Urban soil carbon and nitrogen converge at a continental scale

Tara L. E. Trammell [ID],[1,13] Diane E. Pataki,[2] Richard V. Pouyat,[3] Peter M. Groffman,[4,5] Carl Rosier,[1] Neil Bettez,[5] Jeannine Cavender-Bares [ID],[6] Morgan J. Grove,[7] Sharon J. Hall [ID],[8] James Heffernan,[9] Sarah E. Hobbie,[6] Jennifer L. Morse,[10] Christopher Neill,[11] and Meredith Steele[12]

[1]*Department of Plant and Soil Sciences, University of Delaware, Newark, Delaware 19716 USA*
[2]*School of Biological Sciences, University of Utah, Salt Lake City, Utah 84112 USA*
[3]*Emeritus USDA Forest Service, NRS, Affiliate Faculty Department of Plant and Soil Sciences, University of Delaware, Newark, Delaware 19716 USA*
[4]*Advanced Science Research Center at the Graduate Center, City University of New York, New York 10031 USA*
[5]*Cary Institute of Ecosystem Studies, Millbrook, New York 12545 USA*
[6]*Department of Ecology, Evolution and Behavior, University of Minnesota, St. Paul, Minnesota 55108 USA*
[7]*USDA Forest Service, Baltimore Ecosystem Study, University of Maryland, Baltimore County, Baltimore, Maryland 21227 USA*
[8]*School of Life Sciences, Arizona State University, Tempe, Arizona 85287 USA*
[9]*Nicholas School of the Environment, Duke University, Durham, North Carolina 27708 USA*
[10]*Department of Environmental Science and Management, Portland State University, Portland, Oregon 97207 USA*
[11]*Woods Hole Research Center, 149 Woods Hole Road, Falmouth, Massachusetts 02540 USA*
[12]*School of Plant and Environmental Sciences, Virginia Tech, Blacksburg, Virginia 24061 USA*

Abstract. In urban areas, anthropogenic drivers of ecosystem structure and function are thought to predominate over larger-scale biophysical drivers. Residential yards are influenced by individual homeowner preferences and actions, and these factors are hypothesized to converge yard structure across broad scales. We examined soil total C and total $\delta^{13}C$, organic C and organic $\delta^{13}C$, total N, and $\delta^{15}N$ in residential yards and corresponding reference ecosystems in six cities across the United States that span major climates and ecological biomes (Baltimore, Maryland; Boston, Massachusetts; Los Angeles, California; Miami, Florida; Minneapolis-St. Paul, Minnesota; and Phoenix, Arizona). Across the cities, we found soil C and N concentrations and soil $\delta^{15}N$ were less variable in residential yards compared to reference sites supporting the hypothesis that soil C, N, and $\delta^{15}N$ converge across these cities. Increases in organic soil C, soil N, and soil $\delta^{15}N$ across urban, suburban, and rural residential yards in several cities supported the hypothesis that soils responded similarly to altered resource inputs across cities, contributing to convergence of soil C and N in yards compared to natural systems. Soil C and N dynamics in residential yards showed evidence of increasing C and N inputs to urban soils or dampened decomposition rates over time that are influenced by climate and/or housing age across the cities. In the warmest cities (Los Angeles, Miami, Phoenix), greater organic soil C and higher soil $\delta^{13}C$ in yards compared to reference sites reflected the greater proportion of C_4 plants in these yards. In the two warm arid cities (Los Angeles, Phoenix), total soil $\delta^{13}C$ increased and organic soil $\delta^{13}C$ decreased with increasing home age indicating greater inorganic C in the yards around newer homes. In general, soil organic C and $\delta^{13}C$, soil N, and soil $\delta^{15}N$ increased with increasing home age suggesting increased soil C and N cycling rates and associated ^{12}C and ^{14}N losses over time control yard soil C and N dynamics. This study provides evidence that conversion of native reference ecosystems to residential areas results in convergence of soil C and N at a continental scale. The mechanisms underlying these effects are complex and vary spatially and temporally.

Key words: natural abundance carbon stable isotopes; natural abundance nitrogen stable isotopes; residential yard management; soil C cycling; soil N cycling; urban residential yards.

Introduction

Urban soils provide numerous ecosystem services to city residents across the world through soil ecosystem functions including nutrient cycling, carbon sequestration, and water purification. Soil, the "brown

infrastructure" of cities (Pouyat and Trammell 2019), is the support structure and nutrient source for plant growth, which, in turn, is the green infrastructure used to manage and restore urban water quantity and quality, among other benefits. A large proportion of open green space (i.e., residential areas) in urban areas consists of lawns (52–80% lawn cover; Richards et al. 1984), which are an extensive land cover in cities and important in the lives of urban residents (Milesi et al. 2005, Groffman et al. 2009, Ignatieva et al. 2015). Multiple local-scale anthropogenic drivers such as agricultural history and lawn management and larger biophysical drivers such as regional climate and the urban heat island interact to affect residential yards and their underlying soils.

A prevailing urban ecological concept suggests that human drivers can override biophysical drivers in controlling ecosystem structure and function (Kaye et al. 2006, Alberti 2008). Thus, ecological processes in different urban environments may converge relative to nearby natural ecosystems if urban residents attempt to achieve similar environmental and ecological conditions in different regions ("urban ecosystem convergence hypothesis"; Pouyat et al. 2003). This convergence underlies the idea that conversion of native or agricultural ecosystems to residential ecosystems results in ecological homogenization at the continental scale (Groffman et al. 2014). Previous research on soil chemical properties in five metropolitan areas around the world found evidence for convergence in organic carbon (C) and total nitrogen (N), soil characteristics affected by anthropogenic influences (e.g., lawn management), whereas soil properties associated with parent material (e.g., soil P and K) were more variable across cities (Pouyat et al. 2015). This differentiation in convergence of soil characteristics indicates that soil elements associated with biogenic processes may respond more to human influences than soil characteristics more closely related to natural soil-forming factors.

Analytical approaches that decipher human impacts on urban soil C and N cycling in cities across biomes can increase our understanding of the relative impacts of human vs. natural drivers on soil processes. The amount of atmospheric CO_2 and reactive N in urban environments has greatly increased due to human activities (Idso et al. 1998, 2001, Lovett et al. 2000, Rao et al. 2014). The natural abundance of stable C ($\delta^{13}C$) and N ($\delta^{15}N$) isotopes in soils can be useful indicators of C and N inputs and processes in terrestrial ecosystems. Elevated concentrations of fossil fuel-derived CO_2 in urban environments have the potential to lower plant $\delta^{13}C$ in urban ecosystems (e.g., Lichtfouse et al. 2003), because fossil fuel-derived CO_2 lowers $\delta^{13}C$ in background atmospheric CO_2 (Pataki et al. 2005a, b), which in turn lowers $\delta^{13}C$ in organic matter (OM) entering soil C pools (Ehleringer et al. 2000). Soil $\delta^{15}N$ can act as a tracer of N sources when $\delta^{15}N$ sources are isotopically distinct (Robinson 2001), such as fertilizer, which has an isotopic $\delta^{15}N$ signature of 0‰ compared to wet deposition and NO_x

sources (Kendall et al. 2007, Bijoor et al. 2008). Additionally, microbial processes in soils influence the $\delta^{13}C$ and $\delta^{15}N$ composition. Discrimination against ^{15}N occurs during the majority of soil N processes, resulting in products depleted in ^{15}N relative to substrates. For example, NO_3^- is depleted in ^{15}N relative to the substrate NH_4^+ from which it was produced (Högberg 1997, Dawson et al. 2002). Soil decomposers fractionate against ^{13}C during respiration, thus the $\delta^{13}C$ composition of microbial biomass and its by-products are typically more enriched in ^{13}C than the substrates they consume (Boutton 1996). Soil $\delta^{13}C$ is also strongly correlated with leaf litter $\delta^{13}C$ (Balesdent et al. 1993); aboveground vegetation with isotopically distinct $\delta^{13}C$ values can be traced in soil $\delta^{13}C$ and used to evaluate the relative contribution of C_3 vs. C_4 plant biomass inputs to soil (Boutton 1996). C_4 plant species tend to be enriched in ^{13}C relative to C_3 species since there is less ^{13}C discrimination during photosynthesis in C_4 plants (O'Leary 1988).

