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ARTICLE

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Ecological homogenization of soil properties in the American residential macrosystem

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Abstract

The conversion of native ecosystems to residential ecosystems dominated by lawns has been a prevailing land-use change in the United States over the past 70 years. Similar development patterns and management of residential ecosystems cause many characteristics of residential ecosystems to be more similar to each other across broad continental gradients than that of former native ecosystems. For instance, similar lawn management by irrigation and fertilizer applications has the potential to influence soil carbon (C) and nitrogen (N) pools and processes. We evaluated the mean and variability of total soil C and N stocks, potential net N mineralization and nitrification, soil nitrite (NO_2^-)/nitrate (NO_3^-) and ammonium (NH_4^+) pools, microbial biomass C and N content, microbial respiration, bulk density, soil pH, and moisture content in residential lawns and native ecosystems in six metropolitan areas across a broad climatic gradient in the United States: Baltimore, MD (BAL); Boston, MA (BOS); Los Angeles, CA (LAX); Miami, FL (MIA); Minneapolis–St. Paul, MN (MSP); and Phoenix, AZ (PHX). We observed evidence of higher N cycling in lawn soils, including significant increases in soil NO_2^-/NO_3^-, microbial N pools, and potential net nitrification, and significant decreases in NH_4^+ pools. Self-reported yard fertilizer application in the previous year was linked with increased NO_2^-/ NO_3^- content and decreases in total soil N and C content. Self-reported irrigation in the previous year was associated with decreases in potential net mineralization and potential net nitrification and with increases in bulk density and pH. Residential topsoil had higher total soil C than native topsoil, and microbial biomass C was markedly higher in residential topsoil in the two driest cities (LAX and PHX). Coefficients of variation for most biogeochemical metrics were higher in native soils than in residential soils across all cities, suggesting that residential development homogenizes soil properties and processes at the continental scale.

KEYWORDS
biogeochemistry, carbon, ecological homogenization, fertilization, irrigation, land-use change, native soils, nitrate, nitrogen, residential landscapes, soil chemistry

INTRODUCTION

Urbanization has substantially accelerated land-use change in the United States since the mid-20th century. From 1949 to 2012, the total coverage of urban land in the United States increased from 6 to 28 million ha, predominately at the expense of agricultural land (Bigelow & Borchers, 2017). This growth of the urban area is fueled by suburban growth, which can be tracked through the increase in coverage of turfgrass lawns. Estimates suggest that there is 163,800 km^2 of turfgrass coverage within the contiguous United States—an area three times greater than that of any irrigated crop (Milesi et al., 2005).

Urbanization has well-documented effects on biological diversity, but less attention has been paid to effects on soils (Herrmann et al., 2020; Pouyat et al., 2015; Schmidt et al., 2017; Szlavecz et al., 2011) despite potentially large changes to soil biogeochemistry. New housing development often involves the removal and/or replacement of original topsoil (Yan et al., 2019) and has been linked to reductions in the functional diversity of soil nematode populations (Pavao-Zuckerman & Coleman, 2007), as well as fungal abundance and diversity (Abrego et al., 2020). Soil sealing has increased over time as a result of urbanization, resulting in many fertile, high-quality soils being covered by impervious surface (Li et al., 2015). All of these soil-altering processes have a wide-reaching impact, such that in the United States, at least 31 documented soil series have become extinct, with 27 lost to agriculture and 4 lost to urbanization (Amundson et al., 2003).

Landscaping behaviors in terms of plant selection, fertilizer usage, and irrigation practice strongly shape urban soils. Social norms that influence these landscaping behaviors in urban, suburban, and exurban settings have become increasingly similar despite vast differences in native vegetation, geology, and regional climate across the United States. These norms and behaviors (Nassauer, 2017; Robbins, 2007) drive a process of urban ecological homogenization where continental-scale land-use change yields residential landscapes and ecosystems that are more similar to each other than the diverse native ecosystems that they replaced (Groffman et al., 2014; Pouyat et al., 2015). This homogenization has been documented for hydrography (Steele et al., 2014), microclimate (Hall et al., 2015), plants (Padullés Cubino et al., 2019; Pearse et al., 2018; Wheeler et al., 2017), insects, and birds (Lerman et al., 2021). Homogenization has been documented for a variety of soil variables including C (Herrmann et al., 2020; Pouyat et al., 2015) and N concentrations and $\delta^{15}N$ (Trammell et al., 2020), particle size (Herrmann et al., 2020), and soil chemistry (Pouyat et al., 2008).

While previous research has explored urbanization's effects on and homogenization of total soil C and N (Trammell et al., 2020), less research has examined the effects on specific biogeochemical processes related to C and N cycling. As these processes control the partitioning of N and C among soil pools, as well as the potential environmental impacts of these elements such as travel to ground or surface waters, understanding how their dynamics have been altered remains of crucial concern (Kaye et al., 2006). While the conversion of native ecosystems to agricultural systems has resulted in marked declines in soil C (Qin et al., 2016) and increases in gaseous and hydrologic losses of N (Galloway et al., 2003) across the globe, the ecosystem consequences of conversion to residential lawns are less certain.

In this study, we evaluated biogeochemical metrics related to C and N cycle processes in soils (inorganic N pools, potential net N mineralization and nitrification rates, total and microbial biomass C and N contents, and microbial respiration), as well as soil quality metrics including bulk density, soil moisture, and pH in residential and native reference soils in six US cities: Baltimore, MD (BAL); Boston, MA (BOS); Los Angeles, CA (LAX); Miami, FL (MIA); Minneapolis–St. Paul, MN (MSP); and Phoenix, AZ (PHX), with these cities encompassing a wide range of climatic and natural soil conditions. We assessed the extent to which homogenization is occurring among these metrics, as well as site-specific patterns of alterations in biogeochemical properties. We tested three hypotheses: (1) residential soils will have elevated N pools and cycling rates compared with native soils and that self-reported yard fertilization, irrigation, and pesticide usage will be associated with such increases; (2) residential soils will have higher levels of microbial biomass C and respiration than native soils; and (3) variation in biogeochemical metrics will be higher in native than in residential soils across the continent. The rationale for these hypotheses is that yard management, that is, the use of water, fertilizer, and pesticides, is designed to increase N, water availability, and grass growth, which should result in elevated pools and cycling rates of N, increased flow of C to support microbial biomass and respiration, and homogenization of biogeochemical metrics in lawns relative to native ecosystems at the continental scale.

METHODS

Site selection and soil collection

More than 100,000 households were contacted between 21 November 2011 and 29 December 2011 to determine whether they had individuals over the age of 18 and front and back yards. After this list of households was reduced to ~13,500 homes, a voluntary telephone survey of 9480 individuals was conducted during which permission was obtained for physical sampling of residential landscapes (Locke et al., 2019). We used the potential index for zip code market system (Claritas, 2008) to classify households based on socioeconomic status (low, medium, and high) and population density (urban, suburban, and exurban). Respondents were asked about their yard management practices, including if they had applied fertilizers, pesticides, or irrigation to any part of their yards in the last year (Polsky et al., 2014).

Twenty-one to 31 residences in each city were selected for field visits that included extraction of two 1-m depth soil core from both front and back yards. Sampling occurred in either 2012 or 2013, with collection dates varying by city. Soil sampling occurred over 4 separate days between May and June 2012 for BAL, over 7 separate days in August 2012 for MSP, over 6 separate days in August 2012 for BOS, over 9 separate days in October 2012 for MIA, over 19 separate days between February and April 2013 for PHX, and over 11 separate days in May 2013 for LAX. Cores were extracted using a 3.3-cm-diameter soil corer enclosed in plastic sleeves with end caps, put into coolers, and transported to the laboratory where they were stored at 4°C until further processing.

