

Plant nitrogen concentration and isotopic composition in residential lawns across seven US cities

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Abstract Human drivers are often proposed to be stronger than biophysical drivers in influencing ecosystem structure and function in highly urbanized areas. In residential land cover, private yards are influenced by individual homeowner preferences and actions while also experiencing large-scale human and biophysical drivers. We studied plant nitrogen (%N) and N stable isotopic composition ($\delta^{15}\text{N}$) in residential yards and paired native ecosystems in seven cities across the US that span major ecological biomes and climatic regions: Baltimore, Boston, Los Angeles, Miami, Minneapolis-St. Paul, Phoenix,

and Salt Lake City. We found that residential lawns in three cities had enriched plant $\delta^{15}\text{N}$ ($P < 0.03$) and in six cities higher plant N (%) relative to the associated native ecosystems ($P < 0.05$). Plant $\delta^{15}\text{N}$ was progressively depleted across a gradient of urban density classes in Baltimore and Boston ($P < 0.05$). Lawn fertilization was associated with depleted plant $\delta^{15}\text{N}$ in Boston and Los Angeles ($P < 0.05$), and organic fertilizer additions were associated with enriched plant $\delta^{15}\text{N}$ in Los Angeles and Salt Lake City ($P < 0.04$). Plant $\delta^{15}\text{N}$ was significantly enriched as a function of housing age in Baltimore ($r^2 = 0.27$, $P < 0.02$), Boston ($r^2 = 0.27$, $P < 0.01$), and Los Angeles ($r^2 = 0.34$, $P < 0.01$). These patterns in plant $\delta^{15}\text{N}$ and plant N (%) across these cities suggests that N sources to lawns, as well as greater rates of N cycling combined with subsequent N losses, may be important drivers of plant N dynamics in lawn ecosystems at the national scale.

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Introduction

Cities and their surrounding suburban developments are complex systems experiencing multiple ecological and social drivers that mimic major global changes making cities important representations of global environmental change (Grimm et al. 2008). A socio-ecological approach that incorporates both human and biophysical dimensions is necessary for analysis of urban systems (Alberti 2008; McDonnell 2011). A commonly accepted idea in urban ecology is that human drivers can be stronger than biophysical drivers in controlling ecosystem structure and function (Pickett et al. 2011) and these factors interact in

complex ways across scales. For example, if human preferences (e.g., for green lawns) lead to behaviors and actions (e.g., irrigation) that have a stronger influence on ecosystem responses than biophysical factors (e.g., climate), then anthropogenic drivers should result in urban ecosystems that are more similar in ecosystem structure and function to each other than to the unmanaged ecosystems that surround them. Hence, urban ecosystems over broad geographic scales may be converging to a similar, relatively homogeneous state (Pouyat et al. 2003). While there is some evidence of homogenization of plant structure and convergence of soil characteristics in cities, there is little understanding of the extent to which this pattern occurs across US metropolitan areas (Groffman et al. 2014). Residential areas are an ideal land cover in which to evaluate potential convergence of urban processes since individual homeowner preferences and actions influence residential yards, which also experience larger scale influences of urbanization and local climate.

Turfgrass composes a significant proportion of land area in urbanized landscapes (Robbins and Birkenholtz 2003), and is now the largest irrigated crop in the US (Milesi et al. 2005). As a result, turfgrass has a substantial influence on biogeochemical cycles at regional and national scales. Large inputs of water and nutrients are used to establish and maintain lawns in many residential landscapes (Robbins et al. 2001), and in the absence of water and fertilizer additions, turfgrasses would not survive in many areas, especially the arid west (Milesi et al. 2005). It is possible that irrigation and fertilization are mechanisms leading to convergence of nutrient availability [e.g., nitrogen (N)] in lawns. However, the demographic characteristics of homeowners result in large differences in fertilizer application rates. Fertilizer application rates were greater in wealthier residential yards where homeowners had higher income or more valuable homes than in other areas (Robbins et al. 2001; Osmond and Hardy 2004; Fraser et al. 2013; Martini et al. 2015). In newer residential developments, fertilizer application rates were greater than in older residential lawns, due to lower baseline soil N availability in newer lots (Law et al. 2004). These studies suggest that socioeconomic status (SES) and age of lawn establishment influence homeowner management practices. Thus, there is considerable spatial and temporal variability in these practices at finer spatial scales (i.e., across yards) within cities. To develop an understanding of how social and environmental drivers interact to affect lawn dynamics within and among cities, there is a strong need for integrative measures of lawn biogeochemistry.

While homeowner management practices directly influence urban residential landscapes, the surrounding urbanized landscape also influences residential ecosystems (e.g., pollution from fossil fuel combustion; urban heat island

effects). Understanding how multiple direct and indirect anthropogenic drivers influence N cycling in residential areas can be challenging. Natural abundance ^{15}N ($\delta^{15}\text{N}$) of plants can integrate information about N sources to ecosystems (e.g., fossil fuel combustion products, fertilizer) as well as N cycling and N loss within ecosystems [e.g., soil nitrification (Högberg 1997; Evans 2001; Robinson 2001)]. Foliar $\delta^{15}\text{N}$ is a useful indicator of soil N cycling rates and losses. Discrimination against ^{15}N occurs during most soil N processes, resulting in depleted products relative to the substrate [e.g., nitrification—the product nitrate (NO_3^-) is depleted relative to the substrate NH_4^+ which is enriched (Dawson et al. 2002)]. Thus, ecosystem N losses from leaching of NO_3^- or transformations of dissolved N to gaseous products in nitrification, denitrification, and volatilization results in enrichment of the remaining soil N (Högberg 1997; Pardo et al. 2006, 2007). In addition, bulk soil $\delta^{15}\text{N}$ shows enrichment with increasing soil depth (Nadelhoffer and Fry 1988; Högberg 1997; Boström et al. 2007). Thus, measurements of plant $\delta^{15}\text{N}$ can provide indicators of plant acquisition of soil N and the influence of ecosystem N losses over time.

Plant $\delta^{15}\text{N}$ can also provide a tracer of N sources when $\delta^{15}\text{N}$ of potential sources is isotopically distinct (Robinson 2001). $\delta^{15}\text{N}$ of pollution-derived N deposition is typically isotopically enriched (Ammann et al. 1999; Pearson et al. 2000; Saurer et al. 2004; Redling et al. 2013) while N in inorganic fertilizer is similar to the atmospheric standard and therefore close to 0 ‰ (Bijoor et al. 2008). The amount and timing of fertilizer applications may be a strong determinant of whether plant $\delta^{15}\text{N}$ is close to 0 ‰ (i.e., high fertilizer application, recently applied) or enriched due to enhanced rates of internal N cycling and volatilization and leaching of fertilizer applications over time. Additionally, the magnitude of N inputs from atmospheric deposition relative to fertilizer could influence soil and plant $\delta^{15}\text{N}$.

In this study, we evaluated how anthropogenic activities influence N cycling of residential lawns in US metropolitan areas. We measured plant $\delta^{15}\text{N}$ and N (%) in lawns and paired reference natural areas in seven cities across the US (Baltimore, Boston, Los Angeles, Miami, Minneapolis-St. Paul, Phoenix, and Salt Lake City), which span multiple ecological biomes and climatic regions. The goals of this study were to determine patterns of plant $\delta^{15}\text{N}$ and N (%) at the national scale (across cities), at a finer spatial scale within each city (within cities), and over time (as a function of housing age). We posed the following questions:

1. Are there differences in residential or native site plant $\delta^{15}\text{N}$ and plant N (%) across the seven cities? If so, what drives these differences?
2. How do plant $\delta^{15}\text{N}$ and N (%) vary as a function of urban density, SES, and housing age?