In this study, we evaluated how urbanization influenced soil C and N pools and isotopic signatures in residential yards relative to nearby natural reference areas in U.S. metropolitan areas. We measured soil total C and total $\delta^{13}C$, organic C and organic $\delta^{13}C$, total N, and $\delta^{15}N$ in yards and paired unmanaged natural reference sites in six cities across the United States (Baltimore, Boston, Los Angeles, Miami, Minneapolis-St. Paul, and Phoenix) that spanned biomes and climatic regions as well as housing age. The overall goal of this study was to assess soil C and N pools and isotopic signatures at a range of spatial and temporal scales. Specifically, we sought to evaluate if soil C, N, $\delta^{13}C$, and $\delta^{15}N$ converged across our selected six cities (i.e., continental scale) or whether soil C and N pools and isotopic signatures were controlled at finer spatial scales by gradients of urbanization within cities. Finally, we used housing age to evaluate temporal effects on soil C and N pools and isotopic signatures within yards.

At the continental scale, we hypothesized that total soil C and soil N would converge across residential yards of six cities as a function of residential yard climate (Hall et al. 2015) and management (Polsky et al. 2014). Specifically, we expected greater soil C and N in arid city yards relative to nearby reference areas due to greater organic matter inputs to the soil (grass) compared to arid ecosystems, whereas mesic city yards would be characterized by lower soil C and N relative to the reference sites due to lower organic matter inputs to the soil in lawns relative to forest ecosystems. We hypothesized that soil $\delta^{15}N$ would converge due to increases in $\delta^{15}N$ from urban sources and management practices that enhance soil N cycling rates and associated ^{14}N losses (Trammell et al. 2016). Alternatively, inorganic fertilization inputs that act to decrease $\delta^{15}N$ may drive soil $\delta^{15}N$ divergence. However, this depends on the timing, amount, and frequency of fertilizer applications, and over the long-term increased soil N via fertilizer additions will lead to ^{15}N enrichment. We did not expect soil $\delta^{13}C$ to converge

because variation of several urbanization factors that either increase (i.e., carbonate inputs or greater C_4 plant-derived biomass) or decrease $\delta^{13}C$ (i.e., plants incorporating C emitted from fossil fuels) organic matter inputs to soil will determine the signature of C pools that turn-over in the soil.

At the city scale, we hypothesized that soil N and $\delta^{15}N$ would increase along rural to urban gradients due to greater N deposition closer to urban centers (Lovett et al. 2000, Fang et al. 2011, Rao et al. 2014). Alternatively, N deposition could be small relative to fertilizer inputs, which might not vary along an urbanization gradient. We expected soil C inputs (i.e., productivity) to be unaffected by the intensity of urbanization. Research suggests that greater soil C in urban areas is attributed to greater fertilizer and water inputs in residential/commercial lawns increasing productivity (Trammell et al. 2017). Given that all of the residential sites in this study had the potential for management inputs (i.e., fertilization, irrigation) to increase soil C, we did not anticipate significant change in the variation of soil C along the urban gradient. This hypothesis disregards potential effects of the intensity of urbanization on decomposition (Carreiro et al. 1999), which could affect soil C and ^{13}C. Temporally, we hypothesized that soils in older parcels (containing homes built prior to 1950; e.g., Trammell et al. 2016) would contain greater soil C and N, as previous research demonstrated increasing soil C and N with increasing housing age (Raciti et al. 2011, Gough and Elliott 2012). Finally, we expected that soil $\delta^{13}C$ and $\delta^{15}N$ would be enriched in soils of older parcels due to increased soil C and N cycling combined with associated ^{12}C and ^{14}N losses that enrich soil $\delta^{13}C$ and $\delta^{15}N$.

METHODS

Study area and experimental design

We sampled six major metropolitan areas across the continental United States: Baltimore, Maryland (BAL); Boston, Massachusetts (BOS); Los Angeles, California (LA); Miami, Florida (MIA); Minneapolis-St. Paul, Minnesota (MSP); and Phoenix, Arizona (PHX). All of the cities had total Metropolitan Statistical Area (MSA) populations greater than 1 million (U.S. Census Bureau 2010; Trammell et al. 2016), and were chosen to represent different ecological biomes (U.S. Geological Survey 2008) and major climatic regions (National Climatic Data Center 2014). In all cities, the experimental design included residential yards ($n = 20$–30 per city) and reference sites that represented the region's natural ecosystem(s).

Reference sites were selected to represent the ecological biome (e.g., topography, edaphic characteristics, dominant native vegetation) for each city (e.g., deciduous forest in Baltimore and Sonoran desert in Phoenix). Reference sites received minimal (e.g., intermittent removal of invasive grass in PHX), if any, current management. All reference sites were located outside of the metropolitan areas,

yet due to the spread of urban development, reference sites overlapped with some exurban residential yards in each city. Residential sites were selected from identical PRIZM (Potential Rating Index for Zipcode Markets; Claritas, *available online*)[14] market classification zones across all cities. PRIZM zones incorporate multiple forms of socioeconomic data, such as housing density, economic status, and life stage (see footnote 14). The experimental design for residential yards in each city incorporated an urban to rural gradient with at least three sites located in urban, suburban, and exurban neighborhoods (Appendix S1: Table S1) characterized mainly by differences in population density (as defined by PRIZM). Residential experimental designs for each city also incorporated local differences, such as distance to the coast (LA), land-use history (agriculture/pasture vs. forest/desert in BAL, BOS, and PHX), underlying soil type (sand vs. rocky limestone ridge in MIA, on or off sandy outwash in MSP) or yard type (xeric vs. lawn, PHX). Data on differences across different yard types (e.g., previous ag vs. forest) are not reported in this paper since there are no significant differences in soil C and N across these yard types, which is similar to our findings for lawn plant N and $\delta^{15}N$ (Trammell et al. 2016). Within each residential yard across the cities, the lawn portion of the yard was the only area sampled for this soil study.

Soil C and N

Soils were collected during peak growing season in each city (i.e., summer 2012 BAL, BOS, MSP, and MIA; spring 2013 LA and PHX). Two 1 m depth soil cores (2 cm diameter) were collected from random locations covered by lawn within front and back yards in all residential sites ($n = 2$ cores per yard). If lawn cover was not present in either the front or back yard (e.g., only impervious surface and planting beds), then both soil cores were collected from the lawn in either the front or back yard. In each reference site, soil cores were collected in two random locations along two 100-m transects ($n = 4$ soil cores per reference site). Prior to soil collection with a JMC Environmental Sub-Soil Probe (JMC Soil Samplers, Newton, Iowa, USA), the surface organic material (e.g., leaf litter, sticks, grass) was pushed aside to collect the underlying soil. Soil cores were divided into four depths (0–10 cm, 10–30 cm, 30–60 cm, and 60–100 cm), sorted to remove rocks and roots, dried at 60°C for ≥ 72 h, and sieved (2 mm). The bottom two soil depths (30–60 cm and 60–100 cm) were not analyzed for soil C and N due to the difficulty in isotopic determination at low concentrations of soil C and N in subsoil. The top two soil depths (0–10 cm and 10–30 cm; hereafter referred to as "surface soil" and "subsurface soil") were analyzed for total C, organic C, total N, $\delta^{13}C$, and $\delta^{15}N$. Prior to C and N elemental and isotope analyses, soil samples were ground to a fine powder using a Retsch

Ball Mixer Mill (MM200, Haan, Germany). Total C was calculated from combustion of non-acid-washed samples and organic C was determined from samples treated with acid to remove inorganic C.

