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Multi-scale heterogeneity in vegetation and soil carbon in exurban residential land of southeastern Michigan, USA

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Abstract. Exurban residential land (one housing unit per 0.2–16.2 ha) is growing in importance as a human-dominated land use. Carbon storage in the soils and vegetation of exurban land is poorly known, as are the effects on C storage of choices made by developers and residents. We studied C storage in exurban yards in southeastern Michigan, USA, across a range of parcel sizes and different types of neighborhoods. We divided each residential parcel into ecological zones (EZ) characterized by vegetation, soil, and human behavior such as mowing, irrigation, and raking. We found a heterogeneous mixture of trees and shrubs, turfgrasses, mulched gardens, old-field vegetation, and impervious surfaces. The most extensive zone type was turfgrass with sparse woody vegetation (mean 26% of parcel area), followed by dense woody vegetation (mean 21% of parcel area). Areas of turfgrass with sparse woody vegetation had trees in larger size classes (> 50 cm dbh) than did areas of dense woody vegetation. Using aerial photointerpretation, we scaled up C storage to neighborhoods. Varying C storage by neighborhood type resulted from differences in impervious area (8–26% of parcel area) and area of dense woody vegetation (11–28%). Averaged and multiplied across areas in differing neighborhood types, exurban residential land contained 5240 ± 865 g C/m^2 in vegetation, highly sensitive to large trees, and 13800 ± 1290 g C/m^2 in soils (based on a combined sampling and modeling approach). These contents are greater than for agricultural land in the region, but lower than for mature forest stands. Compared with mature forests, exurban land contained more shrubs and less downed woody debris and it had similar tree size-class distributions up to 40 cm dbh but far fewer trees in larger size classes. If the trees continue to grow, exurban residential land could sequester additional C for decades. Patterns and processes of C storage in exurban residential land were driven by land management practices that affect soil and vegetation, reflecting the choices of designers, developers, and residents. This study provides an example of human-mediated C storage in a coupled human–natural system.

Key words: carbon sequestration; carbon storage; exurban landscape; human-dominated landscape; land use; landscape; scaling; soils; spatial heterogeneity; tree cover; urban landscape.

Introduction

The balance of carbon exchange between the atmosphere and units of the terrestrial surface has long been a key focus of ecosystem and global change science. Land use/land cover (LULC) change is a major cause of changes in terrestrial sources or sinks of carbon (C) to and from the atmosphere (Houghton 1999, Caspersen et al. 2000, Watson et al. 2000). Carbon storage can be altered not only during conversion from one land use or land cover to another, but also by changes in land management practices. Altered dynamics in both vegetation and soil C pools can occur for decades to centuries, causing either a rising or falling trajectory of ecosystem C storage. On prior forest land or abandoned agricultural land, trees can regrow, storing large amounts of C in woody biomass (Caspersen et al. 2000, Rhemtulla et al. 2009). For many types of land conversion or land use change, changes in rates of soil C storage can occur as a result of altered annual inputs of foliar, root, and woody litter, changes in soil management resulting in altered decomposition rates, or both (Watson et al. 2000). These processes have been studied at decade to century time scales in diverse examples of LULC change, such as forest regrowth following harvest (Yanai et al. 2003), conversion of agricultural land to forests (Currie and Nadelhoffer 2002, Hooker and Compton 2003, Laganière et al. 2010), and reclamation of mineland to grassland (Simmons et al. 2008).

An additional type of LULC change occurring in the US is low-density exurban development of residential parcels (Brown et al. 2005). The phenomenon of many

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residential households choosing to live farther from urban centers, together with widespread conversion of agricultural land to residential land in the 20th century, led to an expansion of residential land at the urban–rural fringe. In the conterminous United States, exurban residential land use (defined for this purpose as one housing unit per 0.2–16.2 ha) grew from ~271000 km^2 in 1950 to 1.39 million km^2 in 2000 (Brown et al. 2005). By 2000, land settled at exurban residential densities accounted for 15 times the area of land settled at suburban or urban densities (one housing unit per < 0.2 ha) in the United States (Brown et al. 2005). In comparison to suburban land use, exurban development contributes to sprawl that reaches much farther from urban centers and into previously rural land use.

Relatively little work has addressed changes in carbon pool sizes that occur in the decades that follow the conversion of land from other types of land use and cover to residential use in exurban landscapes (Churkina et al. 2010). To account for C in human-dominated residential landscapes, regional- to global-scale C budgets have needed to rely on extrapolations from wildland systems including forests and grasslands, which have been more widely studied by ecologists (e.g. Botkin et al. 1993, Houghton 1999). However, measurements made in wildlands or lightly impacted ecosystems are likely to poorly represent exurban residential land. The mixtures of grasses, shrubs, and trees, together with the soils or plant–soil assemblages in the residential landscape, result in large part from human activities and human preferences (Nassauer 1995). In southeastern Michigan, residential subdivisions are designed and constructed by firms (hereafter, developers) that choose whether to cut trees or leave trees in place, grade the soil using heavy machinery to improve access and manage drainage, and choose whether to establish horticultural plants, including turf grass. Within the constraints of planning policies (e.g., wetlands protection, zoning), landscape designers and developers determine parcel sizes and shapes, and whether small wetlands, ponds, grassy and shrubby old-field areas, or other natural-like areas are incorporated into subdivisions (Nassauer et al. 2014). Regarding wetlands protection, the Michigan Department of Environmental Quality, Land and Water Management Division, aligns state and federal wetland protection regulations under the state Protection Act (PA) 451 of 1994, which stipulates the conditions that could be imposed on development due to potential impacts on wetlands. Regarding zoning, the Michigan Zoning Enabling Act (Act 110 of 2006) stipulates the laws that local units of government can use in the regulation of development and the use of land. Outside of developer subdivisions, other parcels in the landscape are developed by individual households that purchase prior agricultural or forested land directly and decide whether to keep trees, wetlands, or old fields during home construction (we refer to such parcels as rural lots; Brown et al. 2008).

Over the time scale of decades, residential landowners make numerous household choices that affect vegetation and soils: whether to plant, prune, or remove trees, in the case of either endemic or horticultural trees; whether to fertilize, irrigate, or seed lawns, mow lawns, bag and remove cut turf grass or mulch it with the mower; whether to rake and remove, compost, or burn fallen leaves; and whether to burn fallen trees and large branches (Nassauer et al. 2014, Visscher et al. 2014). These actions produce human-dominated vegetation communities and alter C cycling (Kaye et al. 2006, Ellis and Ramankutty 2008, Luck et al. 2009, Hutchins 2010).

Here we report the results of an empirical study of C storage in the exurban residential landscape. Our study region focuses on 10 counties in southeastern Michigan, USA, which include the Detroit, Ann Arbor, and Flint metropolitan areas. The number of households in this 10-county region increased from 1.92 million to 2.08 million from 1990 to 2000 (US Bureau of the Census 2001), with much of the expansion occurring as low-density, exurban development (Brown et al. 2005, 2008). Exurban expansion is part of a national trend; its rate was 25% greater than the rate of population growth between 1980 and 2000. Nationwide this has led to a cumulative total of 11.8% of land area occupied by exurban residential development and only 1.6% occupied by suburban and urban development (Theobald 2005). In our study region, much of the development has occurred on prior agricultural land, like much of the exurban expansion in the eastern United States (Brown et al. 2005, 2008). Aerial photointerpretation has shown a significant increase in tree cover in townships undergoing exurban expansion in this region (Brown et al. 2008, An et al. 2011). Remote sensing of gross primary production (GPP) in our region has demonstrated a strong association between GPP and increased density of housing units at the exurban fringe. Over the period 1991–1999, GPP increases averaged 125 $g\ C \cdot m^{-2} \cdot yr^{-1}$, a 6.5% increase from an initial 1930 $g\ C \cdot m^{-2} \cdot yr^{-1}$, following densification of a census block group from rural (one housing unit per > 16.2 ha) to exurban densities (Zhao et al. 2007). We expected that C storage in exurban land in this region should be greater than that of agricultural land, but below that of temperate forests.