Soil samples were also collected from native reference ecosystems in each city. Padullés Cubino et al. (2020) provided descriptions of the histories of the native reference areas sampled. Biomes are described by Trammell et al. (2016) and are reproduced within Table 1. In BAL, the native reference site consisted of mature southern Piedmont forest over 75 years of age. In BOS, it was mature hardwood oak-dominated forest over 100 years of age. In LAX, the native reference site was the largest and possibly the only regionally remaining contiguous remnant of coastal sage scrub. In MIA, it was coastal scrub located within natural upland areas within the city. In MSP, sampling was from three major ecosystem types in the region including oak savanna, tallgrass and bluff prairie, and maple-basswood forest. In PHX, the native reference site was Sonoran Desert scrublands without a history of development or agriculture.

Results from samples collected from urban, suburban, and exurban yards in each of the six cities were combined into one "residential" category for comparison with results from the native reference ecosystems. Previous land-use history for residential samples varied both by and within cities but was often unknown. The previous usage included agricultural lands, deserts, forests, and pastures. Sample sizes for native and residential soils collected are reported in Appendix S1: Table S1.

TABLE 1 Descriptive information for each sampled city.

City	Coordinates	Native biome	USDA hardiness zone	Average annual extreme low temperature (°C)	1981–2010 Mean annual temperature (°C) Normal	Min. normal	Max. normal	Total annual precipitation normal (cm)
Baltimore, MD	39°17′25.4″ N, 76°36′43.9″ W	Southern Piedmont forest	Zone 7b	−15 to −12.22	12.78	7.28	18.33	106.38
Boston, MA	42°21′36.3″ N, 71°3′32.0″ W	Northern hardwood forest	Zone 6b	−20.56 to −17.78	10.78	6.67	14.83	111.18
Los Angeles, CA	34°3′8.0″ N, 118°14′37.3″ W	Southern California coastal scrub	Zone 10b	1.67 to 4.45	18.17	12.94	23.89	31.14
Miami, FL	25°45′42.1″ N, 80°11′30.4″ W	South Florida costal scrub	Zone 11a	4.44 to 7.22	25.11	21.11	20.06	157.23
Minneapolis–St. Paul, MN	44°57′3.9″ N, 93°19′26.0″ W	Oak savanna and north central interior forest	Zone 4b	−31.67 to −28.89	7.89	2.89	12.89	77.75
Phoenix, AZ	33°26′54.2″ N, 112°4′26.5 W	Sonora–Mojave Desert scrub	Zone 9b	−3.89 to −1.11	23.89	17.44	30.33	20.40

Note: Latitude and longitude data come from Latitude and Longitude Finder (n.d.). Native biome information comes from the U.S. Geological Survey (2008), as compiled in Trammell et al. (2016). Hardiness zone and average annual extreme low temperatures come from USDA (2012). The average temperature, average minimum/maximum temperatures, and average annual precipitation come from the National Oceanic and Atmospheric Administration (2020) utilizing the overall city area data for each city and information found in the daily/monthly normals displayed under the NOWData.
Abbreviations: max., maximum; min., minimum; USDA, US Department of Agriculture.

Laboratory analysis

Soil cores were divided into segments of 0–10 cm, 10–30 cm, 30–60 cm, and 60+ cm. While most core sections were complete, some did not reach the ultimate depth of their categorical group. For the deepest core segments, many samples terminated before 90 cm depth; hence, all the final depth increment samples are categorized as 60+. Most analyses in this study focused on the 0–10 cm depth core segments as compaction of the cores was an issue at deeper depths, as evidenced by very high bulk density values.

Samples were hand-sorted to remove coarse roots and rocks (>2 mm). The roots and rocks were dried at 105°C, and rock volume was estimated using an assumed density of 2.7 g cm^{-3}. Soil moisture was measured gravimetrically by drying for 48 h at 60°C.

Bulk density was calculated as (total dry mass − rock mass)/(total volume − rock volume). Bulk density values were used to convert pool and flux values to an areal basis (in grams per square meter). Soil pH was assessed using a 1:1 soil–water mix with a pH electrode.

Soil NO_2^-/NO_3^- and NH_4^+ were extracted by adding 40 ml of a 2 M KCl solution to 10 g of soil, agitating, and then allowing the solutions to sit for 2 h. Each sample was filtered, and a Lachat flow injection analyzer was used to colorimetrically analyze each sample for NO_2/NO_3^- and NH_4^+ concentrations.

Potential net N mineralization and nitrification, and microbial respiration were measured by incubating soil samples at field moisture for 10 days in a laboratory setting in lidded glass jars fitted with septa (Robertson et al., 1999). Gas samples were extracted by syringe from each jar. Carbon dioxide (CO_2) content was determined via thermal conductivity gas chromatography and used to quantify microbial respiration activity. Total inorganic N accumulation was used to quantify potential net N mineralization, and NO_3^- accumulation was used to quantify potential net nitrification.

Microbial biomass C and N contents were measured using the chloroform fumigation–incubation method (Jenkinson & Powlson, 1976). In this assay, 20 g of moist soil sample was fumigated to kill microbial organisms. Afterward, these soil samples were inoculated with 0.1 g fresh soil and incubated for 10 days and sampled as described above for the mineralization, nitrification, and respiration incubations. Production of CO_2 and inorganic N from these fumigated and incubated soils was measured as described above to quantify microbial biomass C (assuming 41% extraction efficiency) and microbial biomass N, respectively.

Soil total C and N content was measured with a Carlo Erba Model 1110 elemental analyzer, and data were reported and discussed by Trammell et al. (2020). These data are used in this study only to provide context for the biogeochemical metrics reported here.

Statistical analysis

The data used in this paper are publicly available via the Environmental Data Initiative (EDI) Data Portal (Groffman, 2019). All statistical analyses were conducted in R (version 3.6.2; RStudio Team, 2019). *Q–Q* plots were generated to visualize the data and determine normality. Levene's test was utilized to determine homogeneity of variance. Because data were frequently not normally distributed, Mann–Whitney *U* tests were utilized to determine significant differences ($\alpha = 0.05$) between residential and native sites over all cities, and between native and residential sites within any given city, as well as to evaluate the effects of each of the social drivers (fertilization, irrigation, and pesticide application) on the measured biogeochemical metrics. The results of these tests are displayed using significance values in Figures 1–4. To make comparisons between all cities, as well as for native and reference sites, a Kruskal–Wallis one-way ANOVA was conducted, followed by a Dunn test with a Bonferroni adjustment for post hoc analysis. These results are displayed as compact letter displays in Figures 2 and 4. The coefficient of variation of the means for all residential and all native soils was calculated for each variable. This measure was used to assess differences in variability between native and residential soils with a reduction in variability taken as evidence for homogenization. All values are reported with means and standard errors.

RESULTS

Soil bulk density, moisture content, and pH

Bulk density in topsoil (0–10 cm depth) did not differ between residential and native soils averaged across all six cities (Figure 1). Three cities showed significant differences in bulk density between residential and native topsoil. In PHX and LAX, the two driest cities, bulk density was significantly lower in residential topsoil than in native topsoil (PHX: residential = 1.07 g cm^{-3} ± 0.03, native = 1.41 g cm^{-3} ± 0.09, $p \leq 0.05$; LAX: residential = 0.76 g cm^{-3} ± 0.03, native = 1.01 g cm^{-3} ± 0.07, $p \leq 0.05$). In BOS, bulk density was significantly increased in residential topsoil compared with that in native topsoil (residential = 0.87 g cm^{-3} ± 0.03, native = 0.61 g cm^{-3} ± 0.07, $p \leq 0.05$; Figure 2). Bulk density was higher in yards with self-reported irrigation (irrigation = 0.96 g cm^{-3} ± 0.02, no irrigation = 0.89 g cm^{-3} ± 0.05, $p \leq 0.05$). There was no significant effect of fertilizer or pesticide application on bulk density ($p > 0.05$) (Figure 3).