3. How do lawn fertilization practices influence plant $\delta^{15}\text{N}$ and N (%)?

We expected differences among regions as a function of mean annual temperature (MAT) or mean annual precipitation (MAP). Specifically, we expected differences in residential plant $\delta^{15}\text{N}$ and plant N (%) across cities to be related to MAT (and not MAP) since water stress will be alleviated by irrigation in dry cities, whereas differences across native site plant $\delta^{15}\text{N}$ and plant N (%) may be driven by MAT and MAP. We hypothesized that plant $\delta^{15}\text{N}$ would be more enriched and plant N (%) would increase as a function of urban density and affluence because of higher N inputs, via N deposition and fertilization, and associated rates of internal N cycling and accelerated soil N losses. Furthermore, we hypothesized that lawns of older homes would have more enriched plant $\delta^{15}\text{N}$ because of greater cumulative rates of internal N cycling and losses of N over time, and that fertilization practices would be associated with higher plant N (%) and $\delta^{15}\text{N}$ due to accelerated soil N processes and coupled N losses over time.

Materials and methods

Study area

We sampled seven major metropolitan areas across the United States: Baltimore, MD (BAL); Boston, MA (BOS); Los Angeles, CA (LA); Miami, FL (MIA); Minneapolis-St. Paul, MN (MSP); Phoenix, AZ (PHX); and Salt Lake City, UT (SLC). These cities have total Metropolitan Statistical Area populations greater than 1 million (US Census Bureau

2010; Table 1). They were chosen to represent seven different ecological biomes (US Geological Survey 2008) and major climatic regions (National Climatic Data Center 2014) across the USA (Table 1).

Yard selection and experimental design

Initial telephone surveys (9480 respondents) were conducted in six of the seven cities (BAL, BOS, LA, MIA, MSP, and PHX), and homeowners who indicated willingness to participate in our in-person yard study were contacted for permission to follow up with field sampling of their yard. SLC was included after the initial telephone survey was completed, and SLC residential homeowners were initially contacted via mailed letters. In all cities, the experimental design included residential sites ($n = 17\text{--}30$ per city) and sites that represented the region's native ecosystem(s) (hereafter referred to as "native" sites; $n = 3\text{--}6$ per city; Table 2). Native sites within each city were selected to represent the native ecosystems present before residential landscape development and the extent of the ecological biomes for each city. For example, in Los Angeles, all residential sites sampled were located where the native ecosystem prior to urban development was coastal sage scrub, thus we sampled Californian coastal scrub as the reference native ecosystem (Table 1). Furthermore, in Minneapolis-St. Paul, native sites were selected in interior forest and savanna ecosystems to represent the variety of native ecosystems prior to urbanization where residential sites were positioned. Therefore, native sites were sampled to encompass all the possible native ecosystems that were present prior to urbanization and residential development.

Residential sites were chosen based on the potential rating index for zipcode markets (PRIZM) (Claritas) market

Table 1 Metropolitan statistical area (MSA) population data (US Census Bureau 2010), ecological biome (US Geological Survey 2008), and climate data (National Climatic Data Center 2014) for

Baltimore, MD (BAL); Boston, MA (BOS); Los Angeles, CA (LA); Miami, FL (MIA); Minneapolis-St. Paul, MN (MSP); Phoenix, AZ (PHX); and Salt Lake City, UT (SLC)

City	Founding year	Population rank ^a	Total population ^b	Biome	Climatic region	MAT (°C)	MAP (cm)
BAL	1729	20	2,710,489	Southern piedmont forest	Temperate humid	12.8	106.4
BOS	1630	10	4,552,402	Northern hardwood forest	Temperate humid	10.8	111.2
LA	1781	2	12,828,837	Southern CA coastal scrub	Mediterranean semiarid	17.0	32.6
MIA	1896	8	5,564,635	South FL hardwood hammock	Tropical subhumid	25.1	157.2
MSP	1867	16	3,348,859	Oak savanna/north central interior forest	Temperate subhumid-humid	7.9	77.7
PHX	1868	12	4,192,887	Sonora-Mojave desert scrub	Mediterranean arid	23.9	20.4
SLC	1847	48	1,087,873	Sagebrush shrub-steppe	Mediterranean subhumid-dry	11.6	40.9

MAT Mean annual temperature, MAP mean annual precipitation

^a Population rank of each city according to MSA population

^b Total population for each MSA counted in the 2010 census (number of persons)

Table 2 Factors included in the experimental design for BAL, BOS, LA, MIA, MSP, PHX, and SLC; residential and native site sample sizes are based on the number of lawn bulk plant samples collected and analyzed for each city

City	Residential sites (<i>n</i>)	Native sites (<i>n</i>)	Urban density ^a	Socioeconomic status ^b	Housing age ^c	Previous land use ^d	Soil conditions ^e	Other
BAL	21	3	Y	Y	Y	Y	–	
BOS	30	6	Y	Y	Y	Y	–	
LA	20	3	Y	Y	Y	–	–	Distance to coast ^f
MIA	20	4	Y	Y	–	–	Y	
MSP	20	6	Y	Y	Y	–	Y	Yard landscaping ^g
PHX	17	3	Y	Y	Y	Y	–	
SLC	30	3	Y	Y	Y	–	–	

Y indicates the factor was included in the experimental design for that city, *dash* indicates the factor was not included in the experimental design; for other abbreviations, see Table 1

^a Urban, suburban, exurban

^b High, medium, low

^c Year house built

^d Agriculture, forest, pasture or desert

^e Sandy vs. not sandy

^f Close to the coast vs. further from the coast

^g Lawn, xeric

classification system that incorporates multiple aspects of socioeconomic data, such as housing density, economic status, and lifestyle (Claritas 2008). PRIZM is a national database that allows for stratification of households with consistent residential characteristics across the USA. Within each city, residential sites were selected based on urban density classes (urban, suburban, and exurban) and SES levels (high, medium, and low; based on income and house values). In all seven cities, residential yards with two or three levels of SES in the urban density class were included. In MIA, MSP, and SLC, two levels of SES were also sampled in the suburban density class. To account for local variation in factors controlling residential yard structure and function, additional factors (e.g., housing age, soil conditions, previous land use) were incorporated into experimental designs for each individual city. Previous land use was incorporated into the experimental design for BAL (agriculture and forest), BOS (pasture and forest), and PHX (agriculture and desert). Underlying soil conditions were included in MIA (sand vs. rocky limestone ridge) and MSP (on sandy outwash vs. off sandy outwash) experimental designs. In LA, differences in temperature occur with increasing distance to the coast, thus, residential sites close to (~10 km) and further from (~46 km) the coast were chosen across the LA valley. In PHX, xeriscaping is a common residential landscape type (rock ground cover with drought-tolerant shrubs), thus sites with xeriscaping ($n = 5$) and with more mesic lawn landscaping ($n = 6$) were included. SLC was not used in urban density and housing age analyses due to differences in the experimental design (i.e., urban density—two classes instead of three; housing age—two categories instead of a continuous variable).

Fertilization practices

To assess differences in fertilization practices, results from the telephone survey (BAL, BOS, LA, MIA, MSP, and PHX) were used to assign each yard to fertilized and unfertilized categories. Residents were asked if fertilizers had been applied to any part of their yard in the last year, and all residents sampled in this study either answered “yes” or “no.” In SLC, residents were asked, “If you fertilize your lawn, how often do you do so?” and answers were coded from the following options: every few weeks, every few months, seasonally, as needed, other, I do not fertilize, or I do not know. Residents who answered “every few weeks,” “every few months,” “seasonally,” or “as needed” were coded as Yes to fertilization, and those who answered “I do not fertilize” were coded as No. In SLC and LA, we asked additional questions during in-person interviews about resident fertilization practices (e.g., organic/inorganic fertilizer additions).