This research was conducted as part of a larger collaboration, the SLUCE project (Spatial Land Use Change and Ecological effects, Brown et al. 2008). In the SLUCE project we are taking the perspective of studying a coupled human–natural system in which human choices and behaviors alter ecosystem structure and function, which then affects the delivery of ecosystem services to society (Liu et al. 2007, Walsh and McGinnis 2008). Results of the present study are being used to parameterize the linkage of ecological models to agent-based models, in which developers and residents are explicitly simulated as human agents with environmental decision-making functions, to understand social-ecological drivers of landscape C balance (Robinson et al. 2013).

Methods

Site selection

We measured C pools in vegetation and soils in 26 residential parcels overall, using an approach that allowed us to use remotely sensed maps of vegetation across different types of neighborhood designs to scale our results up to regionally representative exurban residential land. To incorporate human actions in an overarching conceptual and modeling framework, our starting point for site selection was a set of ~600 respondents to an internet-based survey of residential landscape preferences conducted in prior work in this 10-county region (Nassauer et al. 2009). Of the internet-survey respondents from across the region, we selected 66 addresses of residential parcels (ownership lots) that were located in exurban areas within 13 particular townships selected to cover the stages of residential development in the region. Subsequent analysis of development decade and prior land-use history (see following sections) confirmed that parcels in the subset of these 66 chosen for field sampling were regionally representative.

Soil textures range widely in our region depending on surficial geology, with clay-rich soils a minority (National Resources Conservation Service 2008, Kahan et al. 2014). It is also likely that disturbance by residential developers created additional soil heterogeneity (Raciti et al. 2011). Because soil organic matter generally correlates with clay content (Homann et al. 2007), the inclusion of a minority of clay soils in our study could confound comparisons of soil C among vegetation patches. We excluded parcels that were likely to have high clay content using three complementary methods. We excluded those that fell in the Erie-Huron Lake Plain category for surficial geology (National Resources Conservation Service 2008), as well as those where soils had clay or clay loam texture in STATSGO data (National Resources Conservation Service 1995). Finally, because lake-plain and other clay-rich soils in this region tend to occur in lowland, relatively wet topography, we calculated the topographic wetness index (TWI) across the region (Rodhe and Seibert 1999) at 30-m resolution and excluded areas with TWI > 10. From the households that remained (n = 53), we requested voluntary participation in an on-site interview and site survey. Twenty-one households agreed to participate. To include a greater number we systematically located additional households nearby and requested voluntary participation, yielding five additional households for a total of 26 in nine townships. Parcel sizes that we studied ranged from 0.090 to 2.190 ha (median 0.257, mean 0.574 ha).

Our previous research in this region has divided exurban residential parcels into four types based on parcel size, road access and subdivision layout, the amount of tree cover, and other factors (Brown et al. 2008, An et al. 2011). This typology included three types of subdivisions constructed by developers, plus rural lots, which are not subdivided by developers, but are individually subdivided parcels with direct access to a public road. Rural lots range widely in size. Of the other three types, remnant subdivisions tend to have large patches of trees left by the developer; horticultural subdivisions tend to have curved internal roads and planted trees and shrubs; and country subdivisions have mainly linear internal road systems, smaller parcel sizes, and fewer trees. We sampled multiple sites from each of the four types. While land use history was not part of the site selection, we sampled parcels that had been converted to residential land use in each decade from the 1960s to 2000s, as later determined by historical aerial photo-interpretation (see following sections).

Site surveys and sample collection

At each of the 26 participating households we conducted two closely integrated studies: a detailed interview of residents focused on household behaviors and preferences reported by Nassauer et al. (2014) and an ecological survey of soils and vegetation, reported here. We adapted ecological methods that had been designed to measure vegetation and soil C pools in wildland sites (e.g. Harmon et al. 1986, Huntington et al. 1988, Minnesota Department of Natural Resources 2007) to accommodate study of the human-dominated environment. We avoided intensive or destructive sampling of vegetation and designed field surveys to be conducted rapidly. Since landowners were asked to participate voluntarily, intensive repeat visits or destructive sampling would be likely to reduce participation and result in a non-representative sample. The ecological surveys were conducted between 20 June and 5 August 2009. Prior to each survey, all buried utility lines (telephone, cable, power lines) were marked.

Interviews with residents, aerial photointerpretation, and our knowledge of the region allowed us to identify, within each yard, discrete areas of mixed vegetation types. We termed these discrete areas ecological zones (EZs). Zones were defined by the types and densities of vegetation in all vertical layers (herbaceous, shrub, understory and overstory trees) and evident soil characteristics such as standing water or the presence of mulch. We developed one region-wide set of seven EZ definitions that were reused across all parcels: five EZs were dominated by vegetation, together with zone types impervious cover and water (definitions and characteristics are given in Table 1). The purpose of this approach was to allow rapid identification of discrete zones by workers in the field and sampling of soil and vegetation by zone, followed by upscaling using aerial photointerpretation.

Prior to field sampling, crews pre-identified the types and extents of EZs using high-resolution aerial photographs. Field crews emphasized zone types that made up at least 5% of the parcel area to avoid over-dividing yards. EZ types and boundaries were ground-truthed by field workers at each site, with corrections made on printed aerial photos for later digitizing. Field crews then

TABLE 1. Description and importance of ecological zones (EZ).

Ecological zone (EZ) type	Description	n	Mean proportion of parcel area
Turfgrass with sparse woody vegetation	Turfgrass present and managed. Trees or shrubs present, but gaps present between canopies. Edges of zone defined as where edges of woody vegetation shadows fall at mid-day in summer.	24	26.3%
Dense woody vegetation	Managed turfgrass generally absent. Trees and/or shrubs present and dense. Foliar canopy of woody vegetation is closed so that little direct light reaches the herbaceous layer.	8	22.1%
Old field	Managed turfgrass absent. Tall herbaceous vegetation present. Trees or shrubs sometimes present, but with gaps between canopies.	6	20.5%
Turfgrass	Turfgrass present and managed. No woody vegetation present.	24	16.6%
Mixed shrubs, forbs, herbaceous, with mulch[a]	Managed turfgrass absent. Trees sometimes present. Shrubs sometimes present, generally pruned. Managed herbaceous vegetation sometimes present. Managed mulch layer often present.	18	2.7%
Impervious	Structures and pavement.	26	10.8%
Water	Standing or moving water without significant emergent vegetation.	4	1.0%

Notes: Mean proportions of parcel areas are across the entire study for each EZ, n = number of parcels (of 26 total) that contained each EZ zone type. The impervious category comprised structures (n = 26, mean parcel area proportion = 6.4%) and pavement (n = 26, mean parcel area proportion = 4.4%). Mean proportions of parcel areas sum to 100%. (a) Mixed shrubs, forbs, and herbaceous with mulch is hereafter referred to as gardens.

used an adapted relevé method (Barbour et al. 1999, Minnesota Department of Natural Resources 2007) within each EZ for sampling and field measurements. Within an EZ, workers chose a typical area as a center point, then established a transect with a random compass heading extending in both directions, with maximum length 50 m (shorter if it reached the edge of the EZ or the parcel). One transect was established per EZ in two to four of the EZs in each site (excluding impervious and water) depending on their importance. Across the 26 sites (parcels), 80 EZ transects were established overall with lengths ranging from 6.6 to 50 m (median 25.1 m, average three per site).