Topsoil pH was significantly higher ($p \leq 0.001$) in residential (7.16 ± 0.08) than in native topsoil across all cities (6.45 ± 0.18; Figure 1). Although all cities individually displayed elevated pH in their residential topsoil, the difference was only significant in BAL (residential = 6.37 ± 0.11, native = 5.05 ± 0.08, $p \leq 0.01$), LAX (residential = 7.49 ± 0.11, native = 6.16 ± 0.27, $p \leq 0.01$), and PHX (residential = 8.77 ± 0.05, native = 8.10 ± 0.25, $p \leq 0.05$) on a city-by-city basis (Figure 2). Topsoil pH was found to be increased ($p \leq 0.0001$) in yards with self-reported irrigation and pesticide application (irrigation = 7.23 ± 0.08, no irrigation = 6.34 ± 0.16; pesticide application = 7.49 ± 0.11, no pesticide application = 6.80 ± 0.10; Figure 3).

Soil moisture was significantly higher ($p \leq 0.01$) in residential (14.63% ± 0.41) than in native (11.36% ± 1.48) topsoil across all cities (Figure 1). Soil moisture was greater in residential soils than in native topsoil in all cities, except in BAL, the city with the highest native soil moisture. In BAL, soil moisture content was significantly decreased ($p \leq 0.01$) in residential topsoil (16.64% ± 0.55) compared with that in native topsoil (23.50% ± 1.91). The increase in moisture was only significant in the two driest cities, which had the largest differences. In PHX, residential topsoil had a moisture content of 9.81% ± 0.62, whereas the native topsoil had a moisture content of 2.67% ± 0.92 ($p \leq 0.01$). In LAX, this increase in moisture content was even more pronounced with residential topsoil having a moisture content of 16.57% ± 1.28 than with native soils having a moisture content of 3.50% ± 0.99 ($p \leq 0.01$; Figure 2).

Nitrogen pools and processes

Soil pools of both NO_2^-/NO_3^- and NH_4^+ at 0–10 cm were significantly different between native and residential topsoil. Residential topsoil had greater NO_2^-/NO_3^- than native topsoil (residential = 1.41 g N m^{-2} ± 0.10, native = 0.68 g N m^{-2} ± 0.31, $p \leq 0.0001$), yet residential topsoil had lower NH_4^+ content than native topsoil (residential = 0.19 g N m^{-2} ± 0.02, native = 0.29 g N m^{-2} ± 0.04, $p \leq 0.001$; Figure 1). Topsoil NH_4^+ pools generally were much lower than NO_2^-/NO_3^- pools, yielding greater total inorganic N pools in residential than in native topsoil.

There were two significant differences in soil NH_4^+ across cities (Figure 4) with MIA and BAL having higher

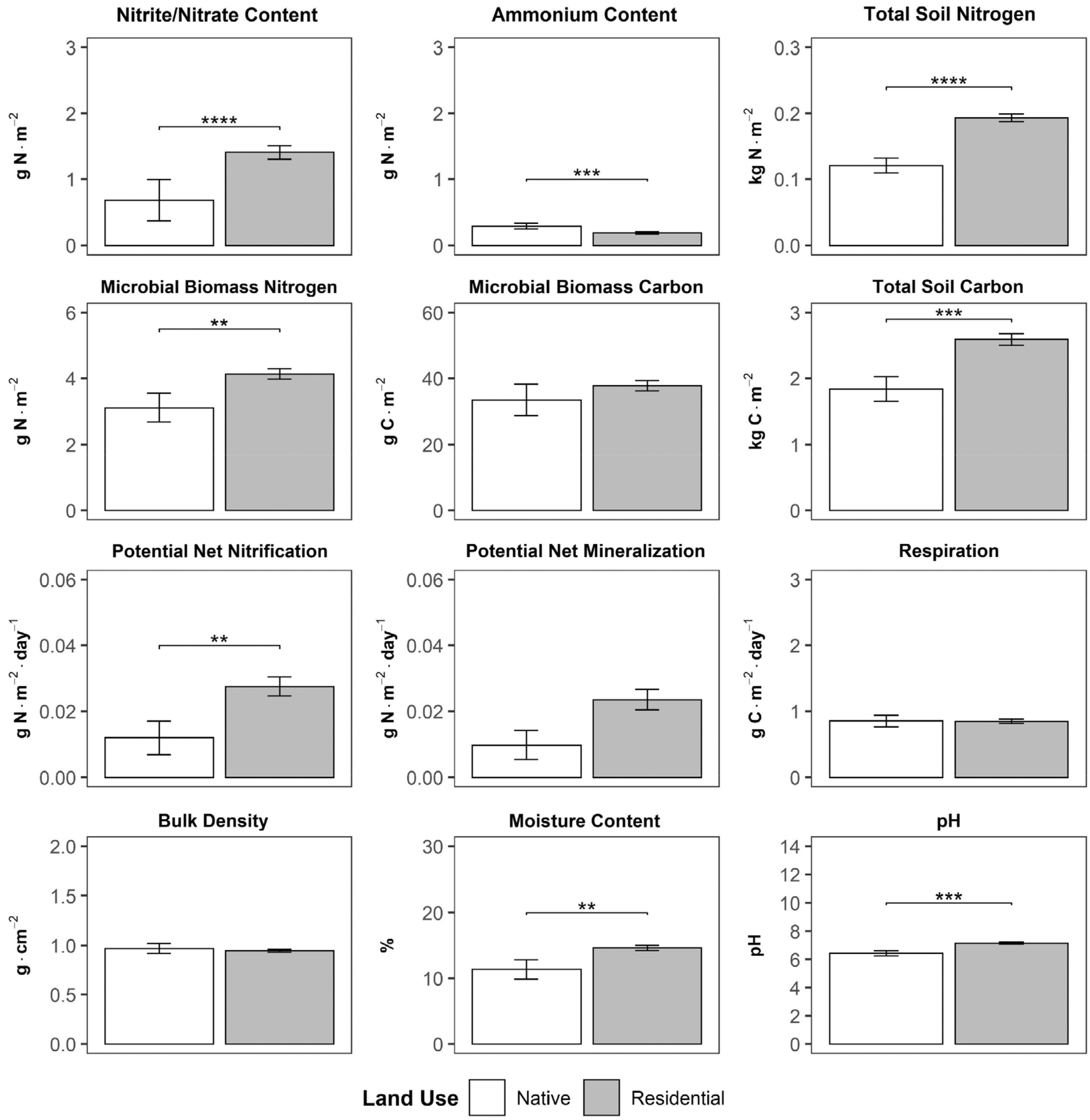

FIGURE 1 Soil C and N cycle variables in topsoil (0–10 cm depth) in residential and native reference soils. Values are means (with ±1 SE) of all residential and native sampling sites in six cities across the continental United States. Asterisks indicate significant differences between native and residential soils. ****$p \leq 0.0001$, ***$p \leq 0.001$, **$p \leq 0.01$, *$p \leq 0.05$.