Plant leaf N

Plant leaf N was measured by collecting bulk plant leaf samples during the peak growing season for each city (i.e., summer 2012 for BAL, BOS, MSP, and MIA; spring 2013 for LA and PHX; summer 2013 for SLC). In the native and residential sites, leaf samples were the plant material collected for the bulk plant samples. Two random bulk plant samples (i.e., lawn clippings) were collected in each lawn (also referred to as residential site). In the native sites, two bulk plant samples were collected in two random locations along 100-m transects. Plant leaf samples were collected in the native sites to capture the variability represented in the same strata as in the residential lawns by collecting leaves from two to three of the most abundant species. In savanna or meadow native sites, samples were ground cover such as grass species. In sites where ground cover vegetation was not present (e.g., eastern deciduous forest, coastal sage scrub), then plants were collected close to the ground, less than 1 m high in almost all cases (e.g., ground shrubs in the coastal sage scrub of LA, leaves from tree saplings in BAL). Thus, plant type and sun exposure varied across the native sites (Table 3). In two cities (LA and SLC), replicate bulk plant samples were collected within 30 cm of each other at each sampling location in the lawn or in the native site. For cities that did not collect replicate samples (i.e., BAL, BOS, MIA, and PHX), each bulk plant sample was divided prior to sample processing as a within-sample replicate for N elemental analysis. In one city (MSP), species-specific plant leaf samples were collected instead of bulk plant samples. Thus, the weighted average for each species was calculated from abundance lawn quadrat data and applied to $\delta^{15}\text{N}$, N (%), and C:N data. After collection, plant leaves were dried at 60 °C for at least 48 h.

In preparation for N elemental and isotope analyses, any plant material (i.e., flowers, roots) unintentionally collected with the leaf samples was removed from the plant bulk

samples. Plant leaf samples were ground to a fine powder using a Retsch ball mixer mill (MM200; Retsch, Haan, Germany). C and N (%) and $\delta^{15}\text{N}$ were measured with a Delta Plus isotope ratio mass spectrometer (Finnigan-MAT, Bremen, Germany) interfaced with an elemental analyzer (model 1110; Carlo Erba, Milan; Stable Isotope Ratio Facility for Environmental Research at the University of Utah, Salt Lake City). All C:N ratios were expressed on a molar basis. Two primary laboratory reference materials, calibrated against National Institute of Standards and Technology and International Atomic Energy Agency certified reference materials, and one secondary laboratory (spinach leaf) reference material were used as internal standards with $\delta^{15}\text{N}$ and N (%) precision of $\pm 0.2\text{‰}$ and $\pm 0.4\text{‰}$, respectively. The stable isotope values were expressed relative to the international standard (atmospheric N_2) in the conventional δ -notation:

$$\delta^{15}\text{N} = \left[\left(\frac{{}^{15}\text{N}_{\text{sample}}/{}^{14}\text{N}_{\text{sample}}}{({}^{15}\text{N}_{\text{standard}}/{}^{14}\text{N}_{\text{standard}})} \right) - 1 \right] \times 1000\text{‰}$$

Statistical analyses

To assess differences in residential and native plant $\delta^{15}\text{N}$ and N (%), we performed non-parametric statistical tests for analyses with an unbalanced design or when normality was difficult to assess due to 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. To determine differences in plant $\delta^{15}\text{N}$ and N (%) within and across native and residential sites, we used a two-factor nested ANOVA. For native sites, transects were nested within cities, and for residential sites, urban density classes (urban, suburban, and exurban) were nested within cities. Further analysis of differences in residential plant $\delta^{15}\text{N}$, N (%), and molar C:N across all seven cities and across urban density classes were analyzed using one-way ANOVA for data that met the assumptions of normality and homoscedasticity, followed by post hoc Tukey honest significant difference tests. The non-parametric Kruskal–Wallis rank sum test followed by post hoc Nemenyi tests [pairwise multiple comparison of mean ranks (PMCMR) R package (Pohlert 2014)] were used when the assumptions for ANOVA were not met after log transformation. Differences in native plant $\delta^{15}\text{N}$, N (%), and molar C:N across all seven cities were analyzed using the non-parametric Kruskal–Wallis rank sum test followed by post hoc Nemenyi tests (Pohlert 2014). Regression analysis was used to determine whether MAT (degrees Celsius) or MAP (centimeters) explained significant variation in residential or native plant $\delta^{15}\text{N}$, N (%), and molar C:N. Differences between residential and native plant $\delta^{15}\text{N}$, N (%),

Table 3 The dominant plant growth forms for each native site type across the cities; sun exposure of plant leaves sampled for each native site type; for abbreviations, see Table 1

City	Native site type	Plant growth form	Sun or shade leaves
BAL	Forest	Forb/shrub/small tree	Shade
BOS	Meadow	Forb/grass	Sun
BOS	Forest	Forb/shrub	Shade
LA	Coastal scrub	Forb/grass/shrub	Sun
MIA	Forest	Forb/grass/vine	Shade
MSP	Savanna	Forb/grass/shrub	Sun
MSP	Forest	Forb/shrub/vine	Shade
PHX	Desert scrub	Forb/grass/shrub	Sun
SLC	Sagebrush shrub	Forb/grass/shrub	Sun

and molar C:N were determined using the non-parametric Mann–Whitney rank sum test.

Statistical differences between fertilized and unfertilized yards within each city were determined using a two-sample *t*-test, unless assumptions of normality were not met after log transformation, in which case the non-parametric Mann–Whitney rank sum test was employed. Differences in plant $\delta^{15}\text{N}$, N (%), and molar C:N between inorganic and organic fertilization treatments in LA and SLC were determined using the non-parametric Mann–Whitney rank sum test. Similarly, the non-parametric Mann–Whitney rank sum test was used to determine significant differences in plant $\delta^{15}\text{N}$, N (%), and molar C:N between SES (high and medium SES in urban and suburban density classes in all cities), previous land use (BAL, BOS, and PHX), soil conditions (MIA and MSP), distance to the coast (LA), and yard design (PHX). An exception was in MIA and MSP which had three levels of SES (high, medium, and low) within the urban density class and a Kruskal–Wallis rank sum test was used. Regression analysis was used to determine whether the year the residence was built explained significant variation in plant $\delta^{15}\text{N}$, N (%), and molar C:N. Finally, Pearson correlations between plant $\delta^{15}\text{N}$ and N (%) or molar C:N were used to explain plant N patterns [Hmisc R package (Harrell et al. 2014)]. All tests for significance are reported at the $\alpha = 0.05$ critical value, and in a few cases the $\alpha \leq 0.10$ critical values are reported as marginally significant to identify potential trends. All statistical analyses were performed using the R statistical package (R Core Team 2013).

Results

Plant N across cities

We found significant differences in plant $\delta^{15}\text{N}$ within and among native sites and residential lawns, and city-scale differences were related to regional climate. Plant $\delta^{15}\text{N}$ was significantly different among transects within native sites ($F_{8,18} = 2.78$, $P < 0.01$) and among the native sites ($F_{8,18} = 15.77$, $P < 0.001$; Fig. 1a). Residential plant $\delta^{15}\text{N}$ was significantly different among residential yards ($F_{6,13} = 1.95$, $P = 0.03$) and among cities ($F_{6,13} = 15.37$, $P < 0.001$; Fig. 1b). Further analysis demonstrated plant $\delta^{15}\text{N}$ was significantly enriched in LA, MIA, and PHX residential lawns compared with BAL, BOS, MSP, and SLC residential lawns ($P < 0.05$), and native plant $\delta^{15}\text{N}$ was significantly enriched in LA, PHX, and SLC compared with BAL, BOS, MIA, and MSP ($P < 0.05$). Across the seven cities, residential plant $\delta^{15}\text{N}$ significantly increased with increasing MAT ($^{\circ}\text{C}$; $r^2 = 0.725$, $P = 0.009$), whereas native plant $\delta^{15}\text{N}$ significantly decreased with increasing