C (%), N (%), $\delta^{13}C$, and $\delta^{15}N$ were measured with a DELTA Plus Isotope Ratio Mass Spectrometer (Finnigan-MAT, Bremen, Germany) interfaced with an elemental analyzer (Model 1110; Carlo Erba, Milan, Italy; Stable Isotope Ratio Facility for Environmental Research [SIRFER] at the University of Utah, Salt Lake City, Utah, USA). All C:N ratios were expressed on a molar basis. Two primary reference materials (PLRM), calibrated against National Institute of Standards and Technology and International Atomic Energy Agency certified reference materials, and one secondary reference material (SLRM, MT soil) were used as internal standards, with precision less than 0.25‰ and 0.65% (organic and total $\delta^{13}C = \pm 0.15$‰, organic and total C = ± 0.6%, $\delta^{15}N = \pm 0.2$‰, and N = ± 0.15%). The stable isotope values were expressed relative to the international standard ($\delta^{13}C$, Vienna-PeeDee Belemnite; $\delta^{15}N$, atmospheric N_2) in the conventional δ-notation:

$$\delta = [(R_{sample}/R_{standard}) - 1] \times 1,000‰$$

where R represents the molar ratio of the heavy to light isotope, which is $^{13}C/^{12}C$ for carbon and $^{15}N/^{14}N$ for nitrogen.

Statistical analyses

The coefficient of variation (CV) was determined for total and organic soil C, total and organic $\delta^{13}C$, total soil N, and soil $\delta^{15}N$ for residential yards and reference sites across the six cities. CV is a useful statistical measure to determine variation in soil variables since the variation is normalized by the mean. Lower CV values suggest convergence in soil characteristics and higher CV values suggest divergence. To assess differences in residential and reference soil C and N, we performed nonparametric statistical tests for analyses with an unbalanced design or when normality was not possible to assess because of low sample sizes within cities or regions (Mann and Whitney 1947, Bissonette 1999). The Bartlett Test for Homogeneity of Variances was used to determine homoscedasticity for all data prior to statistical analyses. Differences between residential and reference soil C and N were assessed using the nonparametric Mann-Whitney rank sum test. Analysis of differences in residential soil C and N across urban density classes were analyzed using one-way analysis of variance (ANOVA) for data that met the assumptions of normality and homoscedasticity, followed by post hoc Tukey HSD tests. The nonparametric Kruskal-Wallis rank sum test followed by post hoc Nemenyi tests (PMCMR R package; Pohlert 2014) were used when the assumptions for ANOVA were not met after log transformation. Regression analysis was used to determine

whether the residence year of construction explained significant variation in soil C and N. The nonparametric Mann-Whitney rank sum test was used to determine significant differences in soil C and N between previous land use (BAL, BOS, and PHX), soil conditions (MIA and MSP), distance to the coast (LA), and yard type (PHX). All tests for significance are reported at the α = 0.05 critical value. All statistical analyses were performed in R version 3.3.2 (R Core Team 2016).

RESULTS

Continental scale: soil C and N

Across the six cities, total soil C, organic soil C, total soil N, and soil $\delta^{15}N$ demonstrated convergence in residential yards compared to reference sites as shown by lower CV values (Fig. 1). In contrast, total soil $\delta^{13}C$ and surface soil organic $\delta^{13}C$ had greater CV values across residential yards compared to reference sites demonstrating divergence in soil $\delta^{13}C$ (Fig. 2).

Surface organic and total soil C were significantly greater in residential yards than reference sites in the warmest cities, LA ($P < 0.001$), MIA ($P = 0.04$), and PHX ($P < 0.05$; organic, two- to threefold increase; total, twofold increase; Fig. 2a, b). Similarly, total soil N was greater in residential surface soils compared to reference sites in all cities except BAL and BOS (1.7–2.5-fold increase, $P < 0.05$; Fig. 2c). In the subsurface soil, total soil C and soil N were significantly greater in residential yards compared to reference sites only in the driest city, PHX ($P = 0.04$, 1.8-fold; Fig. 2e, f).

Organic and total soil $\delta^{13}C$ was significantly greater in residential yards compared to reference sites in the warmest and wettest city, MIA ($P < 0.05$, 1.2–1.7-fold greater), or warmest and driest city, PHX ($P < 0.03$, 1.1–1.4-fold; Fig. 3). In LA, a warm dry city, residential yards had significantly greater total soil $\delta^{13}C$ than reference sites only in the subsurface soil ($P = 0.04$, 1.2-fold; Fig. 3). $\delta^{15}N$ in surface soil N was significantly greater in residential yards compared to reference sites in the cool moist cities (BOS and MSP) and the warm moist city (MIA; $P < 0.01$, 1.4-fold; Fig. 4). Alternatively, $\delta^{15}N$ in subsurface soil was significantly greater in residential yards compared to reference sites in the warmest cities (MIA, $P = 0.02$, 1.9-fold and PHX, $P < 0.10$, 1.6-fold; Fig. 4).

City scale: soil C and N

In residential yards, organic soil C varied significantly along urbanization gradients in five of the six cities, the exception being LA ($P > 0.10$, Fig. 5). In BAL, BOS, and PHX, soil organic C was significantly greater in urban than suburban ($P < 0.05$, 1.4–3.5-fold) and exurban ($P < 0.05$, 1.8–2.7-fold) yards, except for BOS subsurface soils (Fig. 5). In MIA, surface soil organic C was significantly greater in suburban than exurban yards ($P = 0.03$, 2.7-fold; Fig. 4). In MSP, subsurface soil organic C was
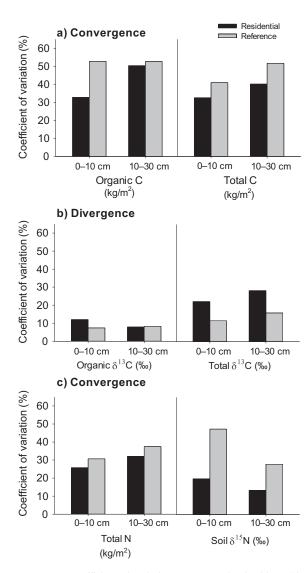
FIG. 1. Coefficient of variation (%) across the six cities exhibits convergence for (a) soil organic and total soil C (kg/m^2) and for (c) soil total N (kg/m^2) and δ^{15}N (‰), yet exhibits divergence for (b) soil organic and total δ^{13}C (‰) in residential yards compared to reference sites.

significantly greater in urban than suburban yards ($P = 0.05$, 1.6-fold; Fig. 5). Total soil C, organic soil δ^{13}C and total soil δ^{13}C did not differ among urban, suburban, and exurban residential yards across the six cities.

Patterns in soil N in residential yards across the urbanization gradient were more consistent than for soil C (Fig. 6). Soil N was significantly greater in urban and suburban than exurban yards in BOS (surface soil, $P < 0.05$, 1.5-fold) and MIA (surface and subsurface soil, $P < 0.001$, 2.4-fold; Fig. 6). In PHX surface ($P < 0.01$, 1.9-fold) and subsurface soil ($P < 0.001$, 2.1-fold) and BAL (surface soil, $P < 0.01$, 1.3-fold), N was significantly greater in urban than suburban and exurban yards (Fig. 6). Soil N was significantly greater in urban than suburban yards in BAL ($P < 0.01$, 1.8-fold)

and MSP (subsurface soils, $P < 0.01$, 1.7-fold). Surface and subsurface soil N did not differ among urban, suburban, and exurban residential yards in LA (Fig. 6).

Soil δ^{15}N was higher in urban than suburban and/or exurban yards in five cities (Fig. 6). Urban soil δ^{15}N was significantly greater compared to suburban and exurban residential soils in BAL (surface and subsurface, $P < 0.01$, 1.8–2.1-fold) BOS (surface, $P < 0.05$, 1.3-fold) and LA (subsurface, $P < 0.05$, 1.3-fold; Fig. 6). Soil δ^{15}N was significantly greater in urban and suburban yards compared to exurban yards in MIA (surface and subsurface, $P < 0.01$, 1.6–2.0-fold), and urban yards were significantly greater in soil δ^{15}N compared to suburban yards in MSP (surface and subsurface, $P < 0.05$, 1.6-fold; Fig. 6). There were no significant differences across the urbanization gradient in soil δ^{15}N in PHX (Fig. 6).