NH_4^+ content in native topsoil than in residential topsoil (MIA residential = 0.22 g N m^{-2} ± 0.06, native = 0.45 g N m^{-2} ± 0.14, $p \leq 0.05$; BAL residential = 0.26 g N m^{-2} ± 0.04, native = 0.70 g N m^{-2} ± 0.10, $p \leq 0.01$). In contrast, NO_2^-/NO_3^- was significantly greater in residential topsoil in BAL (residential = 0.77 g N m^{-2} ± 0.11, native = 0.16 g N m^{-2} ± 0.07, $p \leq 0.01$), BOS (residential = 1.22 g N m^{-2} ± 0.14, native = 0.02 g N m^{-2} ± 0.00, $p \leq 0.001$), MIA (residential = 2.03 g N m^{-2} ± 0.15, native = 0.69 g N m^{-2} ± 0.34, $p \leq 0.01$), and MSP (residential = 1.28 g N m^{-2} ± 0.16, native = 0.30 g N m^{-2} ± 0.08, $p \leq 0.001$; Figure 4).

While NH_4^+ content did not differ significantly ($p > 0.05$) with irrigation, fertilization, or pesticide application, NO_2^-/NO_3^- content was higher in yards with self-reported fertilizer (fertilizer application = 1.50 ± 0.12, no fertilizer = 1.26 ± 0.18, $p \leq 0.05$; Figure 5) and pesticide application (pesticide application = 1.59 ± 0.14, no pesticide application = 1.21 ± 0.15, $p \leq 0.05$; Figure 5) within the previous year.

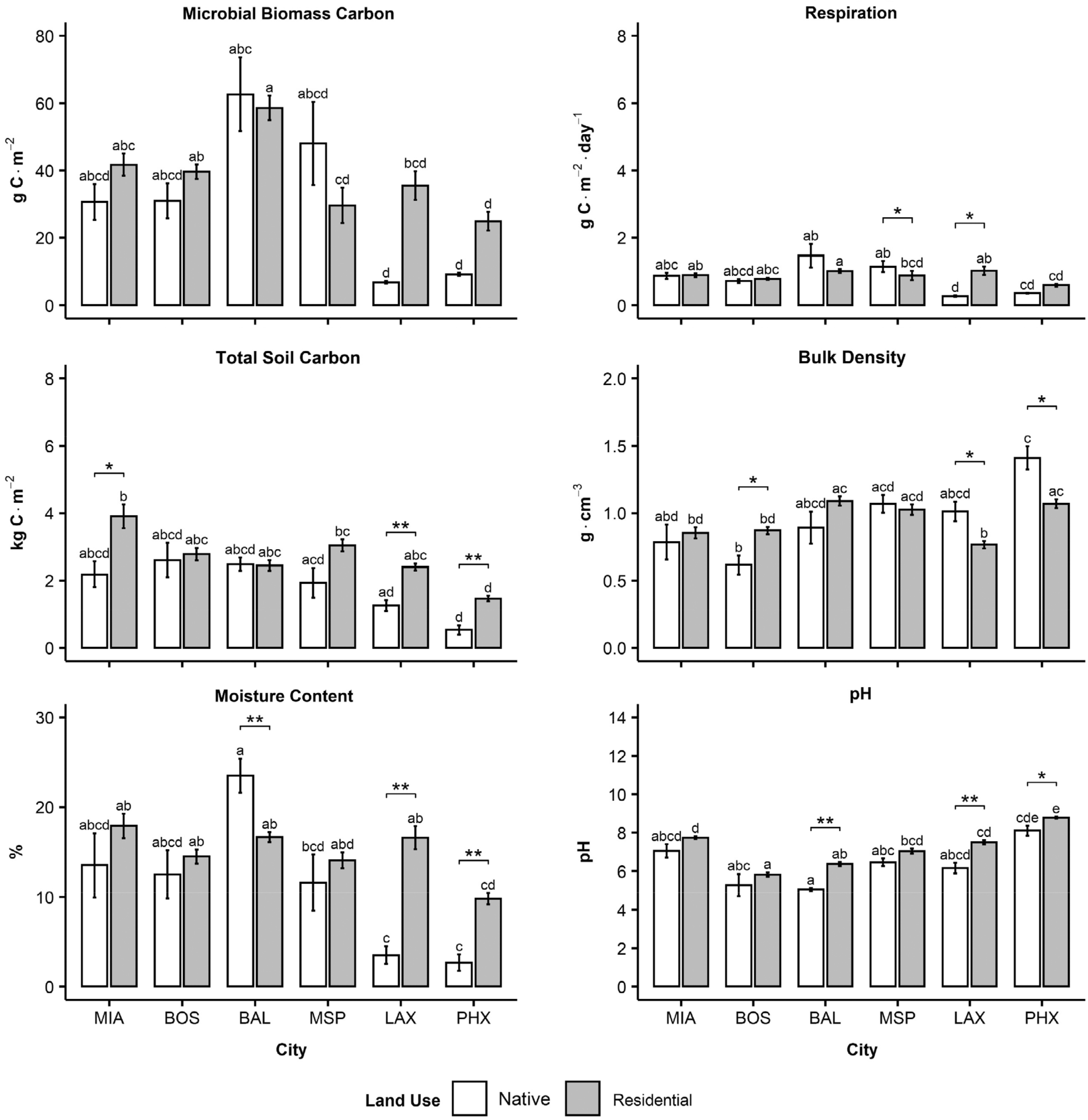

FIGURE 2 Soil C variables, moisture content, bulk density, and pH in topsoil (0–10 cm depth) in residential and native reference soils in six cities across the continental United States. Values are means (±1 SE) of all residential and native sampling sites in each city. Cities are ordered from wettest to driest along the *x*-axis. Asterisks indicate significant differences between native and residential soils. ****$p \leq 0.0001$, ***$p \leq 0.001$, **$p \leq 0.01$, *$p \leq 0.05$. Bars with different letters are significantly different at $p < 0.05$. MIA, Miami, FL; BOS, Boston, MA; BAL, Baltimore, MD; MSP, Minneapolis–St. Paul, MN; LAX, Los Angeles, CA; PHX, Phoenix, AZ.

When averaged across cities, potential net nitrification was significantly higher in residential topsoil (0–10 cm) than in native topsoil (residential $= 0.03$ g N m^{-2} day$^{-1} \pm 0.00$, native $= 0.01$ g N m^{-2} day^{-1} $\pm$ 0.01, $p \leq 0.01$; Figure 1). Potential net mineralization did not differ among residential and native topsoil when averaging all cities ($p > 0.05$; Figure 1). On a city-by-city basis, only BOS displayed significant differences in potential net nitrification between native and residential topsoil (residential $= 0.04$ g N m^{-2} day$^{-1} \pm 0.00$, native $= 0.00$ g N m^{-2} day$^{-1} \pm 0.00$, $p \leq 0.01$; Figure 4). In an analysis across all cities, the self-reported use of irrigation and pesticide