Each EZ transect then defined the longitudinal centerline of a plot 5 m wide and the length of the transect. These EZ plots ranged in area from 33 to 250 m^2 (median 125.5 m^2). These and nested subplots were used to quantify vegetation cover in multiple vertical layers (Barbour et al. 1999). To quantify overstory and understory trees and tall shrubs, the entire EZ plot area was used. Each woody stem with dbh (diameter at breast height; 1.4 m above ground) ≥ 2.5 cm whose center point lay within the plot was tallied. For each multi-stemmed large shrub, the number of stems ≥ 2.5 cm was recorded together with diameter of the median-sized stem on the individual shrub. For each tallied stem, workers recorded dbh, plant functional type (PFT), genus, species (where feasible), condition, canopy position, and height using a clinometer. PFTs were defined as deciduous tree, deciduous shrub, coniferous evergreen tree, and coniferous evergreen shrub. Distinction between tree and shrub was based on typical growth habit for the species; thus understory tree saplings were counted as trees, not shrubs. About 12% of trees and shrubs were not easily identifiable to species because horticultural varieties were encountered (Balmford et al. 1996, Kahan et al. 2014).

Subplots were established to assess herbaceous vegetation cover, and herbaceous vegetation and litter were sampled quantitatively by area. Within each EZ plot, two 2 × 2 m subplots were established in areas judged to be representative (n = 160 overall across the study; Rutkowski and Stottlemeyer 1993). In each, the proportion of herbaceous cover was recorded for later scaling. In each subplot, a 25 × 50 cm template was placed in herbaceous cover (n = 160 overall) and all living and standing dead herbaceous biomass was clipped to the surface of the soil and placed in a paper bag for later determination of dry mass and C content. In garden zones, herbaceous vegetation was not clipped; these were judged to be negligible contributions to carbon and residents preferred not to have their flowers clipped. In each 25 × 50 cm clipped area, the litter layer (Oi horizon) was sampled quantitatively by cutting around a 15 × 15 cm square template with a knife and by placing the litter layer in a paper bag for later determination of dry mass and C content. Directly beneath the first litter sample taken from each EZ plot (n = 80), a bulk density soil corer designed to quantitatively remove a specified volume of soil (5 cm diameter, 15 cm depth). First we sampled the upper soil from the top of the Oe horizon (where present) into the mineral soil (Fisk et al. 2002, Zak et al. 2008) to 15 cm depth, hereafter referred to as a surface soil core. We took a second bulk density soil core from 50 to 65 cm depth. Soil from 15 to 50 cm depth was returned to the hole made by sampling. We selected five sites for two repeat visits to the same plots, yielding 20 additional bulk density surface soil cores. After laboratory analysis, data were aggregated over surface soil cores from the same site and plot. Overall, 180 soil cores were collected and reported here.

EZ transects were used for several additional purposes. First, three non-vegetation cover categories embedded within the EZ (bare soil, rock or pavement, and water) were recorded as lengths along the transect centerline. Second, downed woody debris was quantified. Along the full EZ transect centerline, for each piece of downed woody debris ≥ 2.5 cm diameter that crossed the vertical plane defined by the transect, workers recorded the diameter, angle with the horizontal, and degree of decomposition using standard decay classes of sound, medium decay, and rotten for pieces < 5 cm diameter, and from 1 (sound wood) to 5 (highly decomposed) for pieces ≥ 5 cm diameter (Sollins 1982, Currie and Nadelhoffer 2002). Standard methods were used to scale these data to pools of biomass contained in downed woody debris for each zone (Harmon and Sexton 1996), including wood densities that varied by size and decay class (Currie and Nadelhoffer 2002). Transects were also used to sample woody branches of live trees and shrubs at random ($n = 26$ overall) for use in dry mass to carbon conversions for woody biomass.

On many parcels, woodpiles and compost piles were encountered. Woodpiles were generally stacked wood intended for burning; compost piles generally included prunings, grass clippings, twigs and branches, and leaves. These piles were not observed consistently enough in any one EZ category in the field to be included in the definition of a particular EZ category. We measured the volumes of all woodpiles and compost piles on all 26 parcels.

Laboratory analysis

Vegetation, litter, and soil samples were returned to the University of Michigan for analysis. Soil cores were weighed in their field-moist state then air-dried and re-weighed (for gravimetric moisture determination), live roots discarded, soils sieved (2 mm), and re-weighed. From each sample, a subsample was taken for oven-dry (105°C) mass correction, another (~10 g) was removed for pH measurement in a 1:1 slurry of 0.01 mol/L $CaCl_2$, and another (~50 g) was ground to a fine powder in a ball mill for C analysis. Vegetation and litter samples were air-dried (55°C), weighed, and chopped completely in a food processor. Subsamples (~5 g) were ground to a fine powder in a ball mill for analysis of C. We measured C concentrations in each vegetation, litter, and soil sample individually by dry combustion using a NC 2500 elemental analyzer (CE Elantech, Lakewood, New Jersey, USA) interfaced to a Delta Plus isotope-ratio mass spectrometer (Thermo Finnegan, San Jose, California, USA).

Scaling of C pools to ecological zones

Soil bulk densities were calculated based on corer volume using total sample masses after correction for moisture and coarse fraction (> 2 mm). Bulk densities together with measured C concentrations were used to calculate C pool sizes in 0–15 and 50–65 cm depths. In summaries of soil C used by the Intergovernmental Panel on Climate Change (IPCC), 1 m soil depth is used as a benchmark for comparison of soil C pools among biomes (Watson et al. 2000).

We used a single-exponential model of soil C with depth to estimate the total soil C pool to 1 m depth. This is similar to the log-linear model that was found by Jobbagy and Jackson (2000) to be significant in explaining distributions of soil C with depth in 76% of soil profiles analyzed. To further explore the validity of using this simple model of soil C with depth, we re-analyzed the results for soil organic C (%) with depth, in which complete soil cores were sampled to 1 m depth, reported by Raciti et al. (2011) in a study of residential yards in Baltimore, Maryland, USA. We found that a single-exponential model fit the data of Raciti et al. (2011) with $P = 0.032$ and $r^2 = 0.99$. We thus fit curves of exponential decline in soil C with depth to our two soil C measurements from 0 to 15 cm (which included the Oe horizon) and 50–65 cm depths for each EZ plot individually and integrated each of these curves to a depth of 1 m. In nine of 80 cases the deeper soil sample had a higher C concentration than the shallow sample; in these cases we applied the average of the two C measurements across the entire 1 m depth.

Vegetation biomass was summed in herbaceous and woody categories for each of our 80 EZ plots separately. We scaled herbaceous biomass up from clipped subplots. Herbaceous scaling included two quantitative corrections for non-vegetation cover (bare soil, rock or pavement, water): (1) cover estimates from our representative 2 × 2 m subplots and (2) recorded proportions of these cover categories along the EZ transects. We used published allometric equations to convert tree diameters to aboveground biomass per individual (Ter-Mikaelian and Korzukhin 1997). For shrub biomass, we used our own allometric equations developed by harvesting 44 shrubs locally (data not shown). Belowground (root) biomass was included in vegetation pools, based on aboveground biomass and using broadly estimated shoot:root ratios that differed among woody plants (4:1), shrubs (1.5:1), and grasses (1.5:1) (Leemans 1997). To correct for the fact that many trees in our study were in open environments as opposed to closed-canopy forests, we multiplied allometrically calculated biomass values by a factor of 0.8, empirically measured by McPherson et al. (1994) for urban environments; we applied this factor to all trees (excluding shrubs) in all EZ types except dense tree cover. In addition, for standing dead trees and shrubs, biomass values were reduced by 15% to account for loss of wood density (Harmon 1982), included in vegetation pools and amounted to 16.8% and 6.6%, respectively, of tree and shrub pools. We used the area of each EZ plot to express woody vegetation biomass per unit area. To convert herbaceous, litter, and soil dry mass to C, we used our own analytical measures of C concentration on each sample individually. These values averaged, on an air-dry (55°C) ash-included basis, 0.44 g C/g for herbaceous vegetation, 0.45 g C/g for woody biomass, and 0.33 g C/g for fine litter samples (which contained some mineral grains).