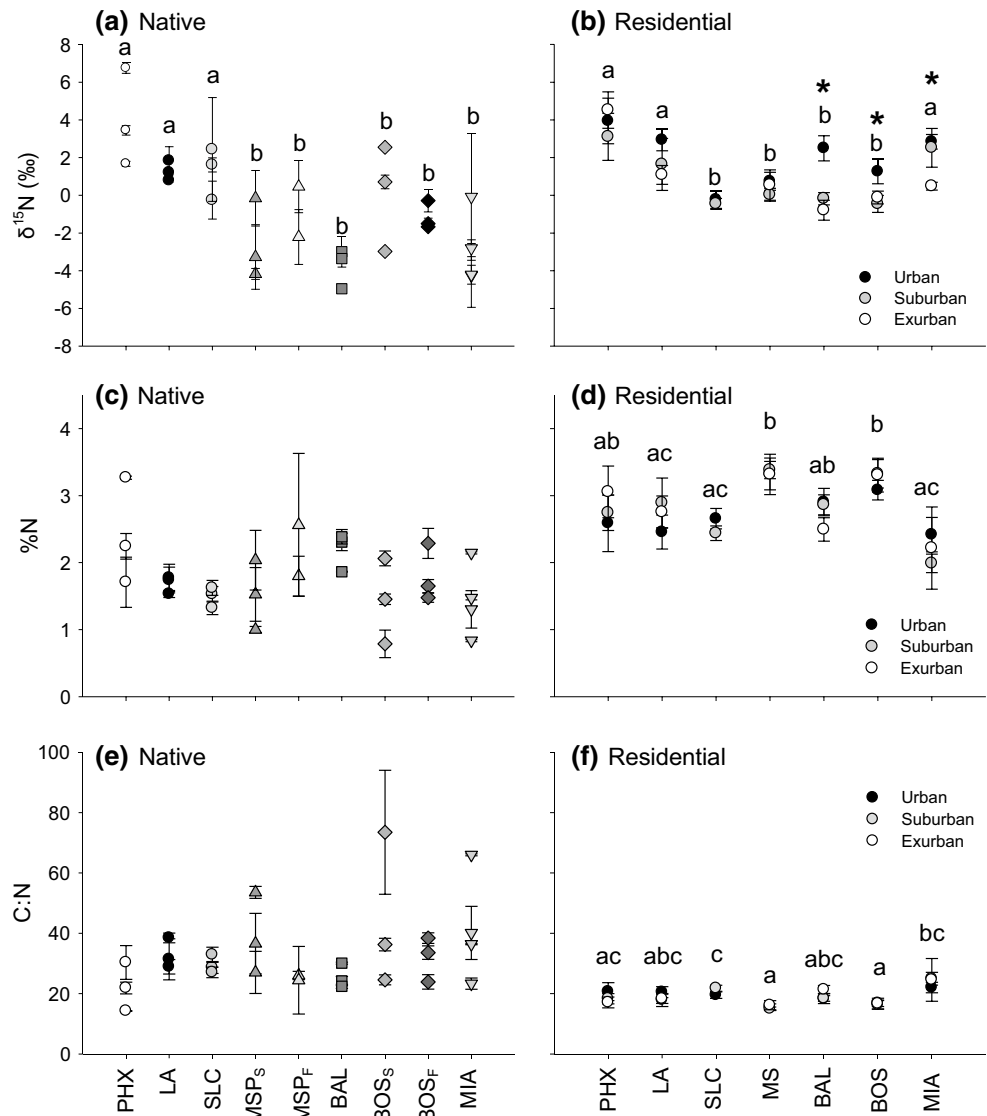
MAP (centimeters; $r^2 = 0.663$, $P = 0.016$; Fig. 2a, b). Furthermore, while we observed variability among the native and residential sites within cities, we also found significant differences between native and residential sites. In three cities, plant $\delta^{15}\text{N}$ was more enriched in residential lawns than in native sites: BAL (mean $\delta^{15}\text{N} = 0.1 \pm 0.35$ and -3.8 ± 0.60 ‰, respectively; $P = 0.001$), MIA (mean $\delta^{15}\text{N} = 2.4 \pm 0.43$ and -2.8 ± 0.98 ‰, respectively; $P = 0.0002$), and MSP (mean $\delta^{15}\text{N} = 0.5 \pm 0.31$ and -1.6 ± 0.77 ‰, respectively; $P = 0.03$). In contrast, plant $\delta^{15}\text{N}$ was more enriched in the native sites than in the residential lawns in SLC (mean $\delta^{15}\text{N} = 1.3 \pm 0.82$ and -0.4 ± 0.24 ‰, respectively; $P = 0.05$). There were no differences between residential and native plant $\delta^{15}\text{N}$ in PHX.

Similar to plant $\delta^{15}\text{N}$ patterns, we found significant differences in plant N (%) within native sites ($F_{8,18} = 3.14$, $P < 0.01$) and among residential yards ($F_{8,18} = 8.54$, $P < 0.001$). Plant %N in BOS and MSP residential lawns was significantly higher than in LA and SLC ($P < 0.05$), and plant %N in MIA residential lawns was significantly lower than in BAL, BOS, MSP, and PHX ($P < 0.05$; Fig. 1d). Similarly, plant molar C:N in MIA residential lawns was significantly higher than in BOS, MSP, and PHX ($P < 0.05$), and plant molar C:N in BOS and MSP was significantly lower than in SLC ($P < 0.05$; Fig. 1f). No significant differences were observed in plant N (%) or molar C:N among native sites (Fig. 1c, e). Similarly, in the residential and native sites, there were no significant relationships between plant N (%) or molar C:N and MAP or MAT (Fig. 2c–f). A direct comparison between residential and native sites revealed that plant N (%) was higher ($P < 0.05$) and molar C:N lower ($P < 0.02$) in residential lawns compared with native sites in all cities except PHX.

Plant N within cities

Patterns of plant $\delta^{15}\text{N}$ across the urbanization gradient differed in the seven cities (Fig. 1c). Plant $\delta^{15}\text{N}$ did not differ among urban, suburban, and exurban residential lawns in PHX and MSP. In LA, there was a trend toward more enriched plant $\delta^{15}\text{N}$ with increasing urban density class (mean $\delta^{15}\text{N}$ in urban = 2.9 ± 0.58 ‰, suburban = 1.6 ± 1.38 ‰, and exurban = 1.1 ± 1.31 ‰); however, this was not significant because of the high variability within the suburban class. In BAL and BOS, plant $\delta^{15}\text{N}$ was enriched in the urban density class ($P < 0.05$). Furthermore, post hoc analyses revealed urban plant $\delta^{15}\text{N}$ (mean $\delta^{15}\text{N} = 2.5 \pm 0.67$ ‰) was significantly enriched compared with suburban (mean $\delta^{15}\text{N} = -0.2 \pm 0.32$ ‰) and exurban (mean $\delta^{15}\text{N} = -0.8 \pm 0.53$ ‰) plant $\delta^{15}\text{N}$ ($P < 0.001$) in BAL, and urban plant $\delta^{15}\text{N}$ (mean $\delta^{15}\text{N} = 1.3 \pm 0.64$ ‰) was significantly enriched compared with suburban plant

Fig. 1a–f For each city, the mean plant stable nitrogen isotopic composition ($\delta^{15}\text{N}$; ‰), N (%), and molar C:N ($\pm 1\text{SE}$) are shown for residential and native sites. The native sites represent the mean $\pm 1\text{SE}$ for two transects at each site. The residential sites represent the mean and $\pm 1\text{SE}$ for the urban (black), suburban (gray), and exurban (white) density classes ($n = 4\text{--}10$ yards). In Minneapolis-St. Paul (MSP) and Boston (BOS), data shown are from two native site types [i.e., savanna ($\text{MSP}_s, \text{BOS}_s$) and forest sites ($\text{MSP}_f, \text{BOS}_f$)]. Cities are shown in order of increasing mean annual precipitation (MAP), from driest (Phoenix, AZ; PHX) to wettest (Miami, FL; MIA). Different letters represent significant differences among native sites or residential yards; asterisks represent significant differences in plant $\delta^{15}\text{N}$ across urban, suburban, and exurban density classes in residential yards. LA Los Angeles, CA; SLC Salt Lake City, UT; BAL Baltimore, MD



$\delta^{15}\text{N}$ (mean $\delta^{15}\text{N} = -0.5 \pm 0.45$ ‰; $P = 0.05$) in BOS. In MIA, there was a marginally significant trend in plant $\delta^{15}\text{N}$ across the urbanization gradient ($P = 0.07$) and post hoc analysis demonstrated significant differences between urban (mean $\delta^{15}\text{N} = 2.8 \pm 0.39$ ‰) and exurban (mean $\delta^{15}\text{N} = 0.5 \pm 0.21$ ‰) residential lawns ($P = 0.002$). No spatial patterns in plant %N (Fig. 1d) or plant molar C:N were observed across the urbanization gradient in the seven cities.