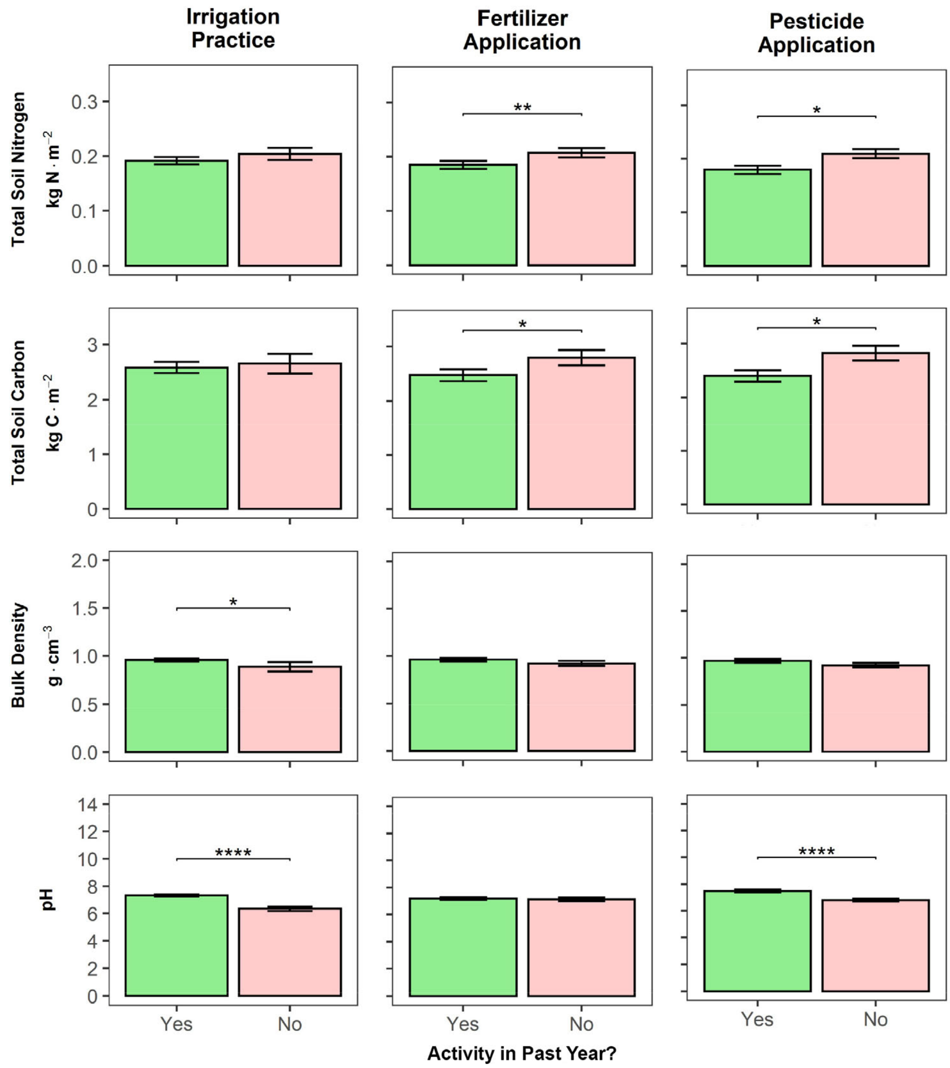

FIGURE 3 Total soil N and C content, bulk density, and pH in topsoil (0–10 cm) as affected by the self-reported use of fertilizer, irrigation, and pesticides in residential soils. Values are means (±1 SE) of all residential sampling sites in six cities across the continental United States. Asterisks indicate significant differences between bars. ****$p \leq 0.0001$, ***$p \leq 0.001$, **$p \leq 0.01$, *$p \leq 0.05$.

application was associated with significantly lower net nitrification (irrigation $= 0.02$ g N m^{-2} day^{-1} ± 0.00, no irrigation $= 0.04$ g N m^{-2} day^{-1} ± 0.00, $p \leq 0.001$; pesticide application $= 0.02$ g N m^{-2} day^{-1} ± 0.00, no pesticide application $= 0.03$ g N m^{-2} day^{-1} ± 0.00, $p \leq 0.05$). Similarly, both irrigation practice (irrigation $= 0.02$ g N m^{-2} day^{-1} ± 0.00, no irrigation $= 0.03$ g N m^{-2} day^{-1} ± 0.00, $p \leq 0.001$) and pesticide application (pesticide application $=$ 0.02 g N m^{-2} day^{-1} ± 0.00, no pesticide application $= 0.03$ g N m^{-2} day^{-1} ± 0.01, $p \leq 0.01$; Figure 5) were associated with significantly lower potential net mineralization. Self-reported fertilizer application had no effect on either potential net nitrification or potential net mineralization ($p > 0.05$; Figure 5).

When averaging the cities, microbial biomass N was higher in residential topsoil than in native topsoil (residential $= 4.14$ g N m^{-2} ± 0.16, native $= 3.12$ g N m^{-2} ± 0.43, $p \leq 0.01$; Figure 1). On a city-by-city basis, only LAX demonstrated a significant difference between native (0.88 g N m^{-2} ± 0.37) and residential (3.67 g N m^{-2} ± 0.55) topsoil ($p \leq 0.05$; Figure 4). Neither self-reported irrigation, fertilization, nor pesticide application was associated with differences in microbial biomass N between residential and native topsoil ($p > 0.05$, data not shown).

When averaging all cities, total soil N pools (data from Trammell et al., 2020) were significantly higher in residential topsoil than in native topsoil (residential $=$ 0.19 kg N m^{-2} ± 0.01, native $= 0.12$ kg N m^{-2} ± 0.01, $p \leq 0.0001$; Figure 1). On a city-by-city basis, total soil N was increased in residential topsoil compared with that in native topsoil in all six cities. However, this increase was only significant in MIA (residential $= 0.25$ kg N m^{-2} ± 0.02, native $= 0.10$ kg N m^{-2} ± 0.02, $p \leq 0.01$), MSP (residential $=$ 0.25 kg N m^{-2} ± 0.01,