Scaling to the landscape

We scaled C pools in vegetation and litter, modeled C pools in soils, and frequency distributions of tree and shrub stem counts up to the heterogeneous landscape surrounding each study parcel (n = 26), using the data from our 80 EZ plots, through a series of steps. First, aerial photographs were obtained from the National Agriculture Imagery Program (NAIP) for 2005 at ~2-m resolution. These were used to determine neighborhood types (rural lots or subdivision types) containing each field-surveyed parcel, based on criteria in Brown et al. (2008) and An et al. (2011). Within a 1 km radius of each study parcel and only within the same neighborhood type as the study parcel at its center (i.e. excluding other neighborhood types within 1 km as well as other land use/land cover, such as golf courses, school fields, agricultural fields, and so on), we then digitized and visually interpreted the areas in these aerial photographs into traditional, or Anderson-based land cover (LC) classifications (Anderson et al. 1976, Cadenasso et al. 2007). These included tree cover, maintained, impervious cover, open natural, water, wetland, and crop with ~2-m resolution (Robinson 2012). For each field-surveyed parcel, EZs mapped in the field were digitized into our five EZ categories and overlaid with the Anderson-based LC classifications in ArcGIS. This overlay created a matrix to map our field-identified EZ categories onto Anderson-based LC categories for scaling to the landscape.

Individual pools of vegetation, litter, and modeled soil C pools, as well as frequency distributions of trees and shrubs, were averaged across the entire study by the five types of EZ. Data were not deemed adequate to allow separation of individual C pools or stem frequencies by combined EZ type and neighborhood type. The areas of each Anderson-based LC category at the landscape scale (1 km radius) surrounding each parcel were then multiplied by our matrix to produce landscape-scale areas for each EZ category within 1 km. These areas were then multiplied by our cross-study averages of vegetation, litter, and soil C pools, as well as frequency distributions of trees and shrubs, by EZ category. In this manner, differential frequencies and areas of EZ categories (such as turfgrass vs. dense trees and shrubs) could produce differences in scaled-up C estimates in vegetation and soils, and frequency distributions of trees and shrubs, for different neighborhoods. We did not measure C pools beneath impervious cover, which included driveways, paved footpaths, and structures; for these areas we used zero vegetation and litter C and 3300 g C/m^2 for mineral soil C (Pouyat et al. 2006).

Woodpiles and compost piles entered our landscape-scaling calculations on the basis of their masses per parcel area, not associated with any EZ areas. We measured volumes and biomass, using certified spring scales, of material selected from six representative piles. Field-moist to air-dry mass corrections were determined on subsamples. Air-dry to oven-dry mass corrections, an ash content correction for compost (9.6%), and C concentrations were used from prior studies of similar materials from forests (Currie and Nadelhoffer 2002, Currie 2003). Observations of woodpiles and compost piles were not deemed sufficient to differentiate these C pools by neighborhood type. The median, parcel area-based pool of C in woodpiles and compost piles, when present, were 164 and 9 g C/m^2, respectively. Medians were used because there was one outlier parcel for woodpiles, with 2660 g C/m^2 averaged over the single parcel. Median values were multiplied by the observed overall frequencies of parcels having a woodpile (0.42) and a compost pile (0.19), producing an average landscape contribution of piles in residential neighborhoods (71 g C/m^2) that we applied across all neighborhood types. These piles were not included in litter or soil pools in EZ C totals, but were included in total ecosystem C sums when scaled to the landscape.

Although 26 parcels were sampled, scaling calculations produced only 22 landscape-scale sets of C pools for further analysis. In one case, aerial photointerpretation identified the neighborhood as urban based on housing density so the site was excluded from further analysis. In three other cases, the parcels were close to one another in the same neighborhood, so in the landscape scaling, the sites were merged.

In upscaling C pools to the landscape, we included a formal analysis of uncertainty propagation. Where x_1, x_2, … x_n are random variables with sample variances s_1^2, s_2^2, … s_n^2, such as variances in the set of C pools we measured at the EZ scale, and where $f(x_1, x_2, \dots x_n)$ is a function we used in upscaling, we calculated s_f as the upscaled uncertainty (Arras 1998):

$$S_f = \sqrt{\left(\frac{\partial_f}{\partial x_1}\right)^2 \cdot S_1^2 + \left(\frac{\partial_f}{\partial x_2}\right)^2 \cdot S_2^2 + \ldots + \left(\frac{\partial_f}{\partial x_n}\right)^2 \cdot S_n^2} \quad (1)$$

Finally, for each of the 22 neighborhoods, we established the time since conversion to residential land and the prior land use history using a series of aerial photographs, one set per decade, taken from the 1950s to 2005 (Brown et al. 2008). Categories of prior land use history were defined as (1) agriculture-cropland if there was any evidence in any aerial photograph of plowing or row crops prior to development; (2) agriculture-pasture/old field, if there was never evidence of plowing or row crops; (3) continuous tree cover; and (4) tree farm/orchard. For EZ- and landscape-scale results, statistical analyses were performed in Stata 11.0 (College Station, Texas, USA). Differences among soil and vegetation C pools by EZ category, subdivision type, and category of prior land use history were tested using ANOVA followed by Bonferroni mean-comparison tests. We also tested whether soil C stocks in surface soil cores, deep soil cores, and mineral soil to 1 m depth correlated with physiographic variables water-holding capacity (WHC), texture class, and surficial geology (National Resources Conservation Service 1995), and topographic wetness index (TWI). For all tests, $P < 0.05$ was considered significant.

Results

Vegetation

By area, the dominant EZ type in the 26 residential yards that we surveyed and sampled was turfgrass with sparse woody, followed by dense woody, old field, and turfgrass zone types (Table 1). Carbon pools in trees, shrubs, and herbaceous vegetation differed among EZ types (Table 2a). For trees, the dense woody zone type contained the most C per area, followed by turfgrass with sparse woody. For shrubs, the dense woody, old field, and garden zone types contained more C per area than other zone types. The most C per area in herbaceous vegetation was found in old field, followed by turfgrass. Total vegetation C storage (summed across trees, shrubs, and herbaceous vegetation, expressed per unit area, g C/m^2) also differed by EZ (Fig. 1), driven mainly by the differences in trees (Table 2a). The dense woody zone type had the greatest total vegetation C per area. The turfgrass with sparse woody zone type had greater total vegetation C per area than the turfgrass zone type, but less than the dense woody type. Old field and garden zone types had intermediate amounts of total vegetation C per area (Table 2a, Fig. 1).

The tree species we encountered most frequently in exurban yards were (in descending order) *Fraxinus americana*, *Acer rubrum*, *Ulmus rubra*, *Carpinus caroliniana*, and *Prunus serotina*. Additional genera that were important but were not in the top five species because they contained multiple species within each genus included *Populus*, *Quercus*, and *Picea*. Other trees that were less frequent but interesting to note for historical and horticultural reasons included *Juglans nigra* (black walnut), present in five different yards, and *Malus* spp. (apple and crab-apple), present in seven yards.

Trees ranged in diameter from 2.5 to 85.0 cm dbh. Frequency distributions of stem size classes showed important differences among EZ types (Fig. 2). The old field zone type had only 30 stems/ha in sizes < 5 cm dbh and no trees larger than 35 cm dbh, whereas gardens had 138 tree stems/ha in sizes < 5 cm dbh and some individual trees recorded up to the 60–65 cm size class. The dense woody zone type had the highest frequency of small trees, with 810 stems/ha in sizes < 5 cm dbh. The dense woody zone had > 100 stems/ha in each 5-cm size class up to

Table 2. Carbon pools per unit area in (a) vegetation and (b) litter and soil components for each type of ecological zone (EZ).

	Carbon pools in vegetation within each ecological zone (g C/m^2)		
(a) Ecological zone	Trees	Shrubs	Herbaceous
Turfgrass with sparse woody vegetation	6163^a (1445)	10^a (7)	80^a (12)
Dense woody vegetation	13910^b (3170)	235^{ab} (224)	41^a (8)
Old field	735^{ac} (649)	394^b (252)	338^b (93)
Turfgrass	4^c (4)	0^a (0)	148^c (30)
Gardens	2514^{ac} (1766)	71^{ab} (20)	n.d.