The percentage of homeowners who fertilized their yards in each city ranged from 55 to 83 %. The majority of homeowners who applied fertilizer in LA (87 %) and SLC (84 %) used inorganic fertilizers, while only a few households applied organic fertilizer (LA $n = 2$, SLC $n = 4$). Plant $\delta^{15}\text{N}$ was significantly enriched in lawns with organic compared to inorganic fertilizer additions in LA (mean $\delta^{15}\text{N} = 3.8 \pm 0.49$ and 1.3 ± 0.56 ‰,

respectively; $P = 0.04$) and SLC (mean $\delta^{15}\text{N} = 0.9 \pm 0.35$ and -0.8 ± 0.21 ‰, respectively; $P = 0.01$). Additionally, there was a marginally significant difference between plant N in lawns with organic (mean %N = 2.0 ± 0.21 %) and inorganic (mean %N = 2.9 ± 0.20 %; $P = 0.08$) fertilizer additions in LA.

Plant $\delta^{15}\text{N}$ was significantly depleted in fertilized versus unfertilized lawns in LA (mean $\delta^{15}\text{N} = 0.9 \pm 0.74$ and 2.9 ± 0.59 ‰, respectively; $P = 0.05$) and BOS (mean $\delta^{15}\text{N} = -0.3 \pm 0.36$ and 1.2 ± 0.40 ‰, respectively; $P = 0.02$; Fig. 3a). In MIA, there was a marginally significant difference between fertilized and unfertilized lawns (mean $\delta^{15}\text{N} = 1.7 \pm 0.36$ and mean $\delta^{15}\text{N} = 3.2 \pm 0.80$ ‰, respectively; $P = 0.10$). There were no differences in plant $\delta^{15}\text{N}$ between fertilized and unfertilized lawns in the other cities. While there was a trend toward greater plant N (%) in fertilized versus unfertilized lawns, except in MIA, this was

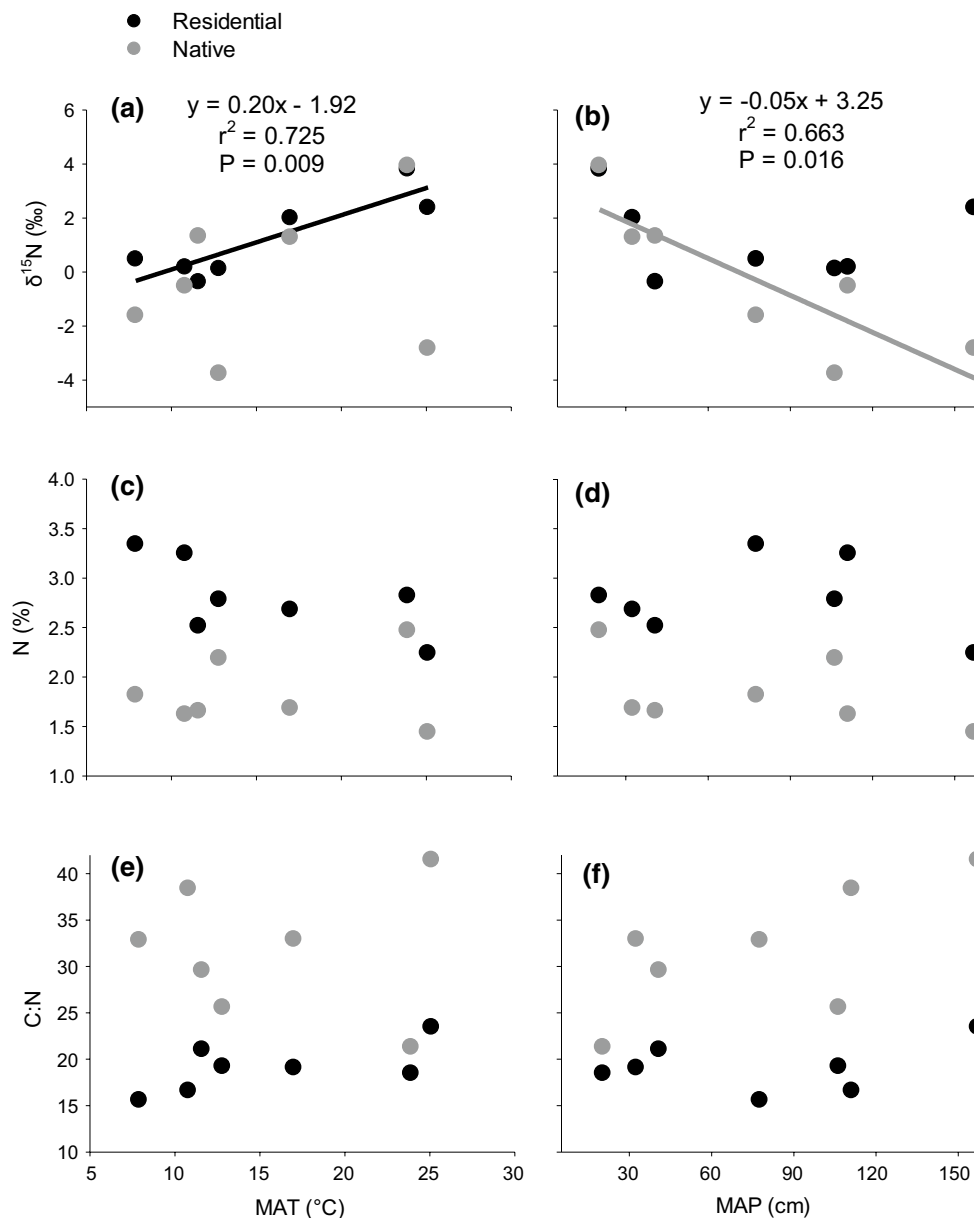


Fig. 2 Mean residential and native plant $\delta^{15}\text{N}$, %N, and molar C:N as a function of mean annual temperature ($^{\circ}\text{C}$; MAT) and MAP (cm)

only marginally significant in LA (mean %N = 3.0 ± 0.24 and 2.4 ± 0.21 %, respectively; $P = 0.07$; Fig. 3b). Similarly, there was a trend for lower plant molar C:N in fertilized versus unfertilized lawns in LA (C:N = 14.5 ± 1.05 and 17.8 ± 1.27 , respectively; $P = 0.06$; Fig. 3c).

Additional factors, such as SES, previous land use, and soil conditions, were not important explanatory variables for plant $\delta^{15}\text{N}$, %N, and molar C:N across the cities. In SLC, there were marginally significant differences in plant $\delta^{15}\text{N}$, %N, and molar C:N with SES in the urban density class, but no differences with SES in the suburban density class. SLC urban residential homes with lower SES

had more enriched plant $\delta^{15}\text{N}$ than homes with higher SES (mean $\delta^{15}\text{N}$ = 0.7 ± 0.65 and -1.2 ± 0.30 ‰, respectively; $P = 0.09$), greater %N (mean %N = 3.0 ± 0.22 and 2.4 ± 0.10 %, respectively; $P = 0.06$), and lower molar C:N (C:N = 17.1 ± 1.32 and 21.8 ± 0.80 , respectively; $P = 0.06$). No significant differences were observed in plant $\delta^{15}\text{N}$, %N, or molar C:N based on previous land use (BAL, BOS, or PHX), soil conditions (MIA or MSP), distance to the coast (LA), or yard landscaping type (PHX). Additionally, no significant correlations were found between plant $\delta^{15}\text{N}$ and plant %N or plant molar C:N in all cities.

