Golubiewski 2006, Pouyat et al. 2006).

Greater N inputs to urban and residential areas may play a role in enhanced soil C and N sequestration (Qian et al. 2003, Raciti et al. 2011). Several studies have found significantly enhanced atmospheric N inputs in urban areas and near roads (Bettez 2009, Templer and McCann 2010). For managed green spaces, N fertilizer inputs are highly variable, but can be very high. A study of urban and suburban lawns in the Baltimore, Maryland, region found that lawn fertilizer inputs ranged from zero to >300 kg N/ha of lawn area and that 56–68% of homeowners in the study area fertilized their lawns (Law et al. 2004).

For residential land uses in our study, soil N concentrations were high in both urban and nonurban areas, and considerably higher than other nonurban land uses (Fig. 3e), which supports the hypothesis by Pouyat et al. (2006, 2009) that management practices may trump other soil forming factors leading to homogenization of soils in residential landscapes. Lawns are a dominant vegetative cover in residential ecosystems and may contribute to enhanced soil C and N sequestration. A number of studies have shown that lawns have dynamic soil C and N fluxes with considerable potential for organic matter accumulation and N retention (Qian and Follett 2002, Golubiewski 2006, Kaye et al. 2006, Raciti et al. 2008). Studies by Golubiewski (2006) and Raciti et al. (2011) found patterns of increasing soil C stocks with development age across their respective chronosequences. These case studies, combined with the strong positive relationship we found between soil C and N concentrations in all land uses, provides support for the hypothesis that enhanced N inputs may contribute to soil C and N sequestration.

Despite evidence for the accumulation of C and N in urban and residential soils, we cannot be certain of the mechanism driving this pattern due to the complex, sometimes opposing factors controlling C and N stocks. For instance, greater soil moisture availability (due to irrigation) and temperatures in lawns (Groffman et al. 2009) can lead to increases in both plant productivity (increases soil C) and decomposition (decreases soil C). Fertilizer and atmospheric N inputs can also simultaneously increase plant productivity and microbial decomposition (Hu et al. 2000). All of these factors

can alter the balance of organic matter production and decomposition that controls soil C and N stocks (Trumbore 1997) and simple stoichiometric relationships may fail to correctly predict the resultant changes in soil carbon stocks (e.g., Craine et al. 2007). The broader evidence for C and N sequestration in urban and residential soils suggests that changes associated with urbanization tend to increase plant productivity and enhance soil C and N sequestration more strongly than they enhance decomposition; however, in this study, the evidence for N sequestration is more compelling than the evidence for C sequestration, which suggests that increased N inputs to the system may not lead to stoichiometrically equivalent increases in soil C sequestration.

C sequestration by urban soils must be considered in the context of greenhouse gas emissions associated with the management of urban green spaces. Studies of irrigated lawns in the western United States have found that lawns can be significant sources of N₂O emissions (Kaye et al. 2004, Hall et al. 2008, Townsend-Small and Czimczik 2010), but long-term measurements of N₂O fluxes in a temperate climate did not reveal significant differences in N₂O emissions from lawns and forests (Groffman et al. 2009). Groffman and Pouyat (2009) found significant differences in CH₄ fluxes with land use and development intensity, but these differences were not significant in the context of regional greenhouse gas emissions. Townsend-Small and Czimczik (2010) calculated a relatively complete greenhouse gas balance for ornamental lawns and found that lawns could be either a source of GHG emissions or a sink, depending on fertilizer inputs. It is clear that urban green spaces have the potential to sequester C, but their net effect on the planet's radiative balance will depend on a complex suite of factors that include land use history (Raciti et al. 2011), management practices (Townsend-Small and Czimczik 2010), albedo (Betts 2000), evapotranspiration (Georgescu et al. 2011), and fluxes of greenhouse gases beyond CO₂ (Hall et al. 2008).