	Carbon pools in litter and soil within each ecological zone (g C m^{-2})				
(b) Ecological zone	Oi horizon	Downed woody debris	Oe and surface mineral soil (0–15 cm)	Deep mineral soil (50–65 cm)	Oe and modeled whole mineral soil (0–100 cm)
Turfgrass with sparse woody vegetation	125^a (14)	2^a (2)	3352^a (278)	1653^a (351)	12850^a (1757)
Dense woody vegetation	621^{ab} (209)	183^b (56)	4397^a (699)	2433^a (930)	20410^a (6160)
Old field	320^a (132)	^{24ab} (24)	3420^a (475)	1884^a (927)	13890^a (4626)
Turfgrass	108^a (12)	0^a (0)	3298^a (256)	1510^a (251)	12640^a (1474)
Gardens	1146^b (254)	71^{ab} (71)	4046^a (475)	1969^a (207)	16190^a (1633)

Notes: Vegetation pools include above- and belowground (root) C. Statistics are calculated across the entire study for each EZ, listed as mean (standard error). Within a carbon pool, values followed by the same superscripted lowercase letters are not significantly different ($P < 0.05$), n.d. = not determined.

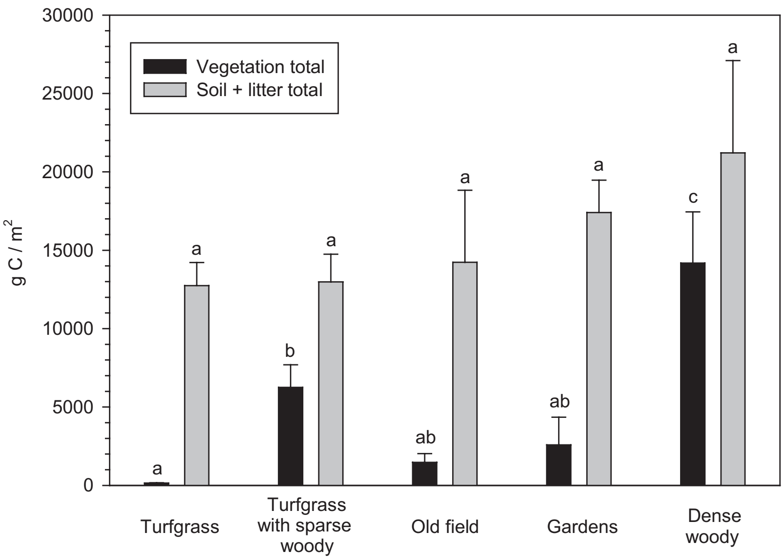

Fig. 1. Vegetation summed C pools (tree, shrub, and herbaceous) and soil summed C pools (litter, downed woody debris, and modeled mineral soil 0–100 cm depth), expressed per unit area (g C/m^2), by ecological zone (EZ) type. Error bars indicate standard errors. Within vegetation and soil totals separately, EZs labeled with the same lowercase letter are not significantly different ($P < 0.05$).

30 cm, but no trees larger than 50 cm dbh. Turfgrass with sparse woody had much fewer tree stems overall (< 100 stems/ha in all size classes) than dense woody zones, but contained trees in several size classes larger than 50 cm, i.e., larger than trees recorded in dense woody zones. The species of these largest trees in turfgrass with sparse woody were *Liriodendron tulipifera*, *Acer saccharinum*, *Acer platanoides*, *Quercus rubra, and J. nigra.*

High numbers of shrubs were recorded, particularly in dense woody, old field, and garden zone types (453, 347, and 296 shrub stems/ha, respectively, in sizes < 5 cm dbh; Fig. 3). The most frequently encountered were (in descending order) *Syringa vulgaris*, *Lonicera* spp., *Picea* spp., and *Thuja occidentalis.* Shrubs often had multiple stems (range one to 11, median three). Infrequent but worth noting was the presence of *Elaeagnus umbellata* (autumn olive, an invasive species and N-fixer). Among a cluster of multiple stems that formed a shrub, no median stem diameter greater than 10 cm was recorded overall, and no median stem diameter greater than 5 cm was present in the turfgrass with woody or garden zone types (Fig. 3).

Soils

Soils had 14% coarse fragments (>2 mm). Bulk densities (oven-dry, coarse fragments included) were 0.85 g/cm^3 for surface soil cores and 1.02 g/cm^3 for deep soil cores. Average values of soil pH were neutral, 6.94 in surface soil cores and 7.08 in deep cores. Mineral soil organic C concentrations were 3.49% ± 0.24% (mean ± standard error [SE]) in surface cores (0–15 cm depth, which included the thin Oe horizon) and 1.41% ± 0.12% in deep cores (50–65 cm depth). In seven of our 80 transects (9%), soil organic C concentration was greater in the deep soil core than the surface soil core. This indicates prior soil disturbance that mixed soils down to 65 cm depth, buried pre-existing surface soil horizons, or that fill was placed on top of existing soil, most likely during residential development; none of these cases occurred in rural lots.

Soil and litter showed few significant differences in C stocks among EZ types (Table 2b). The garden zone type stored more C in litter than other zone types, due to the presence of human-added mulch, while the dense woody zone stored more C in downed woody debris than other zone types. Total litter + modeled soil C to 1 m showed no significant differences among EZ types (Fig. 1). Across the study, pools of mineral soil organic C (surface soil Oe + 0–15 cm, subsoil core 50–65 cm, and modeled total 0–100 cm) also showed no correlations with physiographic variables (WHC, texture class, surficial geology, and TWI).

Landscape-scale ecological zones and C pools

In mapping our ground-based EZ types onto Anderson-based land-cover (LC) categories assessed through aerial photointerpretation, our dense woody mapped mainly onto the LC category tree cover, followed by the LC category open natural (Appendix S1: Table S1). Our turfgrass mapped primarily onto maintained, while our turfgrass with sparse woody mapped onto a mixture of tree cover and maintained. No areas in the yards we studied were identified as crop through aerial photointerpretation (Appendix S1: Table S1). The traditional impervious areas identified through aerial photointerpretation mapped only 69% onto impervious areas identified