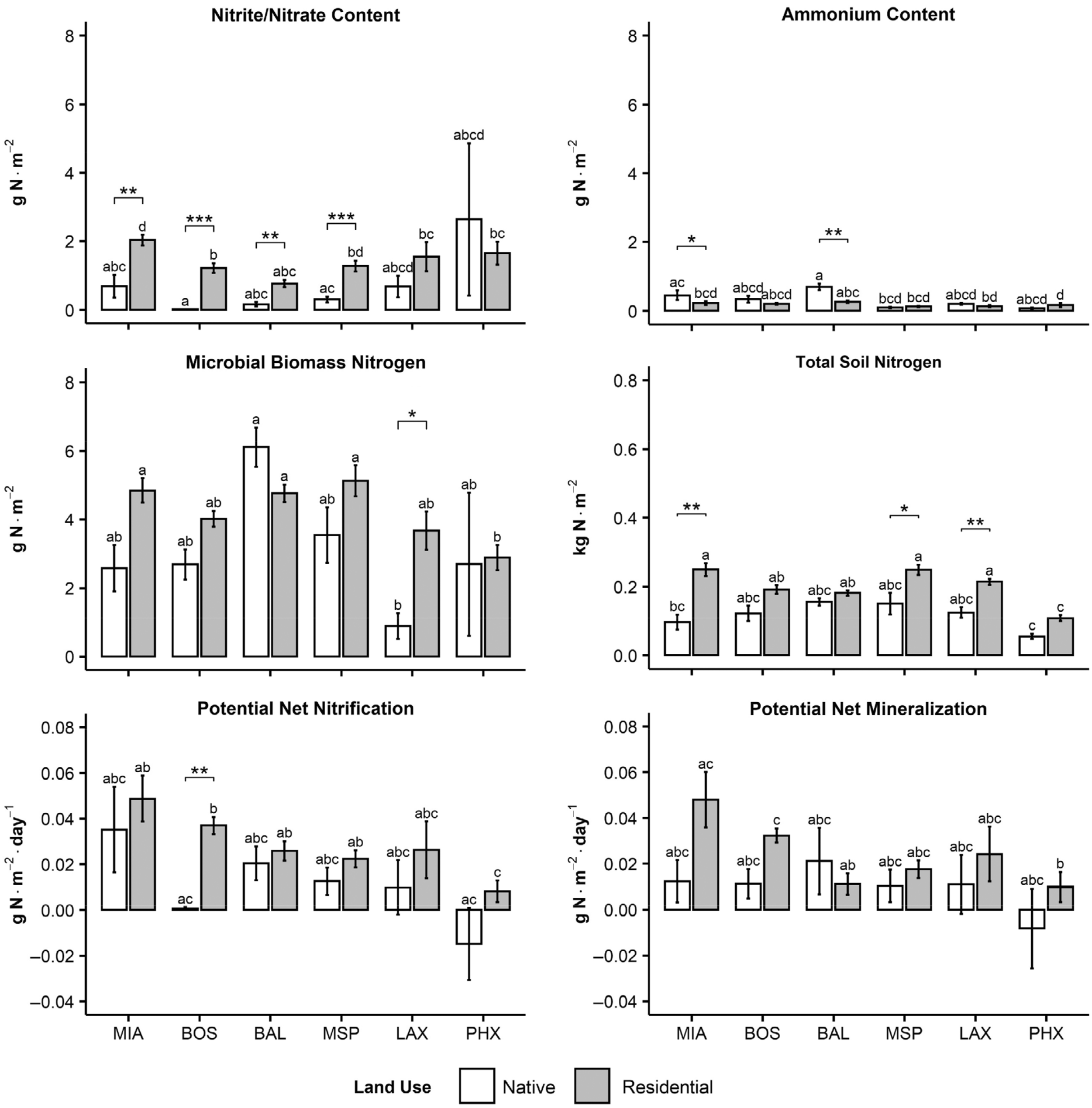

FIGURE 4 Soil N cycle variables in topsoil (0–10 cm depth) in residential and native reference soils in six cities across the continental United States. Values are means (±1 SE) of all residential and native sampling sites in each city. Cities are ordered from wettest to driest along the *x*-axis. Asterisks indicate significant differences between native and residential soils. ****$p \leq 0.0001$, ***$p \leq 0.001$, **$p \leq 0.01$, *$p \leq 0.05$. Bars with different letters are significantly different at $p < 0.05$. MIA, Miami, FL; BOS, Boston, MA; BAL, Baltimore, MD; MSP, Minneapolis–St. Paul, MN; LAX, Los Angeles, CA; PHX, Phoenix, AZ.

native = 0.15 kg N m^{-2} ± 0.03, $p \leq 0.05$), and LAX (residential = 0.21 kg N m^{-2} ± 0.01, native = 0.13 kg N m^{-2} ± 0.02, $p \leq 0.01$; Figure 4).

Carbon pools and fluxes

Neither microbial biomass C nor respiration in topsoil averaged across all six cities differed between native and residential topsoil ($p > 0.05$; Figure 1). However, total soil C (data from Trammell et al., 2020) was higher in residential than in native topsoil (residential = 2.59 kg C m^{-2} ± 0.09, native = 1.84 kg C m^{-2} ± 0.18, $p \leq 0.001$; Figure 1). On a city-by-city basis, there were no significant differences in microbial biomass C between residential and native topsoil, but there were differences in total soil C. In nearly all cities, total C was higher in residential topsoil. This difference was significant in MIA (residential = 3.90 kg C m^{-2} ± 0.35, native = 2.18 kg C m^{-2} ± 0.38, $p \leq 0.05$), LAX (residential = 2.39 kg C m^{-2}

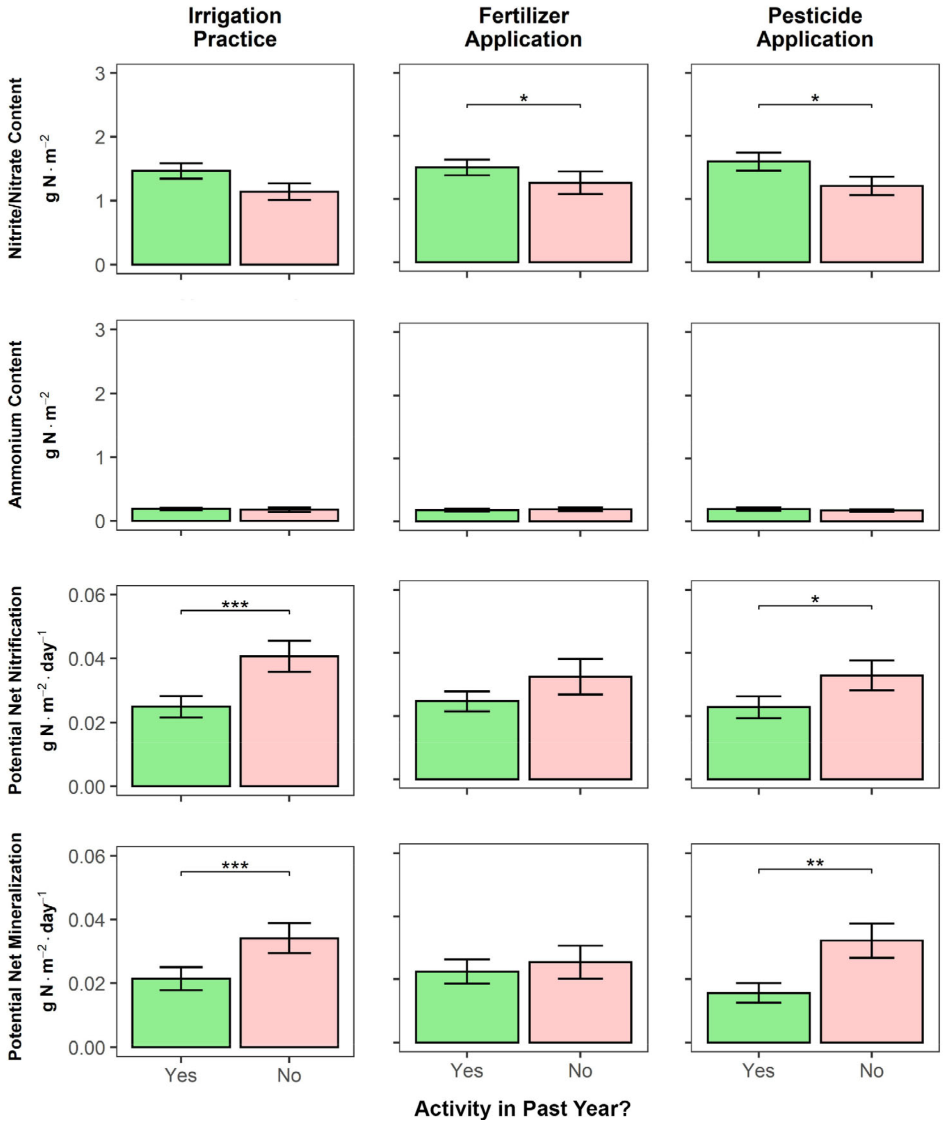

FIGURE 5 NO_2^-/NO_3^- and NH_4^+ contents, and potential net nitrification and mineralization rates in topsoil (0–10 cm) as affected by the self-reported use of fertilizer, irrigation, and pesticides in residential soils. Values are means (±1 SE) of all residential sampling sites in six cities across the continental United States. Asterisks indicate significant differences between bars. ****$p \leq 0.0001$, ***$p \leq 0.001$, **$p \leq 0.01$, *$p \leq 0.05$.