Estimates of urban soil C sequestration must also consider C and N losses associated with the disturbance and burial of soils under impervious surfaces. Collectively, impervious surfaces cover ~113 000 km² of the continental United States and 1032 km² of the Massachusetts portion of the Boston MSA (Fig. 1b, c). Estimates of urban soil C and N stocks (Table 3) and fluxes are highly sensitive to whether impervious surfaces are included in the estimates (Pouyat et al. 2006). When we used clean fill soils as a proxy for C and N density under impervious surfaces (Pouyat et al. 2006) we saw a 36% decrease in soil C and N stocks in urban areas (using our 990-m urban definition), an 11% decrease in nonurban areas, and a 12% decrease over the Boston MSA. These estimates are highly uncertain as few measurements of soil C concentrations have been made for covered soils; we are unaware of any published measurements of soil N under impervious surfaces. The

role of covered soils in biogeochemical cycling, their hydrological and ecological connectivity to nearby uncovered soils, and the overall fate of C and N stocks over time are all important questions for future research, as paved areas will continue to increase with urbanization.

Regional analyses of C and N stocks must also account for wetland soils and vegetation. The Boston MSA contains ~800 km² of herbaceous and forested wetlands (Table 2). The organic matter pools in these wetlands have not been systematically studied and remain a considerable source of uncertainty in our estimates of C and N stocks.

Characterizing gradients of urbanization: the importance of definition and scale

Studies of urban–rural gradients have often used distance from an urban center (or from urban land uses) as a proxy for the intensity of urban development or influence (e.g., Pouyat and McDonnell 1991, Pouyat et al. 1997, Wolf and Gibbs 2004, King et al. 2005, Ziska et al. 2005, Hutyra et al. 2011a); however, distance is not always a good proxy for degree of urbanization nor does it acknowledge the heterogeneity and complexity of anthropogenic ecosystems (McDonnell and Hahs 2008, Pickett et al. 2011). The variations in the two transects used in this study illustrate these points (Fig. 2). For both transects, distance was a poor predictor of key ecosystem properties at the plot scale. Distance was a strong indicator of aboveground biomass at the neighborhood (~1 km²) scale, but only for the ideal case study represented by the northern transect (Fig. 5a, b). It should be noted that transect length can influence the relationship between a response variable and distance, and that different scales of inquiry might have yielded different results. By contrast, ISA, which is a direct measure of the built environment, was a strong predictor of aboveground biomass for both transects.

While distance-based transects will continue to provide useful information about ecosystem processes, it is clear that more objective and quantitative definitions of urbanization are needed to better explain human–ecological patterns and processes within and among urban ecosystems (McDonnell and Hahs 2008, Hutyra et al. 2011a). As McDonnell et al. (1997) argue, many of the conceptual frameworks used to describe urban ecosystems were developed by geographers, social scientists, and economists, and may not be conducive to the study of human–ecological systems. Further, the terms urban, suburban, and rural have variable, sometimes conflicting, meanings that make it difficult to compare results from different studies and regions.

It is also clear from this study that the population-based definition of urban used by the U.S. Census Bureau may be problematic for the study of urban ecosystems. The Census Bureau measures population density using census blocks that vary greatly in area, which can bias the results. We found that for the 5047

census blocks in Massachusetts, the relationship between census block size and population density approximated a power function, $y = 1142.9x^{-1.002}$, where y is area in km^2 and x is persons/ km^2 ($r^2 = 0.91$). Further, the U.S. Census Bureau allows for “jumps” of up to eight kilometers (five miles) over unsettled areas in determining urban extents. For the Boston MSA, the result is an urban extent that includes large tracts of forest land (42% by land area) and calls into question the usefulness of this urban designation for the purposes of ecological studies.

For many urban ecological studies, a political or administrative boundary is used to define the urban area of interest (city, county, MSA), even when the fluxes and interactions of interest extend well beyond these boundaries (Pickett et al. 2011). While it is true that these boundaries can provide useful points of reference, since major changes in laws and zoning policies can occur there, these study boundaries are most often chosen as a convenient default. However, these boundaries often reflect historical legacies that are unique to a particular city or region, which can create challenges for comparing findings between urban areas. For instance, New York City is divided into five boroughs, which are themselves counties. Compare this to Baltimore, Maryland, which is an independent city that is geographically and politically distinct from neighboring Baltimore County. Furthermore, each of these cities flows into highly urbanized metropolitan areas that are both part of the larger ‘BosWash’ urban corridor.