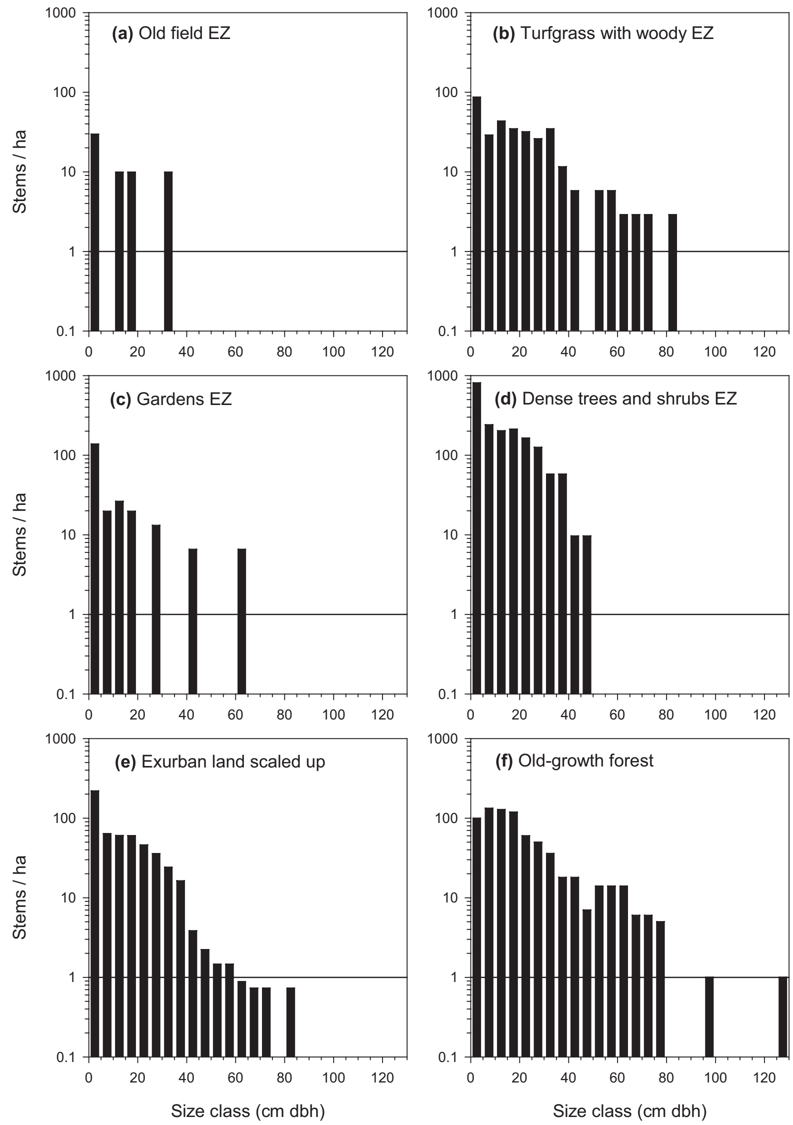

Fig. 2. Frequency distributions of trees in size classes in 5-cm increments. Distributions are shown within (a–d) ecological zone (EZ) types; (e) scaled up to exurban residential land overall; and (f) in a mature northern hardwood forest stand in Michigan, USA (Rutkowski and Stottlemeyer 1993) for comparison. Scale on y-axis is logarithmic, dbh = diameter at breast height (1.4 m).

on the ground, while we identified the remainder as a mixture of turfgrass with sparse woody, dense woody, and gardens.

In our upscaled results, turfgrass with sparse woody was consistently one of the dominant EZ types in all types of exurban neighborhoods (Table 3). Other EZ types varied. Turfgrass cover was greatest in country subdivisions and lowest in remnant subdivisions and rural lots, while dense woody showed the opposite pattern: greatest in remnant subdivisions and rural lots, lowest in country subdivisions (Table 3). Horticultural subdivisions contained the highest areas of old field and showed the most evenly balanced distribution of area across all types of EZs.

Country subdivisions held less C in total vegetation than did either remnant subdivisions or rural lots (Appendix S1: Fig. S1), driven mainly by lower frequencies of trees (data not shown). Similarly, litter and modeled soil C to 1 m, as well as total ecosystem C, were significantly lower in country subdivisions than both remnant subdivisions and rural lots (Appendix S1: Fig. S1). Differences in impervious cover were important contributors to differences in upscaled soil C; country subdivisions had the highest impervious cover at 25.7%, while rural lots had the lowest at 8.3%. Country subdivisions also had low dense woody areas.

The pre-residential land use histories of our individual parcels were 55% cropland, 18% pasture or old field, 18% continuous tree cover, and 9% tree farm or orchard. There were no significant differences by land use history in C storage in surface soil cores, deep soil cores, litter

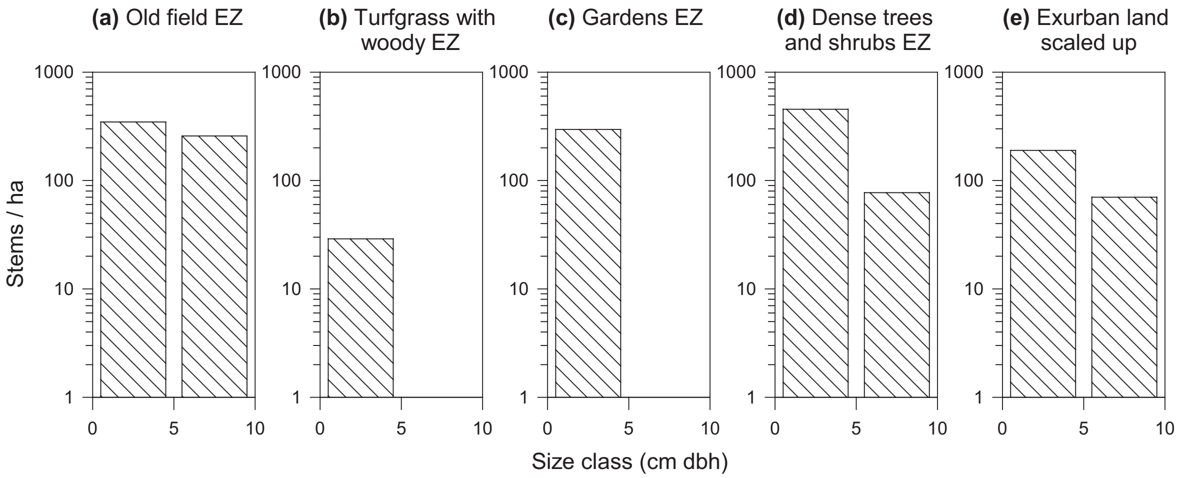

FIG. 3. Frequency distributions of shrubs in size classes (a–d) within ecological zone (EZ) types, and (e) scaled up to exurban residential land overall; *y*-axis is logarithmic.

and soil totals (including modeled mineral soil to 1 m), vegetation C totals, or ecosystem C totals (Appendix S1: Tables S2 and S3). Regressions of upscaled neighborhood C pools against time since development showed rising trends in C storage for vegetation and for ecosystem totals, but were not significant for soil C totals alone (Appendix S1: Fig. S2). To further scale up C storage spatially, we used the proportional areas of the four neighborhood types across southeastern Michigan: 8.1% country, 25.2% horticultural, and 42.2% remnant subdivisions, and 24.6% rural lots (An et al. 2011). We used these to calculate area-weighted values of C in vegetation and soil pools for overall exurban residential land in southeastern Michigan, 19000 ± 1550 g C/m^2 (Table 4).

Discussion

C storage in exurban land compared to forests

In southeastern Michigan, temperate forests were the dominant vegetation prior to European settlement, logging, and clearing for agriculture. The exurban expansion of residential land over the past half-century has occurred on land that was in agriculture or remnants of second-growth forest (Brown et al. 2005, An et al. 2011), including the parcels studied here (Appendix S1: Table S2). Exurban expansion in recent decades has been accompanied with a general increase in tree cover (Zhao et al. 2007, An et al. 2011). Our scaled-up estimates of exurban residential ecosystem C storage were much lower than that of mature forest stands, but in some respects approaching that of regionally averaged forests (Table 4).

Our patches of dense woody vegetation stored C in trees (13910 g C/m^2) at levels that approached, but were 17% below, the average for some mature northern hardwood forests in our biome (Table 4). The frequencies and size-class distributions of trees and shrubs in exurban yards were very different from mature forests. Dense woody zones in exurban yards had tree size distributions highly skewed toward smaller diameters (Fig. 2). Shrubs were virtually absent from old-growth forest (Rutkowski and Stottlemeyer 1993), but present at more than 500 stems/ha in our dense woody zone type and more than 250 stems/ha in exurban yards overall (Fig. 3d,e). However, shrubs stored substantially less C than trees (Table 2a). Interestingly, when scaled up to exurban neighborhoods, the tree size-class distributions were similar to that of mature forest in the size classes < 40 cm

TABLE 3. Average areas of ecological zone (EZ) types by neighborhood type, expressed as a percentage of total area of residential neighborhoods at the landscape scale.

Neighborhood type	Ecological zone type						
	Turfgrass	Turfgrass with sparse woody	Old field	Gardens	Dense trees and shrubs	Impervious	Water
Country	24.4%	24.5%	9.9%	3.6%	11.3%	25.7%	0.6%
Horticultural	17.6	22.7	24.4	2.5	17.1	13.8	2.0
Remnant	11.6	28.4	15.0	2.3	28.3	13.1	1.3
Rural lots	11.0	26.9	22.3	1.9	28.2	8.3	1.5

Note: Within a neighborhood type, average areas sum to 100% (within rounding).