± 0.10, native = 1.25 kg C m^{-2} ± 0.16, $p \leq 0.01$), and PHX (residential = 1.46 kg C m^{-2} ± 0.08, native = 0.53 kg C m^{-2} ± 0.14, $p \leq 0.01$; Figure 2). Respiration was significantly higher in residential topsoil in LAX (residential = 1.02 g C m^{-2} ± 0.13, native = 0.27 g C m^{-2} ± 0.02, $p \leq 0.05$; Figure 2) and significantly higher in native topsoil in MSP (residential = 0.88 g C m^{-2} ± 0.14, native = 1.15 g C m^{-2} ± 0.16, $p \leq 0.05$; Figure 2).

Self-reported fertilizer, irrigation, or pesticide application had no effect on either microbial biomass C or respiration ($p > 0.05$, data not shown). Total soil C was significantly lower in yards with self-reported fertilizer (fertilizer application = 2.48 kg C m^{-2} ± 0.11, no fertilizer application = 2.79 kg C m^{-2} ± 0.14, $p \leq 0.05$; Figure 3) and pesticide application (pesticide application = 2.40 kg C m^{-2} ± 0.11, no pesticide application = 2.82 kg C m^{-2} ± 0.14, $p \leq 0.05$; Figure 3).

Homogenization in residential soils

Nearly all measured biogeochemical metrics had a higher coefficient of variation for native topsoil than for residential topsoil (Figure 6). The only exception was NH_4^+ content, where variation was higher for residential than for native topsoil. The difference in variation was greatest for NO_2^-/NO_3^- content and potential net nitrification.

Patterns with depth

Values of both microbial biomass C and N, and respiration decreased markedly with depth across nearly all cities (Appendix S2: Figure S1). In contrast, potential net nitrification and mineralization seemed to remain relatively consistent, even at the 60+ cm depth.

For pH, residential soil in all cities exhibited reduced acidity, expressed even at the deepest parts of the soil cores (Appendix S2: Figure S2). Soil compaction during sampling produced elevated bulk density values at depth at many sites, so depth patterns are only evaluated on a per gram of soil basis (Appendix S2: Figures S1–S3).

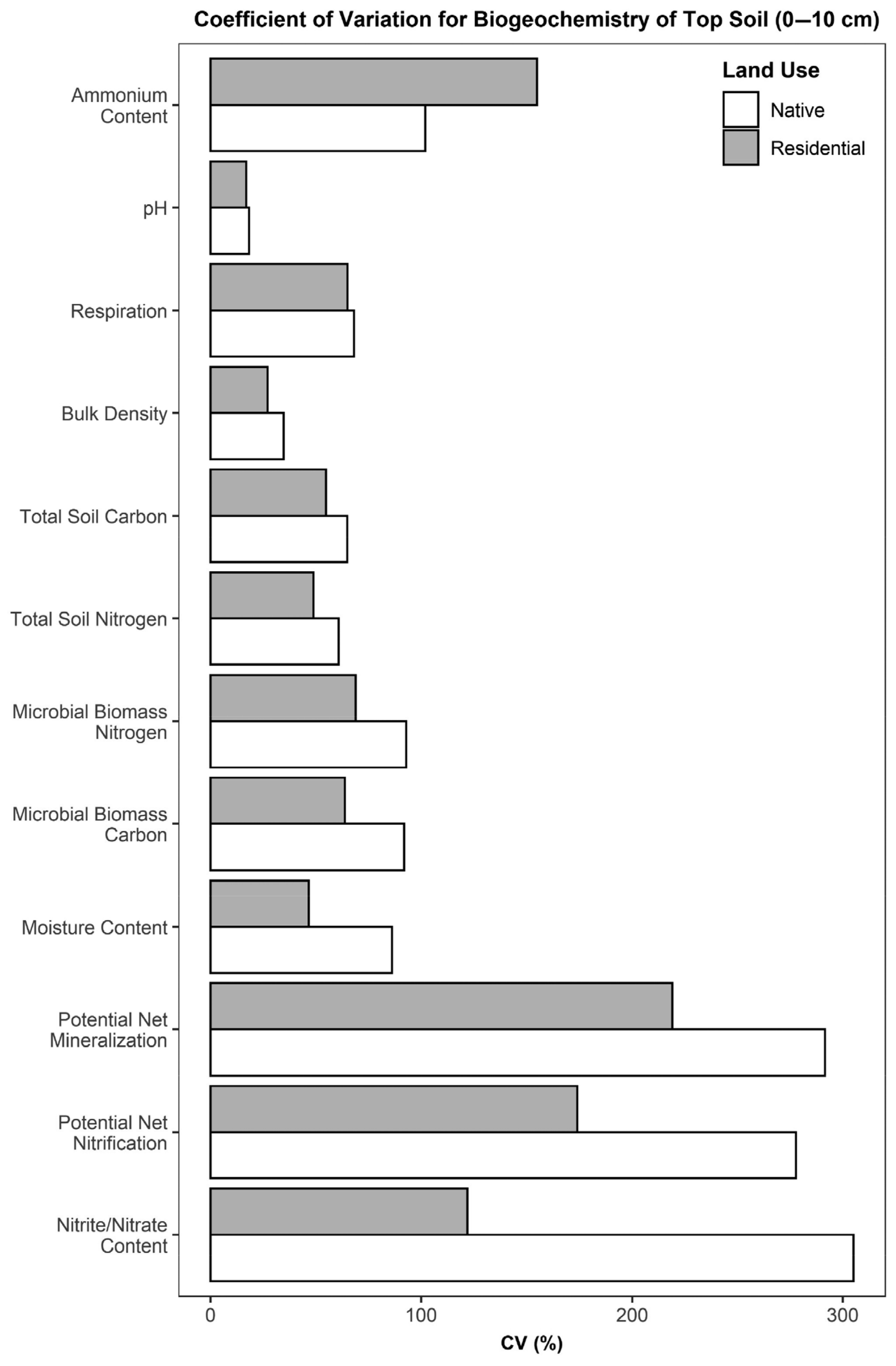

FIGURE 6 Coefficient of variation (CV) for soil C and N cycle processes in topsoil (0–10 cm depth). Values are the coefficient of variation of the means of all residential ($n = 279$) and native ($n = 44$) samples within six cities across the continental United States.

DISCUSSION

Soil nitrogen cycling

Higher total N, NO_2^-/NO_3^- content, and microbial biomass N in residential topsoil supported our hypothesis that soils in residential ecosystems will have elevated N pools and cycling rates compared with soils from native ecosystems. While NH_4^+ content was lower in residential topsoil, this was likely caused by the increased rate of nitrification in these soils, a process that converts NH_4^+ to NO_2^-/NO_3^- (Figure 1). There has long been concern about hydrologic and gaseous N losses from residential lawns caused by elevated N supply (Robbins, 2007). Lawns can be fertilized and irrigated, which can stimulate N leaching from these systems (Morton et al., 1988). This leaching is of great concern because NO_3^- is a drinking water pollutant and an agent of eutrophication around the world (Byrnes et al., 2020). Consistent with our results, Trammell et al. (2016) reported higher plant N content in fertilized than in unfertilized lawns from these same sites.

While we did observe clear evidence that land conversion to residential lawns accelerates N cycling in cities across the United States, we did not observe changes in potential net N mineralization and microbial biomass N content, and we did not observe any effects of self-reported fertilizer input on N cycling. These results were consistent with previous studies in BAL (Groffman et al., 2009; Raciti et al., 2008; Raciti, Groffman, Jenkins, Pouyat, & Fahey, 2011) and elsewhere (Bachman et al., 2016; Bock & Easton, 2020; Hall et al., 2009; Herrmann & Cadenasso, 2017; Wang et al., 2014) that find limited differences in soil N cycling processes between fertilized and unfertilized lawns. This outcome may be because our measure of fertilizer use here (self-reported in a telephone survey) was not quantitative or time-specific. In addition, our sampling was not linked to fertilizer application, so we were not able to evaluate short-term effects of fertilizer on soil N cycling measures when they would be expected to be most likely.