Temporal scale: soil C and N

As expected, soil organic C significantly increased in both surface and subsurface soil with housing age in BAL ($P < 0.01$), BOS ($P < 0.001$), PHX ($P < 0.01$), and LA (only subsurface) residential yards ($P = 0.02$); however, there was no relationship with housing age in MIA or MSP residential yards ($P > 0.10$; Appendix S1: Fig. S1). Similarly, organic soil δ^{13}C increased in both surface and subsurface soils with housing age in BOS ($P < 0.01$), PHX ($P < 0.01$), and LA (only subsurface, $P = 0.01$), and there was no relationship in BAL, MIA, or MSP ($P > 0.10$; Appendix S1: Fig. S1). Like soil organic C, total soil C increased with housing age in both surface and subsurface soil in BOS ($P < 0.05$) and BAL ($P < 0.01$; Appendix S1: Fig. S2). Total soil δ^{13}C followed similar patterns to total soil C in BOS yards ($P < 0.05$), whereas total soil δ^{13}C decreased with increasing housing age in LA (subsurface, $P = 0.02$) and PHX (surface, $P = 0.05$; Appendix S1: Fig. S2).

Also as expected, soil N significantly increased with housing age in both surface and subsurface soils of BAL ($P < 0.02$), BOS ($P = 0.03$), PHX ($P < 0.001$), and LA (only subsurface, $P = 0.01$) residential yards; in contrast, there was no relationship between soil N and housing age in MIA and MSP residential yards (Fig. 7). Soil δ^{15}N had stronger relationships with housing age than soil N in all cities except PHX (Fig. 7). Soil δ^{15}N increased with housing age in both surface and subsurface soils in BAL ($P < 0.01$), BOS ($P < 0.01$), LA ($P < 0.01$), PHX ($P < 0.03$), MSP (only surface, $P = 0.03$), and MIA (only subsurface, $P = 0.017$) residential yards (Fig. 7).

DISCUSSION

In six cities across the continental United States, soil C and N content and soil δ^{15}N were less variable across cities in residential yards compared to reference sites supporting our hypothesis that soil C, N, and δ^{15}N would converge in residential yards and the idea that conversion of native reference ecosystems to residential
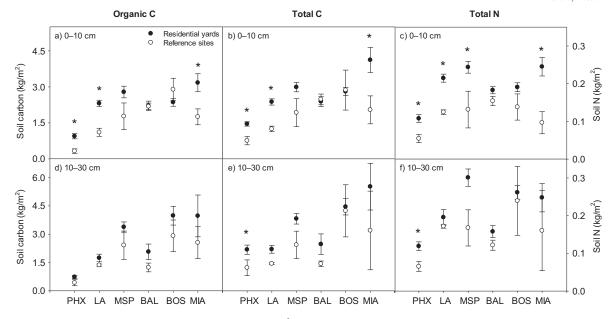
FIG. 2. Mean soil organic C, total C, and total N (kg/m²) in the surface (0–10 cm) and subsurface (10–30 cm) soil layers are shown for residential yards (black) and reference sites (white). Cities are shown in order of increasing mean annual precipitation (MAP), from driest (Phoenix, Arizona; PHX) to wettest (Miami, Florida; MIA). Asterisks represent significant differences ($P \leq$ 0.05) in soil C or N between residential yards and reference sites. Error bars show ±1 SE. BOS, Boston, Massachusetts; BAL, Baltimore, Maryland; LA, Los Angeles, California; MSP, Minneapolis-St. Paul, Minnesota.
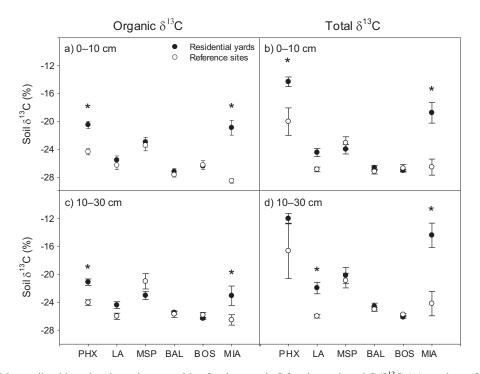

FIG. 3. Mean soil stable carbon isotopic composition for the organic C fraction and total C (δ^{13}C, ‰) are shown for residential yards (black) and reference sites (white) for the surface (0–10 cm) and subsurface (10–30 cm) soil layers. Error bars show ±1 SE. Cities are shown in order of increasing mean annual precipitation (MAP), from driest (Phoenix, Arizona; PHX) to wettest (Miami, Florida; MIA). Asterisks represent significant differences ($P \leq$ 0.05) in soil δ^{13}C between residential yards and reference sites. BOS, Boston, Massachusetts; BAL, Baltimore, Maryland; LA, Los Angeles, California; MSP, Minneapolis-St. Paul, Minnesota.
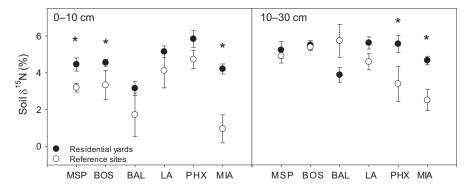
Fig. 4. Mean soil stable nitrogen isotopic composition (δ^{15}N, ‰) in the surface soil (0–10 cm) and subsurface soil (10–30 cm) are shown for residential yards (black) and reference sites (white). Error bars show ±1 SE. Cities are shown in order of increasing mean annual temperature (MAT), from coldest (MSP) to warmest (MIA). Asterisks represent significant differences ($P \leq 0.05$) in soil δ^{15}N between residential yards and reference sites.
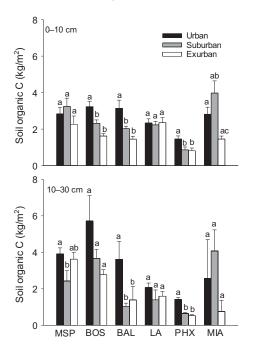

Fig. 5. Mean soil organic C (SOC; kg/m^2) in the surface (0–10 cm) and subsurface (10–30 cm) soil layers are shown for urban (black), suburban (gray), and exurban (white) residential yards. Error bars show +1 SE. Cities are shown in order of increasing mean annual temperature (MAT), from coldest (MSP) to warmest (MIA). Different letters represent significant differences ($P \leq 0.05$) in SOC across urban, suburban, and exurban residential yards within each city.

ecosystems results in ecological homogenization at the continental scale. We also observed increases in soil C and N in residential yards in both arid and mesic cities, suggesting that this conversion increases inputs and/or decreases decomposition of organic material across diverse hydroclimatic conditions. The difference in variability across cities between residential yards and reference ecosystems was greater in surface than subsurface soils, indicating the importance of enhanced soil C and N inputs to surface soils in residential yards (Fig. 1).

Increases in soil organic C, soil N, and soil δ^{15}N across the urbanization gradient in several cities supported the assumption that soil processes respond to similar human management and resource inputs across cities and that this contributed to less variability of soil C and N in yards compared to natural reference ecosystems.