Table 4. Carbon pools (g C/m^2) in exurban residential land compared with other ecosystems and landscape C pools from the literature.

Ecosystem type	O hor.	Piles	DWD	Min. soil	Soil total	Herb	Shrub	Trees	Veg. total	Ecosystem Total	Ref
This study											
Exurban residential land in southeastern Michigan, USA	281 ± 50	71 ± 6	49 ± 12	13 400 ± 1 290	13 800 ± 1 290	119 ± 18	138 ± 65	4 980 ± 846	5 240 ± 865	19 000 ± 1 550	1
Mature forest sites											
Mature northern hardwood forest, New Hampshire, USA	2 970	-	468	12 800	16 200	-	-	12 800	12 800	29 100	2
Mature northern hardwood forest, northern Michigan	450	-	-	-	-	1.4	0.2	16 600	16 600	-	3
Mature sugar maple forests, northern Michigan	614	-	-	8 810	9 420	-	-	20 600	20 600	30 000	4
Regional average forests											
Pre-settlement temperate deciduous forest in Wisconsin, USA	-	-	-	-	-	-	-	500–10 000	500–10 000	-	5
Pre-settlement savanna in Wisconsin	-	-	-	-	-	-	-	<500	<500	-	5
Forest inventory, United States north central region average	900	-	1 250	10 900	13 050	-	200	5 850	6 050	19 100	6
Forest inventory, Lake states average	1 700	-	-	12 400	14 100	-	-	8 140	8 140	22 200	7
Agricultural land											
Sites in Ohio, USA and Ontario, Canada	-	-	-	8 450	8 450	-	-	-	-	-	8
Average cropland in Michigan	-	-	-	6 060	6 060	-	-	-	-	-	9

Notes: Carbon pools included surface organic horizon of soil including fine litter (O hor.), compost piles and woodpiles (a pool that is unique to residential land; piles), downed woody debris (DWD), mineral soil organic C (min), total soil (soil), herbaceous vegetation (herb), total vegetation (veg), and ecosystem total (total). Soil depths are to 1 m unless otherwise noted. Vegetation C includes above- and belowground tissues unless otherwise noted. Shrub and tree categories generally exclude stems < 2.5 cm dbh (diameter at breast height, 1.4 m). All values rounded to three significant digits; total soil, vegetation, and ecosystem sums may differ from individual pools due to rounding. *Sources:* (1) This study. An area-weighted average for exurban residential land was calculated using areal proportions of neighborhood types in the study region (An et al. 2011). Note that impervious areas are included in these C pools scaled to the landscape. Ecosystem total includes piles (woodpiles and compost piles) in addition to soil and vegetation totals. For O horizon, only Oi is included. Mineral soil includes Oe horizon (where present). Uncertainties are upscaled standard deviations from a formal analysis of uncertainty propagation. (2) Fahey et al. (2005); Johnson et al. (1991) provides detail on soil sampling methods. Mineral soils sampled to bedrock or start of C horizon. Trees category here includes living (12 000 g C/m^2), standing dead (657 g C/m^2), and dead roots (188 g C/m^2). (3) Rutkowski and Stottlemeyer (1993). Vegetation C included aboveground only (13 300 g C/m^2), multiplied here by 1.25 to estimate above- and belowground C (roots). (4) Pregitzer et al. (2008). Average of four control sites. Vegetation C included aboveground only (16 500 g C/m^2), multiplied here by 1.25 to estimate above- and belowground C (roots). Mineral soil depth 70 cm. (5) Rhemtulla et al. (2009). Vegetation C pools include aboveground woody vegetation only. Some small areas of pre-settlement forest in Wisconsin had vegetation C > 10 000 g C/m^2. (6) Turner et al. (1995). (7) Grigal and Ohmann (1992). (8) Paul et al. (2003), average of three sites in row-crop agriculture. Includes mineral soil horizons only. (9) Mitchell et al. (1997). Results of a database meta-analysis of croplands in Michigan; soil depths vary among sites and can be up to 2 m.

dbh. The major difference from mature forest was that in exurban yards many fewer trees were present at sizes greater than 40 cm dbh (Fig. 2e,f).

Downed woody debris (DWD) stored much less C in exurban land than in typical forests. In northern hardwood forests in the Great Lakes region, Lorimer and Goodburn (1998) reported average C pool sizes in DWD of 1435 g C/m^2 in old-growth forest, 735 g C/m^2 in select-harvested forest, and 300 g C/m^2 in 65–75 year old second-growth forest. Forest inventory data for the United States north central region reported a regionally averaged pool size of 1250 g C/m^2. By comparison, exurban C pools in DWD were quite small even in the dense woody zones: 183 g C/m^2 (Table 2b). Judging by the tree size-class distributions we found, these small DWD pools may reflect the relatively young age of these forest patches. As the trees in dense woody zones age and undergo self-thinning, unless trees are removed, a pulse of DWD is likely to occur (Harmon et al. 1986, Currie and Nadelhoffer 2002). Averaged over the residential landscape, pools of DWD were smaller still at 49 g C/m^2. A slightly greater C pool, 71 g C/m^2, was present in woodpiles and compost piles (the latter including foliage and grass clippings), unique to the residential landscape.

Characterizing heterogeneity in human-dominated residential land

As other ecologists working in human-dominated areas have noted (Cadenasso et al. 2007), Anderson-based LC categories were insufficient to describe the multi-scale heterogeneity. Our ecological zone (EZ) approach allowed us to characterize highly heterogeneous vegetation associations at the sub-parcel scale, while defining zone types that could be re-used across different types of neighborhoods. It also allowed us to work quickly, which encouraged more landowners to participate. Because our colleagues had conducted a resident behavioral interview (Nassauer et al. 2014) beforehand, we were able to speed the process of dividing each yard into EZ types and to better define and interpret those zone types in terms of human preferences and practices.

Our EZ approach had similarities and differences to other recent approaches. The ecotope approach (Ellis et al. 2000, 2006) was similar to ours in that it identified sub-parcel scale landscape components that were reused systematically across the landscape. In its early development, Ellis et al. (2000) combined aerial photointerpretation with local knowledge, ground observations, and household surveys, as did our methods development. The ecotope approach addressed entire 1-km^2 grid cells and included more physiographic information than our EZs. The HERCULES approach (Cadenasso et al. 2007) sought to quantify heterogeneity in human settlements from urban to exurban, with a greater emphasis on distinguishing types of impervious cover for use in densely populated areas. It was based completely on aerial photointerpretation, with no field observations or household interviews. It defined six categories of cover (two building types, two surface types, and two vegetation categories), with discrete ranges in each category. Landscape patches arose bottom-up from differences in any category. This approach could be systematically applied to large areas and upscale directly from parcels to landscapes, but it introduces uncertainty in the use of categorical ranges of cover.

Fissore et al. (2012) conducted fieldwork on randomly selected residential parcels in the Minneapolis (Minnesota, USA) area with homeowner permission. On each large parcel (> 0.1 ha) comparable to ours, they established five 8 m diameter plots at random without regard to the type of vegetation and scaled their tree measurement data directly to the parcel. Our approach first divided parcels into a set of zones that differed in vegetation, soil, and management and that were reused from one neighborhood to the next, similar to the ecotope approach (Ellis et al. 2006). This enabled us to discover differences in vegetation C storage between dense woody and turfgrass with woody categories, which might have been more difficult to discover if we had used random plots. Knapp et al. (2012), also working in the Minneapolis area, used landscape-level data to assign housing-density values to each yard, somewhat similar to our use of neighborhood typologies. Where yards were too large to identify all vegetation, Knapp et al. (2012) established transects (2 m wide) in either lawns or woodlots. These transects were similar to our EZ plots and the lawns and woodlots were similar to our zones, but our approach identified a greater variety of ecological zones.