The problems we describe arise when ecological questions are framed in the context of urban–rural classifications, so ecologists could instead frame their research using quantitative measures of urbanization. However, there are no standard, ecologically relevant metrics of “urbanness” presently in use. Further, it is unclear which metric or metrics would be most useful for this purpose across urban areas. This is a nontrivial problem that has been explored to some extent (e.g., McDonnell et al. 1997), but further work in this area is needed.

The most promising metrics of urban intensity reflect the degree of human modification to the physical, chemical, and biotic landscape. For instance, it is apparent that ISA is correlated with soil and vegetation C and N stocks. This measure of urban intensity is, in turn, correlated with a host of other environmental factors that change with urbanization, such as temperature and growing-season length (Zhang et al. 2004), atmospheric N deposition (Bettez 2009), CO_2 concentration (George et al. 2007), ground-level ozone (Gregg et al. 2003), water availability (Martin and Stabler 2002), and heavy metals in soil (Pouyat et al. 2008). Population density, a metric commonly used to delineate urban areas, was also correlated with the C and N stocks in our study, but more weakly than ISA because nonresidential land uses have zero population density, despite objectively and subjectively contributing to the

intensity of urban development in a region. Thus, the connections between measures of urbanization and environmental factors may provide mechanistic insights regarding how urbanization influences ecological form and function.

We suggest that a moving-window approach, like the one we used to define urban intensity in this study, can provide a continuous measure of biophysical conditions that is independent of political boundaries and may allow for more useful comparisons between disparate urban areas. We chose ISA, measured over an $\sim 1\text{-km}^2$ neighborhood area, as a proxy for the intensity of urban development because it provided a simple, objective measure of the physical environment. Our original stratification scheme used population density to further divide the resulting urban area (defined as $>25\%$ neighborhood ISA) into two classes (high population density and lower population density), but it became clear that ISA provided similar information (Fig. 4) and more explanatory power than population density by accounting for developed areas that did not contain any population (e.g., commercial and industrial areas). We chose a neighborhood size of $\sim 1\text{ km}^2$ because this resulted in relatively cohesive urban areas. Smaller scales (270 m and 90 m) resulted in scattered islands and tracks of urban land use around individual buildings and parking lots and across major highways (Fig. 6a, b). We used a single, univariate measure of urbanization for this study, but we believe that a broader index (or set of indices) that includes social, political, physical, and economic factors would help advance the study of coupled human–ecological systems. Such an index might provide insight into the complex interactions between humans and the built and natural environment and facilitate more detailed comparisons between urban areas around the globe (McDonnell and Hahs 2008).

CONCLUSIONS

The process of urbanization can alter aboveground and belowground C and N stocks. Developed land uses in the Boston MSA contained considerable aboveground biomass, but much less than the forests they typically replaced. In urban areas, forest and nonresidential land uses may contain greater concentrations of soil C and N than their nonurban counterparts. Residential land uses, by contrast, contained relatively high concentrations of soil C and N regardless of urban intensity. Factors such as temperature, growing-season length, CO_2 concentration, and N deposition can vary substantially across urban gradients and may contribute to enhanced soil C sequestration in unmanaged, or less intensely managed, green spaces. For highly managed green spaces, such as residential lawns, management activities may overwhelm other soil-forming factors. While there is evidence for C sequestration in urban soils, this sequestration must be considered in the context of greenhouse gas emissions associated with the maintenance of urban green spaces and in the

context of overall urban greenhouse gas emissions. The soils under impervious surfaces must also be accounted for in regional C and N stocks, but the properties of these soils have not been well characterized and constitute an important area for future research.

We found that estimates of urban C and N stocks changed dramatically depending on the definition of urban that we used. Based on this outcome, we recommend that ecologists be more explicit when using the terms “urban,” “suburban,” and “rural,” by clearly defining their meaning in the context of the study system. Where reasonable, ecologists should consider exploring ecological responses in relation to quantitative measures of urbanization, such as impervious surface area or the proportion of developed land use in the neighborhood around field plots. Alternatively, they might explore more direct measures of urban influence on study systems or organisms, such as the concentrations of heavy metals in soil (Pouyat et al. 2008) or temperature changes that arise from urban heat islands (Zhang et al. 2004). Finally, more consistent and quantitative measures of urbanization would help to advance the study of urban ecosystems by providing greater insight into drivers of environmental patterns and processes and facilitating comparisons within and among urban areas.

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