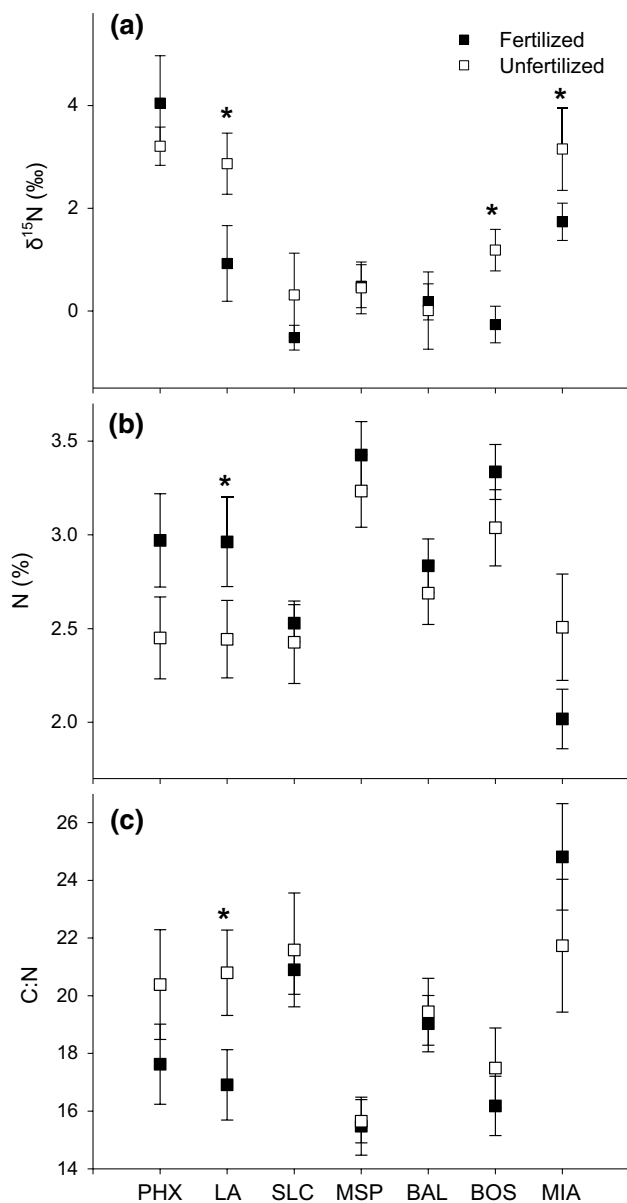


Fig. 3 Plant $\delta^{15}\text{N}$ (‰), % N, and molar C:N based on self-reported homeowner survey results answering the question whether they do fertilize (yes; black) or do not fertilize (no; white) their yard in PHX, LA, SLC, MSP, BAL, BOS, and MIA (± 1 SE). Cities are shown in order of increasing MAP, from driest (PHX) to wettest (MIA) city. Asterisk denotes significant difference between fertilized and unfertilized yards. For abbreviations, see Fig. 1

Plant N vs. housing age

Plant $\delta^{15}\text{N}$ significantly increased with housing age in BAL ($r^2 = 0.269$, $P = 0.02$), BOS ($r^2 = 0.276$, $P = 0.003$), and LA ($r^2 = 0.342$, $P = 0.007$; Fig. 4b, d, e), and there was no relationship in PHX, MIA, or MSP. Plant N (%) was marginally higher in lawns of newer homes in MIA ($r^2 = 0.180$, $P = 0.06$) and LA ($r^2 = 0.147$, $P = 0.09$;

Fig. 5b, f), and there was no relationship in PHX, MSP, BAL, or BOS. Similarly, in LA and MIA, plant molar C:N was marginally lower in lawns of newer homes ($r^2 = 0.144$, $P = 0.10$ and $r^2 = 0.164$, $P = 0.08$, respectively), and there was no relationship in PHX, MSP, BAL, or BOS.

Discussion

Across the seven cities, we found differences in residential and native plant $\delta^{15}\text{N}$ and plant N (%). The coldest cities showed the largest concentrations of N in lawn foliage and the warmest and driest cities had the highest plant $\delta^{15}\text{N}$, suggesting that the environmental impacts (coupled internal N cycling rates and N losses) of residential ecosystems are more marked in warmer and drier regions. In six cities, residential lawns had higher plant N (%) and in three cities enriched plant $\delta^{15}\text{N}$ relative to the native ecosystems, potentially due to differences in N cycling and/or plant species composition. Socio-ecological drivers at finer spatial scales within the city provide insight as to the potential mechanisms that caused enrichment of plant $\delta^{15}\text{N}$ and higher plant N in residential lawns relative to native ecosystems. Plant $\delta^{15}\text{N}$ was influenced by indirect (i.e., urban density, housing age) and direct (i.e., lawn fertilization) socio-ecological drivers, whereas the impacts of land-use legacies and soil conditions were less discernable.

Plant N across cities

Higher plant N (%) and enriched plant $\delta^{15}\text{N}$ in residential lawns compared with native ecosystems was consistent with higher N inputs from anthropogenic activities as measured in previous studies (Lovett et al. 2000; Bettez and Groffman 2013; Rao et al. 2014), and many of these activities are concentrated in urban ecosystems (Kaye et al. 2006; Grimm et al. 2008). Elevated fossil fuel combustion products around urban areas and roads were associated with enriched plant $\delta^{15}\text{N}$ (Ammann et al. 1999; Pearson et al. 2000; Saurer et al. 2004; Redling et al. 2013). Additionally, following high N fertilization applications, plant $\delta^{15}\text{N}$ enrichment occurred (Bijoor et al. 2008). The high plant N (%) observed in residential lawns compared to native ecosystems in our study may also result from increased N availability followed by increased soil N cycling relative to native ecosystems; i.e., N mineralization and nitrification resulting from altered urban soil conditions, such as high soil moisture from irrigation. As part of a larger project, we measured potential soil N mineralization and nitrification rates for six of the seven cities. We found residential yards had greater potential N mineralization rates in BAL ($P < 0.05$) and greater potential nitrification rates in BOS ($P < 0.05$) compared to the native sites (Groffman et al.,