The identification of turfgrass with sparse woody vegetation as a distinct zone type was a success that came out of our approach. It occurred in more parcels and covered more area overall than dense woody zones (Table 1). It contained far fewer trees and shrubs in small size classes than did dense woody, but contained occasional large trees (>50 cm dbh), absent from dense woody zones (Fig. 2). Downed woody debris was virtually absent in turfgrass with sparse woody; in this zone type, residents removed fallen trees and large branches to compost piles or burned them as firewood or yard waste.

Uncertainty in measuring and upscaling C pools

Several sources of uncertainty entered into our upscaling methods for soil and vegetation C. In the EZ plot approach, individual large trees could be included or excluded depending on the random placement of a transect. Our modeling of mineral soil C to 1 m depth also generated uncertainty. We compared our exponential model of soil C decline with depth against low and high assumptions based on linear extrapolations from our surface and subsoil cores. In all cases, the exponential model fell between the high and low linear estimates; the average absolute difference was 18%.

Choices made in site selection contributed to uncertainty. We excluded neighborhoods in topographically low-lying areas or with clay-rich soils to avoid confounding the analysis of C pools in different neighborhood types. The parcels we thus avoided could hold higher mineral soil C. Another uncertainty lies in soil organic C beneath impervious surfaces. The value we used from the literature, 3300 g C/m^2 (Pouyat et al. 2006), was only 16–26% of the average values we measured in other EZ types (Table 2b). Recent literature reports a wide range of variation in soil C beneath impervious surfaces (Edmondson et al. 2012, Raciti et al. 2012, Zong-Qiang et al. 2014). Future research could include the effects on mineral soil C of developer practices used in grading the land and in construction of structures, basements, and road beds for roads and driveways.

Assigning discrete categories of prior land use generated uncertainty. Our review of historical aerial photographs revealed that some parcels had multiple land use transitions, e.g., row crops, followed by old field, then shrub cover before being converted to residential land. Our regressions of upscaled neighborhood C pools against time were also highly uncertain because our set of sites was not designed as a chronosequence (Yanai et al. 2003). Differing amounts of tree cover were present in each neighborhood at the time of conversion, particularly among different neighborhood types. Developer practices may also have changed over time, affecting parcel sizes and the amount of impervious area (house sizes, driveway sizes, and roads), which could strongly affect patterns of C storage (Robinson 2012). The lack of significant differences in mineral soil C pools in different EZ types may also be a result of differences in neighborhood vegetation or age and possibly the need for decades or more for soil C stocks to show a significant difference in C storage following vegetation change. More explicit attention to temporal changes in soil C could be an area for future research.

Effects of human choices on C storage

The largest trees we encountered (>50 cm dbh) are too large to have been planted by residents or developers and thus predate the conversion to residential land. Agricultural land in southeastern Michigan has scattered trees outside of woodlots: trees occur along roads and in windbreaks, they surround farmhouses and farm buildings and are left as shade trees for animals. When agricultural land is converted to residential these legacy trees sometimes remain, depending upon design and development choices, and become part of exurban yards. The largest trees we observed were found in the turfgrass with sparse woody zone type, indicating that developers and residents have allowed these large trees to remain in a maintained area or "zone of care" (Nassauer et al. 2014, Visscher et al. 2014) within the parcels. Our prior work has shown that tree cover has increased in exurban land of southeastern Michigan since conversion to residential land (Zhao et al. 2007, An et al. 2011). Our current finding of a positive trend in tree C over time since development (Appendix S1: Fig. S2) is corroborated by a land-cover change analysis which determined that aboveground C storage increased over time in exurban parcels in this region (Huang et al. 2014).

The differences in size-class frequency distributions between our results for exurban land and mature forests (Fig. 2) indicate that there is great potential for additional carbon storage in the exurban landscape if resident preferences and ecological conditions allow trees in exurban yards to grow larger. In an unmanaged forest undergoing succession from oak–hickory to a sugar maple-dominated forest in Indiana, USA, as the forest aged and the overall aboveground biomass increased from 7700 to 10550 g C/m^2 and the bulk of the woody biomass went from being in the 45–55 cm dbh size classes in early succession to the 75–95 cm dbh size classes in later succession (Spetich and Parker 1998). Among forest ecosystem C pools, if trees are not harvested, tree growth is likely to be the most rapidly changing component of a C storage trajectory (Fahey et al. 2010).

Temporal changes in tree C storage in regional forest land are difficult to detect, partly because regional forests are highly heterogeneous, but also because many forests are actively managed and harvested. Based on 185 FIA (Forest Inventory and Analysis) plots in the United States north central region over the most recent 5-year period, Woodall (2010) found slight decreases in living wood, standing dead, and downed CWD totaling −38 g C·m^{-2}·yr^{-1}. This is a small change over time and was not significantly different from zero. In contrast, in the carbon accounting study conducted by Fissore et al. (2012) for residential land in Minnesota, accumulation of C in trees was the main change in C storage over time, amounting to 189 g C·m^{-2}·yr^{-1}. Thus, if trees in the exurban environment are allowed to continue to grow or, if additional tree planting occurs as is typical in this setting (Nassauer et al. 2014, Visscher et al. 2014), the woody vegetation in exurban land could sequester more C over time than managed, secondary forests in the region.

Turfgrass with sparse woody is a vegetation association that is an entirely human construction; it would not exist in its present form without human design and management. In the way that it combines trees and grasses it is like a savanna. Some savanna existed in the pre-settlement vegetation of the upper Midwest, however our turfgrass with sparse woody zone contained 10-fold greater C storage per unit area in woody vegetation than did the pre-settlement savanna of Wisconsin, USA (Rhemtulla et al. 2009, Tables 2a, 4). Researchers studying C fluxes in residential yards in the Minneapolis–St Paul, Minnesota metropolitan area also found that the mixture of grasses and trees in the landscape was key to determining C storage (Fissore et al. 2012). Using a C accounting model and results from a household survey, Fissore et al. (2012) found the major sources of C to residential yards was grass growth, wood production in trees, and leaf litter from trees. A small proportion of yards (13%) contained areas dense with trees, similar to our identification of 22% of parcel areas as dense woody vegetation.

Other choices made by residential landowners can potentially affect soil C storage. In our study area, foliar litter from coniferous evergreen trees is typically not removed and in some areas deciduous leaf litter is not removed; in some cases the mulch-mowing of foliar litter, a common practice, may contribute to C storage (Visscher et al. 2014). In their study of urban residential land in Minnesota, Fissore et al. (2012) estimated that soil C was slowly increasing over time in residential yards (25 g $C \cdot m^{-2} \cdot yr^{-1}$), due to C inputs from grass litter and tree foliar litter. In a study of urban residential yards in Maryland, Raciti et al. (2011) found that a chronosequence regression for soil C against housing age revealed a significant increase over time in surface soil C (humus layer and mineral soil to 10 cm depth). In our region, developers and residents plant or cultivate turfgrass or leave some areas in old-field vegetation that includes grasses. Grasses typically allocate high proportions of NPP belowground (Bonan et al. 2003). Grasses are not the natural vegetation in this region because there is ample moisture for trees (Kuchler 1964), but human preferences and activities maintain large areas of grasses, which are then highly productive with the high moisture, N-rich soil, and warm growing season in southeastern Michigan (Milesi et al. 2005). Irrigation and fertilization of yards may contribute to grass production (Hutchins 2010, Fissore et al. 2012, Visscher et al. 2014), and potentially to shrub or tree NPP. Nassauer et al. (2014) found that in 17 of our 26 parcels, residents applied fertilizer in at least part of the parcel. Ecological studies have shown that nitrogen addition increases NPP in nitrogen-limited vegetation including both trees and grasses across a wide range of biomes (LeBauer and Treseder 2008). The likely effects of these human choices and activities on landscape C storage illustrate that C storage in the exurban landscape arises as a result of coupled human–natural processes.

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Supporting Information

Additional supporting information may be found in the online version of this article at http://onlinelibrary.wiley.com/doi/10.1890/15-0817/suppinfo

Data Availability

Data associated with this paper have been deposited in Dryad: http://dx.doi.org/10.5061/dryad.7g6v3