Continental scale: soil C and N

Our finding of greater soil C and N variability across cities in reference sites compared to residential yards supports the idea that soil properties associated with biogenic and anthropogenic factors converge across cities with differing climates and ecological biomes (Pouyat et al. 2015), and that conversion of native reference ecosystems to residential ecosystems results in ecological homogenization at the continental scale (Groffman et al. 2014). In addition to lower variability, organic and total C were greater, although not statistically, in residential yards than in reference sites in all cities, except BAL and BOS surface soil, and were statistically significant in the warm, mesic city (MIA) and in the two arid cities (LA and PHX; Fig. 1). Previous research found greater soil C in residential yards than reference sites due to greater inputs of aboveground biomass in arid cities (Golubiewski 2006), as well as in humid cities (Raciti et al. 2011). While MIA had the greatest mean annual precipitation across our cities, low soil water holding capacity and dry seasons require irrigation for lawn maintenance in MIA (Romero and Dukes 2013) suggesting cities with substantial management inputs have the greatest increase in organic and total soil C. In cities with greater aboveground biomass in reference than residential ecosystems (e.g., forests vs. yards), lawn soils might be expected to have lower soil C than reference soils due to reduced organic matter inputs compared to the native reference ecosystems (Pouyat et al. 2009). High aboveground biomass in reference ecosystems may explain the lack of a significant difference between residential and reference soil C in BAL and BOS where residential yards had similar soil C to reference sites. However, previous work in Baltimore (Raciti et al.
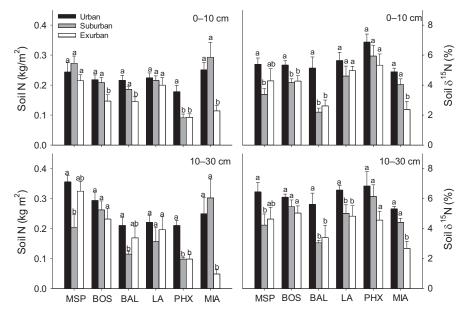
FIG. 6. Mean soil $\delta^{15}N$ (‰) and N (kg/m^2) in the surface (0–10 cm) and subsurface (10–30 cm) soil layers are shown for urban (black), suburban (gray), and exurban (white) residential yards. Error bars show +1 SE. Cities are shown in order of increasing mean annual temperature (MAT), from coldest (MSP) to warmest (MIA). Different letters represent significant differences ($P \leq 0.05$) in soil N and $\delta^{15}N$ across urban, suburban, and exurban residential yards within each city.

2011) found higher soil C in yards than in forests, especially deeper in the soil profile (30–100 cm). These results suggest that evaluations of residential land-use conversion need to consider the effects of management on inputs of both aboveground and belowground organic matter, as well as effects on decomposition of these inputs.

Lawn management inputs, such as N fertilizer and irrigation, are widespread and significant in residential yards (Robbins et al. 2001), and these likely contribute to greater total soil N content in residential yards compared to reference sites. In the surface soil, this pattern was significant in all cities except BAL and BOS (Fig. 1). Lower soil N in yards within these two cities may be due to lower application rate of N fertilizer by homeowners and/or fewer homeowners apply fertilizer to their yards compared to the other cities. The percentage of households that apply fertilizer was lowest in Baltimore (52%) among our six cities, but Boston was right at the average (64%; Groffman et al. 2016). It is also important to note that fertilizer additions are a larger source of N (~100 compared to ~10 kg N·ha^{-1}·yr^{-1} in atmospheric deposition; Aber et al. 1989) and thus are likely to be a strong driver of soil N pools and processes, even if not all yards are fertilized in every year.

Similar to soil N content, soil $\delta^{15}N$ was enriched in residential yards compared to reference sites, although this was only significant in MIA, MSP, and BOS (Fig. 3). Higher soil $\delta^{15}N$ is associated with greater N cycling rates and associated N losses that produce an enriched soil N pool, because ^{14}N is preferentially lost as nitrate and gaseous emissions (Dawson et al. 2002, Pardo et al. 2006, 2007). Clearly, increased inputs and greater rates of cycling and loss result in increases in soil $\delta^{15}N$ in residential compared to reference sites, but these effects vary with the nature and extent of both the inputs and rates of cycling and loss.

In contrast to organic and total soil C, total soil N, and soil $\delta^{15}N$, total and organic soil $\delta^{13}C$ were more variable across residential yards than reference sites (Fig. 2). This supported our prediction that sources of $\delta^{13}C$ in urban environments both enrich and deplete $\delta^{13}C$ and, thus, will result in divergent soil $\delta^{13}C$ across residential yards compared to natural reference systems depending on the dominant process in each yard (e.g., high C$_4$ plant abundance). Two C sources can greatly increase soil $\delta^{13}C$: (1) organic inputs from C$_4$ plants (O'Leary 1988, Boutton 1996) or (2) calcium carbonate inputs from building materials (Boeckx et al. 2006). In contrast, urban soil $\delta^{13}C$ can be decreased due to (1) incorporation of fossil fuel-derived CO$_2$ into plants and ultimately soils (Lichtfouse et al. 2003, Wang and Pataki 2010), or (2) variation in the ratio of intracellular to ambient ratio of CO$_2$ partial pressures (c_i/c_a) in C$_3$ plants (Farquhar et al. 1989, Pataki et al. 2005a, b). In the surface and subsurface soils, organic and total soil $\delta^{13}C$ were similar between residential yards and reference sites in all cities except PHX and MIA (Fig. 2). Higher organic soil $\delta^{13}C$ in MIA and PHX in residential yards compared to reference sites was consistent with a greater fraction of C$_4$ plants in yards of these cities, relative to LA, MSP, BAL, BOS (Trammell et al. 2019). In contrast, in LA residential yards, organic soil $\delta^{13}C$ did not contribute to significantly greater total soil $\delta^{13}C$, suggesting that inorganic sources of $\delta^{13}C$ were more important in LA than organic sources.
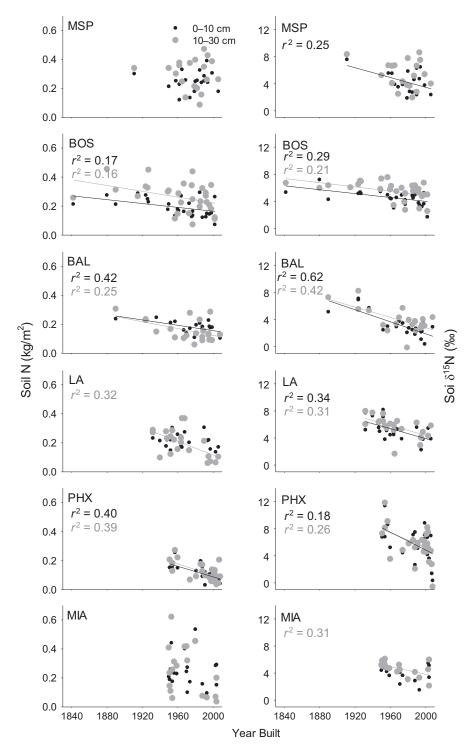
Fig. 7. Relationship between soil N (kg/m^2) and soil δ^{15}N (‰) with the year the residence was built for the surface soil (black, 0–10 cm) and the subsoil (gray, 10–30 cm) in MSP, BOS, BAL, LA, PHX, and MIA.

City scale: soil C and N

As expected, we found no significant differences in total soil C, total soil δ^{13}C, and organic soil δ^{13}C across urban, suburban, and exurban residential yards.

Previous research suggests that residential yards have high potential to sequester and store soil C (Gough and Elliott 2012, Selhorst and Lal 2013, Campbell et al. 2014). We measured significantly greater organic soil C in urban residential yards compared to suburban and

exurban yards in BAL, BOS, and PHX (Fig. 4). These findings suggest that the proportion of organic C to total soil C increases in the urban compared to suburban and exurban yards, since total soil C did not change across urbanization gradients. In BAL, BOS, and PHX, homes in urban residential parcels were substantially older than suburban (33–42 yr older) and exurban (43–54 yr older) homes, whereas LA, MIA, MSP homes in the urban, suburban, and exurban residential parcels were more similar in age (urban 12–28 yr older than suburban, and 22–33 yr older than exurban). The older residential home age in BAL, BOS, and PHX urban yards is a probable explanation for the proportional increase in organic soil C in these urban yards since there has been more time (~20 yr) for enhanced productivity from management inputs to proportionally increase organic matter inputs to the soil (Trammell et al. 2018). The proportional increase in organic soil C in older yards may also be partially explained by the presence of more residual inorganic C associated with construction in younger than older yards both within and between cities.