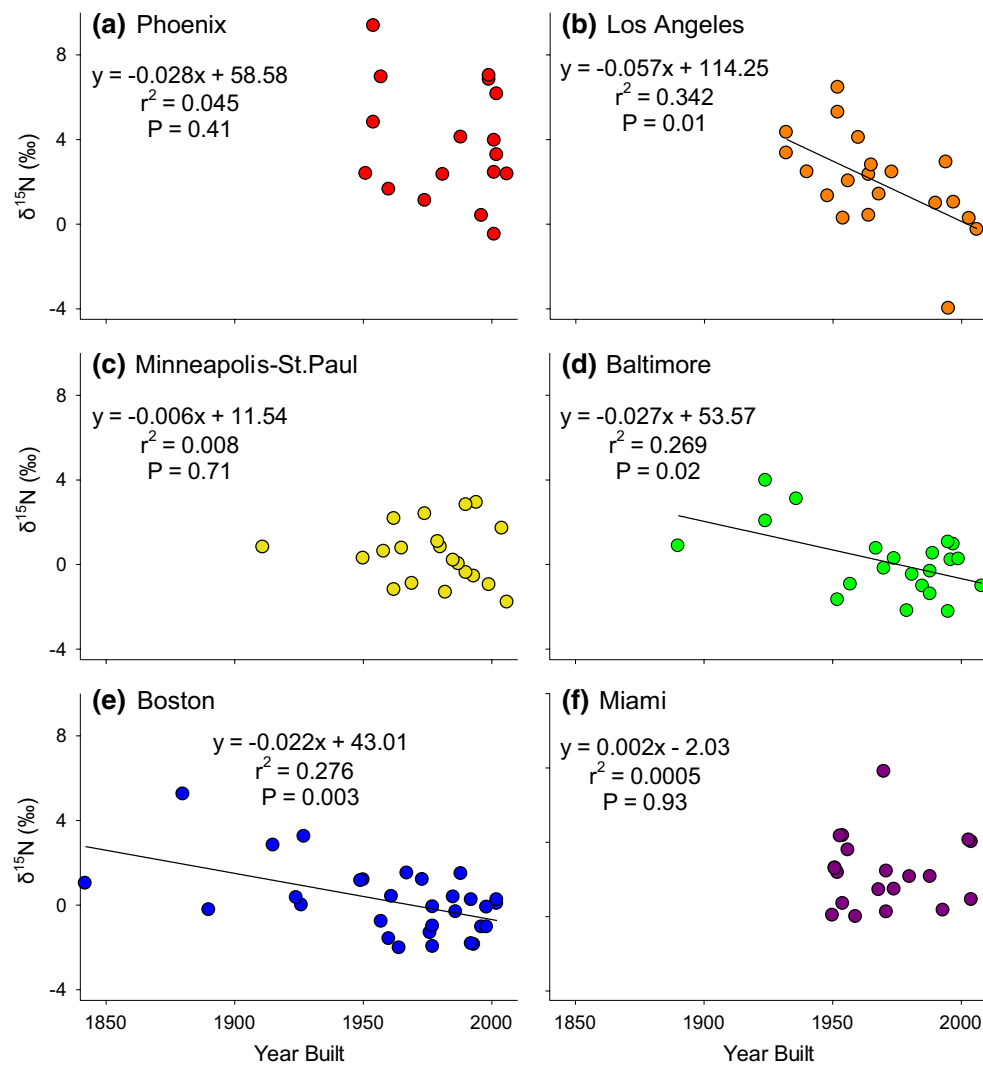


Fig. 4 Relationship between plant $\delta^{15}\text{N}$ (‰) and year the residence was built in **a** Phoenix, **b** Los Angeles, **c** Minneapolis-St. Paul, **d** Baltimore, **e** Boston, and **f** Miami (± 1 SE)

unpublished data), suggesting increased soil N cycling rates and associated N losses may be an important cause of the enriched plant $\delta^{15}\text{N}$ observed in the residential yards.

Patterns in residential plant $\delta^{15}\text{N}$ and N (%) and in native plant $\delta^{15}\text{N}$ observed across the seven cities were consistent with regional climatic differences. The positive association between residential plant $\delta^{15}\text{N}$ and MAT across the cities is consistent with previous studies that have reported enriched plant $\delta^{15}\text{N}$ in warm climates, likely due to greater N cycling and losses in warm soils (Evans and Ehleringer 1993; Jenerete et al. 2006). In the native sites, negative associations between plant $\delta^{15}\text{N}$ patterns and precipitation could occur because of either lower soil N cycling and N availability or complete denitrification of soil NO_3^- pools, which does not leave behind enriched soil N pools (Craine et al. 2009). In MIA, warm mesic conditions may result in

accelerated soil N cycling and high N losses, followed by low soil N availability and greater reliance on mycorrhizal N uptake which may produce depleted plant $\delta^{15}\text{N}$ (Craine et al. 2009; Hobbie and Högberg 2012; Högberg et al. 2011, 2013).

The high plant N (%) observed in the coldest cities (BOS, MSP) could be due to less microbial processing and leaching of soil organic matter and fertilizer. Conversely, in MIA residential lawns, low plant N (%) may be a result of large soil N losses in the warm, moist climate (Huyler et al. 2014). The enriched plant $\delta^{15}\text{N}$ in MIA residential lawns relative to the other mesic cities is consistent with high rates of soil N cycling and losses. We did not observe differences in native plant N (%) across the cities as there was high variability within several native ecosystems (e.g., PHX, MIA). This is most likely due to more varied life

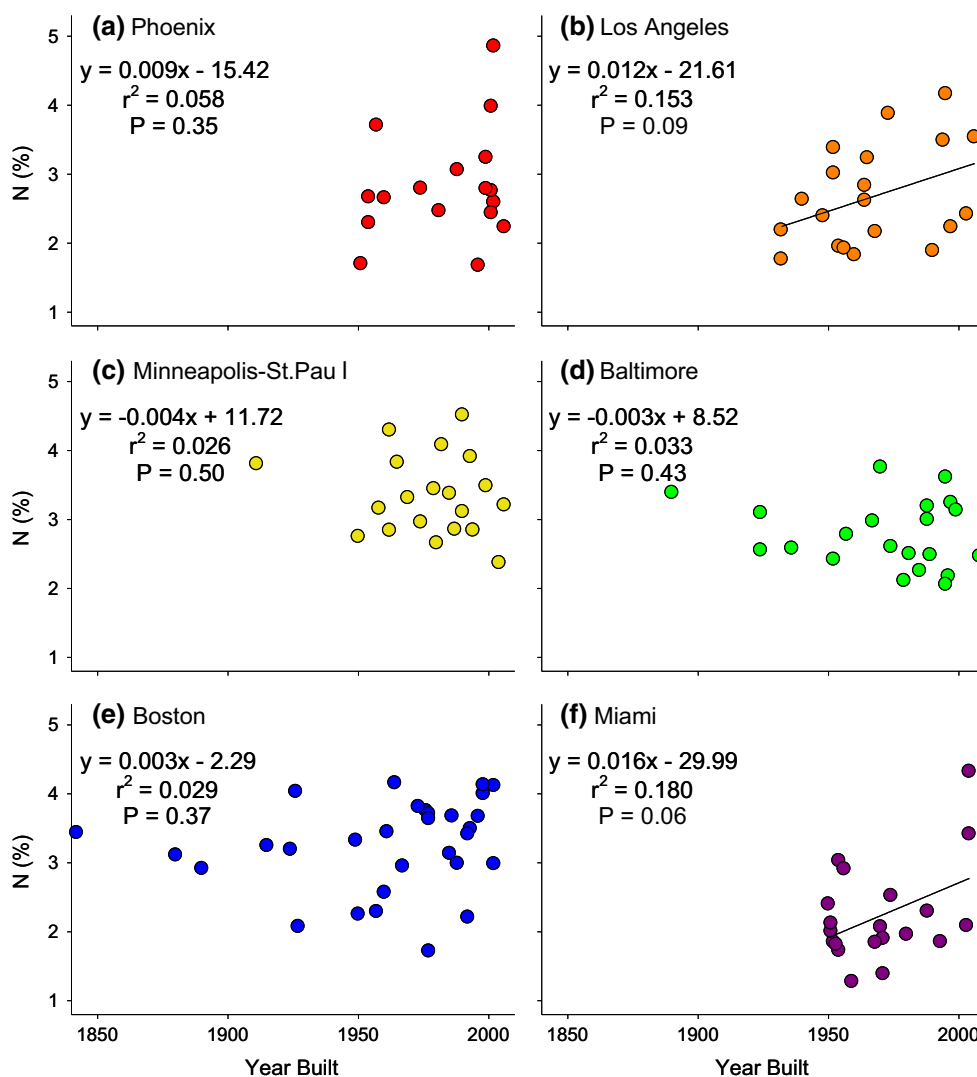


Fig. 5 Relationship between plant % N and year the residence was built in **a** Phoenix, **b** Los Angeles, **c** Minneapolis-St. Paul, **d** Baltimore, **e** Boston, and **f** Miami (± 1 SE)

forms and mycorrhizal status (e.g., trees, shrubs, herbaceous plants, or cacti) and more species-specific differences in plant N (%) in native ecosystems than in most residential lawns (e.g., grass and herbaceous plants).

Differences in residential plant $\delta^{15}\text{N}$ and N (%) across the seven cities may also result from differences in species and plant functional types. Less than a quarter of the residential lawns in the semi-arid, western cities (LA, SLC, and PHX) contained symbiotic N-fixing plant species compared to mesic cities where these species were more common (67–90 % of yards in BAL, BOS, and MIA). This was likely due to regional differences in management intensity: lawns in western, semi-arid cities require a high degree of resource inputs for establishment and maintenance (Milesi et al. 2005), and N fixation tends to be less abundant in fertilized ecosystems (Suding et al. 2005). However,

differences in plant $\delta^{15}\text{N}$ and N (%) across the residential sites did not follow patterns in N-fixing plant species (e.g., enriched $\delta^{15}\text{N}$ in MIA where 83 % of the yards have N-fixing species). Additionally, the proportion of C_3 plant species was not a determinant of plant N (%) across the cities. While the presence of these plant functional types in residential lawns did not have a strong influence on plant $\delta^{15}\text{N}$ or N (%) at the national scale, it is possible that unmeasured differences in abundance of functional groups could influence residential plant $\delta^{15}\text{N}$ and N (%) across our study sites.