In addition to enhancements in N availability by lawn management, larger-scale regional N deposition patterns can alter N availability across an urbanization gradient. In four of the six cities, we found statistically greater soil N content and soil $\delta^{15}N$ in urban than suburban and/or exurban residential yards (Fig. 5). In many cities, N deposition is greater in urban compared to surrounding suburban and exurban areas (Lovett et al. 2000, Bettez and Groffman 2013, Rao et al. 2014), and greater N deposition is linked to higher soil N content and soil NO_3 leaching, which increases ^{15}N (Aber et al. 1989, Bytnerowicz and Fenn 1996, Padgett et al. 1999). In LA, we observed no significant difference in soil N content across the gradient in either surface or subsurface soils suggesting that N deposition patterns may be more consistent across this large metropolitan area, or that N fertilizer inputs in suburban or exurban residential yards resulted in similar soil N across the urbanization gradient (Law et al. 2004). However, LA urban yards had greater soil $\delta^{15}N$ indicating the potential importance of different sources of N deposition. The $\delta^{15}N$ in N deposition is higher when the primary sources are NO_3^- in wet deposition and NO_x in dry deposition (Kendall et al. 2007, Felix and Elliott 2014) suggesting that higher N deposition contributes to greater soil $\delta^{15}N$ across most cities. Alternatively, greater N deposition in urban yards may enhance soil N cycling rates and associated ^{14}N losses resulting in higher soil ^{15}N pools (Aber et al. 1989, Padgett et al. 1999), and cities with older neighborhoods in the urban areas may experience greater soil N losses over time than the younger homes in suburban and exurban areas, enriching the remaining soil N and contributing to the differences in soil $\delta^{15}N$ (Högberg 1997).

Temporal scale: soil C and N

Previous research has shown that parcel age can be a strong determinant of soil C and N dynamics in yards

(Raciti et al. 2011, Trammell et al. 2017). Across our cities, we found increases in organic soil C and $\delta^{13}C$, soil N, and soil $\delta^{15}N$ in yards surrounding older homes. While some cities had increasing total soil C and $\delta^{13}C$ with increasing home age, the opposite trend was observed in LA and PHX. This unexpected result offers insight into the sources of C inputs to soil in residential yards across cities with differing climates, management practices, and/or development patterns. While organic soil C and soil $\delta^{13}C$ were significantly greater in LA and PHX yards around older homes, greater total soil $\delta^{13}C$ in yards around newer homes in LA and PHX suggests a significant contribution from inorganic C to the total soil C pool. In these two arid cities, higher inorganic C in soils around newer homes may be due to yards developed on the alkaline parent material in semiarid ecosystems (i.e., LA and PHX; Machette 1985), pedogenic carbonate formation since irrigation water is a source of Ca^{2+} (Bughio et al. 2016), or calcium carbonate inputs from building materials during development activities (Boeckx et al. 2006).

Lawn management inputs, such as fertilizer and irrigation, can greatly increase soil C storage by enhancing aboveground productivity and subsequent organic matter inputs to soil (Bandaranayake et al. 2003, Qian et al. 2003). Greater soil organic C in older residential yards in BAL, BOS, LA, and PHX suggests the accumulation of organic matter inputs significantly increased soil organic C pools over time (Fig. 6). The simultaneous increase in soil organic $\delta^{13}C$ in BOS, LA, and PHX suggests that increased soil C cycling combined with associated ^{12}C losses enriched soil $\delta^{13}C$ over time (Boström et al. 2007). The isotopic fractionation during SOM decay can be a function of biotic (e.g., organic matter quality) and abiotic factors (e.g., soil water; Ehleringer et al. 2000, Wang et al. 2015), and even slight fractionation due to microbial activity and soil-respired CO_2 could accumulate into observable increases in soil $\delta^{13}C$ over decades (Högberg et al. 2005). However, in BAL, soil organic $\delta^{13}C$ did not increase with soil organic C over time. This variable response across cities may be caused by differences in the balance between production and decomposition (Trammell et al. 2018) or to changes in vegetation, e.g., increased inputs from trees as yards age. Similarly, we cannot exclude the influence of past land-use history (e.g., forest vs. agriculture) on soil C dynamics in residential yards (Lewis et al. 2006, 2014, Raciti et al. 2011). However, we found no difference in soil C or N in residential yards with differing previous land-use history in BAL, BOS, or PHX, which indicates influences from yard management are a stronger determinant of differences in soil C and N over time than previous land use.

In BAL and BOS, soil total C was greater in older residential yards than in younger yards suggesting that organic C inputs were an important contributor to increased total soil C in cities with warm mesic growing seasons. However, we found no increase in soil organic or total C in MSP or MIA yards around older homes. MIA is the warmest and wettest city in our study suggesting

that greater decomposition rates coupled with enhanced aboveground production rates may result in no change in soil C pools over time (Huyler et al. 2014). In the coldest city, MSP, lawn management inputs may not stimulate production or decomposition resulting in less change in soil C pools over time due to lower rates of productivity and decay in this cool climate (Chapin et al. 2002).

Total soil N exhibited patterns similar to soil organic and total C across all six cities. Total N increased with increasing home age in BAL, BOS, LA, and PHX, whereas there was no relationship with home age in MIA or MSP (Fig. 8). However, across all six cities, we observed greater soil δ^{15}N in yards around older homes. Soil δ^{15}N enrichment results from a proportional increase in enriched N sources (Kendall et al. 2007), or from soil N cycling processes that increase soil ^{14}N losses (Högberg 1997, Robinson 2001). Older homes in BAL, BOS, and PHX are in urban rather than suburban/exurban residential areas suggesting that enriched urban δ^{15}N sources may contribute to greater soil δ^{15}N in yards around older homes (Ammann et al. 1999, Saurer et al. 2004). Since greater N deposition and availability results in increased soil N cycling rates (Pardo et al. 2006, 2007), greater soil δ^{15}N in older BAL, BOS, and PHX yards is most likely a combination of enriched urban N sources and greater soil N cycling rates associated with soil ^{14}N losses over time. In LA, MIA, and MSP, greater soil δ^{15}N most likely arises from accelerated soil N cycling rates and associated ^{14}N losses since older homes are not located in highly urbanized locations in these three cities. Our results indicate that soil N cycling rates associated with soil ^{14}N losses are important drivers of soil N dynamics across residential yards. This suggests that, while yards may retain soil N initially after lawn establishment, although no accumulation in N was found after park lawn establishment over time (significant N_2O losses with fertilization; Townsend-Small and Czimczik 2010), they may become N sources over time (Petrovic 1990, Trammell et al. 2016, Smith et al. 2018).

Conclusion

This study provides evidence that conversion of native reference ecosystems to residential areas results in convergence of soil C and N levels and ecological homogenization at the continental scale in North America. Moreover, this conversion leads to increases in C and N sequestration that are likely significant at the continental scale given the large area affected by this conversion (Milesi et al. 2005). The mechanisms behind these effects are complex and involve a mixture of the importance of C_3 vs. C_4 plants, atmospheric deposition, natural vs. anthropogenic sources of inorganic C, and increased cycling rates and losses of C and N that play out over space (urban to rural gradients) and time.

Despite these complexities, much of the convergence associated with residential land-use conversion was driven most likely by enhanced C and N inputs to residential yards, which are driven by lawn management practices. Refining our estimates of whole system C and N budgets of residential yards will improve our understanding of the costs and consequences of management practices for yard C and N retention at multiple scales.

Our results further suggest that systematic comparison of reference and residential ecosystems across broad hydroclimatic gradients is a powerful platform for highlighting the importance of studying these processes in urban areas, as well as for evaluating convergence and homogenization dynamics and for the emerging fields of continental scale science and Macrosystems Biology (Heffernan et al. 2014). Insights at these scales are likely to be critically important for understanding and responding to grand challenges in environmental science related to climate and land-use change.

Literature Cited

Aber, J. D., K. J. Nadelhoffer, P. Steudler, and J. M. Melillo. 1989. Nitrogen saturation in northern forest ecosystems. BioScience 39:378–386.

Alberti, M. 2008. Advances in urban Ecology: integrating humans and ecological processes in urban ecosystems. Springer, New York, New York, USA.

Ammann, M., R. Siegwolf, F. Pichlmayer, M. Suter, M. Saurer, and C. Brunold. 1999. Estimating the uptake of traffic-derived NO_2 from ^{15}N abundance in Norway spruce needles. Oecologia 118:124–131.

Balesdent, J., C. Girardin, and A. Mariotti. 1993. Site-related ^{13}C of tree leaves and soil organic matter in a temperate forest. Ecology 74:1713–1721.

Bandaranayake, W., Y. L. Qian, W. J. Parton, D. S. Ojima, and R. F. Follett. 2003. Estimation of soil organic carbon changes in turfgrass systems using the CENTURY model. Agronomy Journal 95:558–563.

Bettez, N. D., and P. M. Groffman. 2013. Nitrogen deposition in and near an urban ecosystem. Environmental Science & Technology 47:6047–6051.

Bijoor, N. S., C. I. Czimczik, D. E. Pataki, and S. A. Billings. 2008. Effects of temperature and fertilization on nitrogen cycling and community composition of an urban lawn. Global Change Biology 14:2119–2131.

Bissonette, J. A. 1999. Small sample size problems in wildlife ecology: a contingent analytical approach. Wildlife Biology 5:65–71.

Boeckx, P., M. Van Meirvenne, F. Raulo, and O. Van Cleemput. 2006. Spatial patterns of δ^{13}C and δ^{15}N in the urban topsoil of Gent, Belgium. Organic Geochemistry 37:1383–1393.

Boström, B., D. Comstedt, and A. Ekbald. 2007. Isotope fractionation and ^{13}C enrichment in soil profiles during the decomposition of soil organic matter. Oecologia 153:89–98.

Boutton, T. W. 1996. Stable carbon isotope ratios of organic matter and their use as indicators of vegetation and climate

changes. Pages 47–82 *in* S. Yamasaki and T. W. Boutton, editor. Mass spectrometry of soils. Marcel Dekker, New York, New York, USA.

Bughio, M. A., P. Wang, F. Meng, C. Qing, Y. Kuzyakov, X. Wang, and S. A. Junejo. 2016. Neoformation of pedogenic carbonates by irrigation and fertilization and their contribution to carbon sequestration in soil. Geoderma 262:12–19.

Bytnerowicz, A., and M. E. Fenn. 1996. Nitrogen deposition in California forests: a review. Environmental Pollution 92:127–146.

Campbell, C. D., J. R. Seiler, P. E. Wiseman, B. D. Strahm, and J. F. Munsell. 2014. Soil carbon dynamics in residential lawns converted from Appalachian mixed oak stands. Forests 5:425–438.

Carreiro, M. M., K. Howe, D. F. Parkhurst, and R. V. Pouyat. 1999. Variation in quality and decomposability of red oak leaf litter along an urban-rural gradient. Biology and Fertility of Soils 30:258–268.

Chapin, F. S., P. A. Matson, and H. A. Mooney. 2002. Principles of terrestrial ecosystem ecology. Springer Science and Business Media, Berlin, Germany.

Dawson, T. E., S. Mambelli, A. H. Plamboeck, P. H. Templer, and K. P. Tu. 2002. Stable isotopes in plant ecology. Annual Review of Ecology and Systematics 33:507–559.

Ehleringer, J. R., N. Buchmann, and L. B. Flanagan. 2000. Carbon isotope ratios in belowground carbon cycle processes. Ecological Applications 10:412–422.

Fang, Y., P. Gundersen, R. D. Vogt, K. Koba, F. Chen, X. Y. Chen, and M. Yoh. 2011. Atmospheric deposition and leaching of nitrogen in Chinese forest ecosystems. Journal of Forest Research 16:341–350.

Farquhar, G. D., J. R. Ehleringer, and K. T. Hubick. 1989. Carbon isotope discrimination and photosynthesis. Annual Review of Plant Biology 40:503–537.

Felix, J. D., and E. M. Elliott. 2014. Isotopic composition of passively collected nitrogen dioxide emissions: vehicle, soil and livestock source signatures. Atmospheric Environment 92:359–366.

Golubiewski, N. E. 2006. Urbanization increases grassland carbon pools: effects of landscaping in Colorado's front range. Ecological Applications 16:555–571.

Gough, C. M., and H. L. Elliott. 2012. Lawn soil carbon storage in abandoned residential properties: an examination of ecosystem structure and function following partial human-natural decoupling. Journal of Environmental Management 98:155–162.

Groffman, P. M., C. O. Williams, R. V. Pouyat, L. E. Band, and I. D. Yesilonis. 2009. Nitrate leaching and nitrous oxide flux in urban forests and grasslands. Journal of Environmental Quality 38:1848–1860.

Groffman, P. M., et al. 2014. Ecological homogenization of urban USA. Frontiers in Ecology and the Environment 12:74–81.

Groffman, P. M., et al. 2016. Satisfaction, water and fertilizer use in the American residential macrosystem. Environmental Research Letters 11:034004.

Hall, S. J., et al. 2015. Convergence of microclimate in residential landscapes across diverse cities in the United States. Landscape Ecology 31:101–117.

Heffernan, J. B., et al. 2014. Macrosystems ecology: understanding ecological patterns and processes at continental scales. Frontiers in Ecology and the Environment 12:5–14.

Högberg, P. 1997. Tansley review no. 95: ^{15}N natural abundance in soil-plant systems. New Phytologist 137:179–203.

Högberg, P., A. Ekblad, A. Nordgren, A. H. Plamboeck, A. Ohlsson, S. Bhupinderpal-Singh, and M. Höogberg. 2005. Factors determining the ^{13}C abundance of soil-respired CO_2 in boreal forests. Pages 47–68 *in* L. B. Flanagan, J. R. Ehleringer, and D. E. Pataki, editors. Stable isotopes and biosphere-atmosphere interactions: processes and biological controls. Elsevier, New York, New York, USA.

Huyler, A., A. H. Chappelka, S. A. Prior, and G. L. Somers. 2014. Drivers of soil carbon in residential 'pure lawns' in Auburn, Alabama. Urban Ecosystems 17:205–219.

Idso, C. D., S. B. Idso, and R. C. Balling Jr. 1998. The urban CO_2 dome of Phoenix, Arizona. Physical Geography 19:95–108.

Idso, C. D., S. B. Idso, and R. C. Balling Jr. 2001. An intensive two-week study of an urban CO_2 dome in Phoenix, Arizona, USA. Atmospheric Environment 35:995–1000.

Ignatieva, M., et al. 2015. Lawn as a cultural and ecological phenomenon: a conceptual framework for transdisciplinary research. Urban Forestry & Urban Greening 14:383–387.

Kaye, J. P., P. M. Groffman, N. B. Grimm, L. A. Baker, and R. V. Pouyat. 2006. A distinct urban biogeochemistry? Trends in Ecology & Evolution 21:192–199.

Kendall, C., E. M. Elliott, and S. D. Wankel. 2007. Tracing anthropogenic inputs of nitrogen to ecosystems. Pages 375–449 *in* K. Lajtha and R. H. Michener, editors. Stable isotopes in ecology and environmental science. Second edition. Blackwell Scientific Publications, Malden, Massachusetts, USA.

Law, N. L., L. E. Band, and J. M. Grove. 2004. Nitrogen input from residential lawn care practices in suburban watersheds in Baltimore County, MD. Journal of Environmental Planning and Management 47:737–755.

Lewis, D. B., J. P. Kaye, C. Gries, A. P. Kinzig, and C. L. Redman. 2006. Agrarian legacy in soil nutrient pools of urbanizing arid lands. Global Change Biology 12:703–709.

Lewis, D. B., J. P. Kaye, and A. P. Kinzig. 2014. Legacies of agriculture and urbanization in labile and stable organic carbon and nitrogen in Sonoran Desert soils. Ecosphere 5:1–18.

Lichtfouse, E., M. Lichtfouse, and A. Jaffrézic. 2003. δ^{13}C values of grasses as a novel indicator of pollution by fossil-fuel-derived greenhouse gas CO_2 in urban areas. Environmental Science & Technology 37:87–89.

Lovett, G. M., M. M. Traynor, R. V. Pouyat, M. M. Carreiro, W.-X. Zhu, and J. W. Baxter. 2000. Atmospheric deposition to oak forests along an urban-rural gradient. Environmental Science and Technology 34:4294–4300.

Machette, M. N. 1985. Calcific soils of the southwestern United States. Pages 1–21 *in* D. L. Weide, editor. Soils and quarternary geology of the southwestern United States. Special Paper 203. Geological Society of America, Boulder, Colorado, USA.