Plant N within cities

Urban density gradients involve multiple factors (e.g., age of residence, regional N deposition) that could influence

plant N (%) and $\delta^{15}\text{N}$ (McDonnell and Pickett 1990; Carreiro and Tripler 2005). As expected, urban lawns had more enriched plant $\delta^{15}\text{N}$ than suburban and exurban lawns in several cities (Fig. 1c). There are two possible explanations for this enrichment: (a) regional N deposition, and (b) increased N cycling rates and associated N losses over time. Regional N deposition is greater in urban areas than in surrounding suburban and rural areas in most cities (Lovett et al. 2000; Bettez and Groffman 2013; Rao et al. 2014), and NO_3^- in wet deposition and NO_x sources typically have enriched $\delta^{15}\text{N}$ values (Kendall et al. 2007; Felix and Elliott 2014). Additionally, ample N supply in urban residential lawns either from regional sources or from fertilization may lead to coupled increased internal N cycling and N losses over time resulting in enriched soil N pools (Högberg 1997). Our finding that newer homes with increasing distance from the urban core had isotopically lighter plant $\delta^{15}\text{N}$ supported the hypothesis that older residential developments closer to the urban core have increased N cycling rates and associated N losses over time. We observed greater rates of potential N mineralization in residential yards of older homes in BOS ($r^2 = 0.232$, $P = 0.004$; Groffman et al., unpublished data) supporting increased N cycling rates and N losses over time as a plausible explanation for our observed enriched plant $\delta^{15}\text{N}$ in older residential yards. While plant $\delta^{15}\text{N}$ was more enriched in urban residential lawns as expected, we saw no evidence of differences in plant N (%) based on the gradient of urban density. As regional sources of N deposition decrease across urban density gradients (Lovett et al. 2000), it is possible that greater N fertilization inputs in newer suburban or exurban developments further from the urban core (Law et al. 2004) result in similar plant N (%) across urban density classes.

Homeowners have a direct impact on N cycling through fertilizer additions to residential lawns, which are widespread (Robbins et al. 2001). In this study, at least half of the homeowners in the seven cities applied fertilizers to their yard (55–83 % of homeowners). Inorganic N fertilizers have $\delta^{15}\text{N}$ values close to 0 ‰ (Bijoor et al. 2008) which is similar to lawn $\delta^{15}\text{N}$ in BAL, BOS, SLC, and MSP. In LA and SLC, lawns with organic fertilizer additions had significantly enriched plant $\delta^{15}\text{N}$ compared to lawns with inorganic fertilizer additions, as is expected from uptake of N that has undergone microbial transformations. Unfortunately, we do not have information about the type of fertilizer used in most yards in this study, which may explain variability in plant $\delta^{15}\text{N}$ patterns with respect to fertilization in BAL, MSP, and PHX. It is expected that the influence of fertilization on plant $\delta^{15}\text{N}$ enrichment will differ if plants directly utilize N in fertilizer ($\delta^{15}\text{N}$ close to 0 ‰) versus soil N that has undergone microbial transformations [enriched $\delta^{15}\text{N}$ (Stiegler et al. 2013)]. Future studies could

improve our ability to detect differences in plant $\delta^{15}\text{N}$ and N (%) in residential lawns by asking homeowners specific questions about lawn fertilization practices, such as type, amount, and timing.

While we hypothesized that SES might influence plant $\delta^{15}\text{N}$ and N (%) via differences in lawn care practices, we only observed patterns related to SES in SLC urban residential lawns. In SLC, the lawns in yards of more affluent homeowners had more depleted plant $\delta^{15}\text{N}$ than in yards of less affluent homeowners, potentially due to greater fertilizer inputs in more affluent residences. Similarly, residential lawns in wealthier neighborhoods had higher soil N ($\text{NO}_3\text{-N}$) concentrations and greater soil C storage, and it was postulated that wealthier homeowners more intensively manage their lawns with greater fertilizer inputs (Hope et al. 2005; Gough and Elliott 2012). Higher use of lawn fertilizer has been associated with SES (Robbins et al. 2001; Fraser et al. 2013). However, the proportion of households that fertilize their yards in this study suggests that we did not sample neighborhoods where homeowners were unable to afford fertilization inputs. It is also possible that the SES of the PRIZM classes used in this study were not distinct enough to result in significant differences in lawn plant N cycling.

Plant N vs. housing age

As expected, older homes in BAL, BOS, and LA had more enriched plant $\delta^{15}\text{N}$ (Fig. 4). Previous studies on lawn C and N cycling have found greater soil C (Golubiewski 2006; Gough and Elliott 2012) and N (Law et al. 2004; Raciti et al. 2011) in yards of older compared with newer housing developments. Lawn management practices such as fertilization and lawn clipping supplements increase the potential for soil C and N accumulation (Qian et al. 2003). Soil N accumulation in lawns suggests that these systems are capable of N retention (Gold et al. 1990; Petrovic 1990; Groffman et al. 2004; Raciti et al. 2011). This ability of residential lawns to retain N could have substantial implications for N runoff in urban landscapes; however, we do not know how long this retention capacity can be maintained. Since the natural abundance of plant $\delta^{15}\text{N}$ integrates soil N cycling and losses, plant $\delta^{15}\text{N}$ can provide insight about N retention capability over time. While three cities, BAL, BOS, and LA, had more enriched plant $\delta^{15}\text{N}$, there was no relationship between home age and plant $\delta^{15}\text{N}$ in MIA, MSP, or PHX, which may be due to the lack of homes built prior to 1950 in these cities (except for one home in MSP). Thus, the retention capacity of residential lawns may only last 50–60 years, which corresponds to studies that have shown that C accumulation potential in residential lawns persists for 40–50 years (Pouyat et al. 2009; Townsend-Small and Czimeczik 2010).

The absence of a relationship between plant N (%) and age of residence in BAL, BOS, MSP, or PHX may result from high N inputs from regional deposition or fertilizer additions supplying equivalent N for plant uptake in newer yards compared with older yards where N inputs may accumulate over time. A previous study found that N fertilizer application rates were greater in newer homes where the soil N concentrations were lower (Law et al. 2004). This may explain greater plant N (%) observed in newer yards in MIA and LA, if these homeowners applied more fertilizer than residents in older homes. Greater N inputs from atmospheric deposition and fertilizer additions may alleviate the need for plant N acquisition from mineralized soil organic matter pools, thus contributing to N retention (Raciti et al. 2008). Mechanistic studies of C and N cycling are needed to determine the degree to which N fertilizer applications and other management practices can promote C and N accumulation and retention in residential lawns. An improved mechanistic understanding of C and N accumulation in residential lawns can contribute to the development of efficient lawn management practices that adjust inputs of organic matter and fertilizer with lawn age.

Conclusion

This study provides support that human activities are contributing to altered plant N cycling in residential lawns across the USA. As expected, plant N (%) was higher in residential lawns than in the native ecosystems and plant $\delta^{15}\text{N}$ was more enriched in residential lawns in three cities, suggesting that urbanization and management practices may be increasing N cycling and loss across the continent. This pattern appears to be driven by processes that accumulate over time and the magnitude of this increase was variable among different households and cities. Older residential lawns had more enriched plant $\delta^{15}\text{N}$ possibly due to increased internal N cycling and ecosystem loss of N over time. The coldest cities showed the largest accumulations of N in lawn foliage and the warmest and driest cities had the highest plant $\delta^{15}\text{N}$, suggesting that the environmental impacts (N losses) of residential ecosystems are more marked in warmer and drier regions. There were significant differences in plant N (%) of fertilized vs. unfertilized lawns in only one city, suggesting that the type and amount of fertilizer may be highly variable. However, greater mean plant N (%) of fertilized lawns in six of the seven cities indicates fertilizer additions to lawns may be an important source of residential N as well as other urban N sources such as atmospheric deposition. While cumulative N losses may be high in residential lawns, there is significant potential for N cycling and retention in lawns. Understanding these factors will help to identify and improve lawn management practices across the US.

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