Mann, H. B., and D. R. Whitney. 1947. On a test of whether one of two random variables is stochastically larger than the other. Annals of Mathematical Statistics 18:50–60.

Milesi, C., S. W. Running, C. D. Elvidge, J. B. Dietz, B. T. Tuttle, and R. R. Nemani. 2005. Mapping and modeling the biogeochemical cycling of turf grasses in the United States. Environmental Management 36:426–438.

National Climatic Data Center (NCDC). 2014. National Oceanic and Atmospheric Administration (NOAA) satellite and information service. 1981–2010 Climate Normals. http://www.ncdc.noaa.gov/

O'Leary, M. H. 1988. Carbon isotopes in photosynthesis. BioScience 38:328–336.

Padgett, P. E., E. B. Allen, A. Bytnerowicz, and R. Minich. 1999. Changes in soil inorganic nitrogen as related to atmospheric nitrogenous pollutants in southern California. Atmospheric Environment 33:769–781.

Pardo, L. H., S. G. McNulty, J. L. Boggs, and S. Duke. 2007. Regional patterns in foliar ^{15}N across a gradient of nitrogen deposition in the northeastern US. Environmental Pollution 149:293–302.

Pardo, L. H., et al. 2006. Regional assessment of N saturation using foliar and root δ^{15}N. Biogeochemistry 80:143–171.

Pataki, D. E., B. J. Tyler, R. E. Peterson, A. P. Nair, W. J. Steenburgh, and E. R. Pardyjak. 2005*a*. Can carbon dioxide be used as a tracer of urban atmospheric transport? Journal of Geophysical Research: Atmospheres 110:1–8.

Pataki, D. E., S. E. Bush, and J. R. Ehleringer. 2005*b*. Stable isotopes as a tool in urban ecology. Pages 199–214 *in* L. B. Flanagan, J. R. Ehleringer, and D. E. Pataki, editors. Stable isotopes and biosphere-atmosphere interactions: processes and biological controls. Elsevier Academic Press, Cambridge, Massachusetts, USA.

Petrovic, A. M. 1990. The fate of nitrogenous fertilizers applied to turfgrass. Journal of Environment Quality 19:1–14.

Pohlert, T. 2014. The Pairwise Multiple Comparison of Mean Ranks Package (PMCMR). R Package. http://ftp.ussg.iu.edu/CRAN/

Polsky, C., et al. 2014. Assessing the homogenization of urban land management with an application to US residential lawn care. Proceedings of the National Academy of Sciences USA 111:4432–4437.

Pouyat, R. V., P. M. Groffman, I. Yesilonis, and L. Hernandez. 2003. Soil carbon pools and fluxes in urban ecosystems. Pages 347–362 *in* J. M. Kimble, L. S. Heath, R. A. Birdsey, and R. Lal, editors. The potential of U.S. forest soils to sequester carbon and mitigate the greenhouse effect. CRC Press, Boca Raton, Florida, USA.

Pouyat, R. V., and T. L. E. Trammell. 2019. Climate change and urban forest soils. 36, 189–211 *in* M. Busse, D. Dumroese, D. Morris, and C. Giardina, editors. Global change and forest soils: demands and adaptions of a finite natural resource. Elsevier, New York, New York, USA.

Pouyat, R. V., I. D. Yesilonis, M. Dombos, K. Szlavecz, H. Setälä, S. Cilliers, E. Hornung, D. J. Kotze, and S. Yarwood. 2015. A global comparison of surface soil characteristics across five cities: a test of the urban ecosystem convergence hypothesis. Soil Science 180:1–10.

Pouyat, R. V., I. D. Yesilonis, and N. E. Golubiewski. 2009. A comparison of soil organic carbon stocks between residential turf grass and native soil. Urban Ecosystems 12:45–62.

Qian, Y. L., W. Bandaranayake, W. J. Parton, B. Mecham, M. A. Harivandi, and A. R. Mosier. 2003. Long-term effects of clipping and nitrogen management in turfgrass on soil organic carbon and nitrogen dynamics: the CENTURY model simulation. Journal of Environmental Quality 32:1694–1700.

R Core Team. 2016. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. http://www.R-project.org/

Raciti, S. M., P. M. Groffman, J. C. Jenkins, R. V. Pouyat, T. J. Fahey, S. T. A. Pickett, and M. L. Cadenasso. 2011. Accumulation of carbon and nitrogen in residential soils with different land-use histories. Ecosystems 14:287–297.

Rao, P., L. Hutyra, S. Raciti, and P. Templer. 2014. Atmospheric nitrogen inputs and losses along an urbanization gradient from Boston to Harvard Forest, MA. Biogeochemistry 121:229–245.

Richards, N. A., J. R. Mallette, R. J. Simpson, and E. A. Macie. 1984. Residential greenspace and vegetation in a mature city: Syracuse, New York. Urban Ecology 8:99–125.

Robbins, P., A. Polderman, and T. Birkenholtz. 2001. Lawns and toxins: an ecology of the city. Cities 18:369–380.

Robinson, D. 2001. δ^{15}N as an integrator of the nitrogen cycle. Trends in Ecology & Evolution 16:153–162.

Romero, C. C., and M. D. Dukes. 2013. Net irrigation requirements for Florida turfgrasses. Irrigation Science 31:1213–1224.

Saurer, M., P. Cherubini, M. Ammann, B. De Cinti, and R. Siegwolf. 2004. First detection of nitrogen from NO_x in tree rings: a $^{15}N/^{14}N$ study near a motorway. Atmospheric Environment 38:2779–2787.

Selhorst, A., and R. Lal. 2013. Net carbon sequestration potential and emissions in home lawn turfgrasses of the United States. Environmental Management 51:198–208.

Smith, R. M., J. C. Williamson, D. E. Pataki, J. R. Ehleringer, and P. Dennison. 2018. Soil carbon and nitrogen accumulation in residential lawns of the Salt Lake Valley, Utah. Oecologia 187:1107–1118.

Townsend-Small, A., and C. I. Czimczik. 2010. Carbon sequestration and greenhouse gas emissions in urban turf. Geophysical Research Letters 37:L02707.

Trammell, T. L. E., S. Day, R. V. Pouyat, C. Rosier, B. Scharenbroch, and I. Yesilonis. 2018. Drivers of urban soil carbon dynamics. Pages 93–120 *in* R. Lal and B. A. Stewart, editors. Urban soils: advances in soil science. Taylor and Francis Publishing, Boca Raton, Florida, USA.

Trammell, T. L. E., D. E. Pataki, J. Cavender-Bares, P. M. Groffman, S. J. Hall, J. B. Heffernan, S. E. Hobbie, J. L. Morse, C. Neill, and K. C. Nelson. 2016. Plant nitrogen concentration and isotopic composition in residential lawns across seven U.S. cities. Oecologia 181:271–285.

Trammell, T. L. E., R. V. Pouyat, M. M. Carreiro, and I. Yesilonis. 2017. Drivers of soil and tree carbon dynamics in urban residential lawns: a modeling approach. Ecological Applications. https://doi.org/10.1002/eap.1502

Trammell, T. L. E., et al. 2019. Climate and lawn management interact to control the distribution of C_4 plants in residential lawns across seven U.S. cities. Ecological Applications. e01884.

U.S. Census Bureau. 2010. Annual estimates of the resident population: April 1, 2010 to July 1, 2013. U.S. Census Bureau, Population Division. Release Date: March 2014. http://www.census.gov/population/metro/data/index.html

U.S. Geological Survey (USGS). 2008. USGS Global Ecosystems Data Viewer. http://rmgsc.cr.usgs.gov/ecosystems/dataviewer.shtml

Wang, G., Y. Jia, and L. Wei. 2015. Effects of environmental and biotic factors on carbon isotopic fractionation during decomposition of soil organic matter. Scientific Reports 5:11043.

Wang, W., and D. E. Pataki. 2010. Spatial patterns of plant isotope tracers in the Los Angeles urban region. Landscape Ecology 25:35–52.

SUPPORTING INFORMATION

DATA AVAILABILITY