While most studies have focused on the importance of fertilization as a driver of N cycling in lawns (Carey et al., 2012; Easton & Petrovic, 2004; Petrovic, 1990), our results show that irrigation and pesticide practices can also be important. Irrigation practices were associated with lower potential net mineralization, potential net nitrification, and microbial biomass N, suggesting a decrease in microbial N cycling activity. This decrease could be driven by stimulation of N immobilization by increasing microbial activity as irrigation can relieve water limitation of microbial growth and activity, and pesticides can serve as a substrate for this growth and activity (Robertson & Groffman, 2014). Immobilization could also be stimulated indirectly via stimulation of plant growth and supply of labile C to the soil (Hart et al., 1994). There could also be direct toxic effects of pesticide application on microbial biomass and activity, but many controlled studies have shown few of these direct effects (Moorman, 1989; Reisinger et al., 2016).

It is reasonable to assume inherent differences in net N mineralization (a measure of soils' ability to supply inorganic N) that occur among native ecosystems may potentially influence N cycling after conversion to residential land use (Groffman et al., 2016). Net mineralization varies widely among ecosystems, with the lowest values found in shrub ecosystems, the highest values found in wetlands, and overall decreases with increasing latitude (Liu et al., 2017). Variation in potential net N mineralization has been found to be predictive of N loss following disturbance of forest ecosystems (Vitousek et al., 1979). However, we found that continental-scale variation in N mineralization did not appear to influence patterns of net N mineralization in residential soils, which varied much less in this variable than in native soils. Native ecosystems in MIA, LAX, and PHX were scrubland; in BAL and BOS, the native ecosystems were forest; and in MSP, there were both forest and grassland native ecosystems (Table 1). The driest city in our study, PHX had the lowest rates of total soil N, potential net mineralization, and potential net nitrification. It also exhibited the highest average NO_2^-/NO_3^- content, though the variation was very high (Figure 4). Thus, the influence of lawn management appeared to overwhelm any legacy effect of the native vegetation.

Soil carbon cycling

The significant increase in total soil C in residential soils reported by Trammell et al. (2020) led to our hypothesis that microbial biomass C and respiration would be higher in residential than in native soils. However, these effects were only observed in the two driest cities, LAX and PHX (Figure 2). This was likely driven by greater net primary productivity caused by inputs of water and fertilizer in these arid environments, and it is not surprising that conversion of desert ecosystems to heavily subsidized residential lawns results in increases in both above- and belowground C cycle processes (Bijoor et al., 2008; Hall et al., 2009; Townsend-Small & Czimczik, 2010). The increases in respiration suggest that soil C turnover, as well as pool size, increased in irrigated residential soils in these cities.

Ultimately, the C impact of residential landscapes will depend on interactions between climate, C turnover, C stabilization, and land use. Addition of water and nutrients clearly increases soil C levels in residential soils in arid regions (Trammell et al., 2020). The conversion of agricultural lands, which tend to have low soil C levels, to lawns is also likely to result in increases in soil C, in many regions, but again, uncertainties about turnover remain (Raciti, Groffman, Jenkins, Pouyat, Fahey, Cadenasso, & Pickett, 2011). Any full accounting of the C balance of residential lawns should account for CO_2 emissions from fossil fuel burning landscape management equipment, as well as emissions associated with irrigation water transport and fertilizer production (Townsend-Small & Czimczik, 2010).

Soil pH

Consistent with previous studies, pH was higher in residential than in native soils. In a comparison of five global cities, Pouyat et al. (2015) found that pH was generally higher in turf or ruderal soils than in native or remnant soils. When comparing the soil pH of agricultural grasslands with native woodland and grassland sites in Massachusetts, Neill et al. (2007) found that pH was increased in agricultural soils. Asabere et al. (2018) found that soil pH was substantially increased in urban soils compared with that in forest soils and rural arable soils for the upper 10 cm of depth in Kumasi, Ghana. Park et al. (2010) found that pH was increased in soils sampled near roads compared with that in more interior samples and more so that elevated pH was correlated with soil calcium, suggesting a connection between the limestone ($CaCO_3$) utilized in cement production in urban areas.

Although pH was increased in residential soils, variation was not affected, suggesting that pH is not a component of ecological homogenization among our cities. Pouyat et al. (2015) note that pH is influenced by both geologic and anthropogenic factors, which may complicate the analysis of pH homogenization. The overarching increase in soil pH in residential soils has significant ecological implications. Nitrification, which produces NO_2^-/NO_3^-, has been shown to be highly sensitive to increases in pH, and thus, the pH increase may at least partially underlie the stimulation of soil N cycling in residential soils that we observed. Yan et al. (2016) found that bacterial diversity and community composition were positively correlated with soil pH in urban soils in China. Neill et al. (2007) found that non-native species density was positively correlated with pH in agricultural grasslands in coastal Massachusetts, USA.

The importance of management

Evaluating the effects of lawn management practices on biogeochemical variables has been challenging because although controlled field and laboratory studies have shown how fertilizer use on lawns creates clear risks for water and air pollution (Morton et al., 1988), studies of lawns managed by homeowners show variations in fertilizer applications (Fraser et al., 2013), mixed evidence for environmental effects (Bachman et al., 2016; Bock & Easton, 2020; Carey et al., 2012; Groffman et al., 2009; Petrovic, 1990), and that the behaviors that drive this soil variability are complex and multi-scale (Campbell et al., 2014; Harris et al., 2012, 2013; Hayden et al., 2015; Larson et al., 2020; Locke et al., 2018; Martini et al., 2013, 2014; Polsky et al., 2014; Sisser et al., 2016).

Although fertilizer application was associated with higher NO_2^-/NO_3^- content, it was also associated with a decrease in total N content. Fertilizer application did not significantly influence potential net mineralization, potential net nitrification, and microbial biomass. Several factors may influence soil N cycling indices and make influences of fertilizer applications hard to detect, including varying time since fertilizer application, a legacy of past fertilizer inputs, or locally high atmospheric N deposition rates (Bettez & Groffman, 2013; Decina et al., 2018; Hobbie et al., 2017; Lovett et al., 2000).

An additional surprising result was that self-reported use of irrigation water in the last year was associated with lower levels of potential net N mineralization and net nitrification. Irrigation may have stimulated plant growth and uptake of N, reducing N availability and cycling by the soil microbial community. It is important to note that these results indicate nothing about the potential for hydrologic losses associated with overwatering, especially in the periods just after fertilizer application. It is also possible that irrigation is more common in drier cities, such as PHX, where rates of potential net mineralization and nitrification in native soils are the lowest among all sampled sites. Similarly, self-reported irrigation practice was associated with higher bulk density, and bulk density values for native topsoil in PHX are the highest of all sampled sites (Figure 4).

Homogenization in residential soils

We interpret the lower coefficient of variation for all soil metrics except NH_4^+ pools in residential compared with those in native topsoil to be strong evidence for homogenization of soil processes. These findings add to the growing evidence of ecological homogenization of the urban United States or the creation of an American residential macrosystem (Groffman et al., 2016, 2017; Pearse et al., 2016; Trammell et al., 2016, 2020; Wheeler et al., 2017). That said, it is also important to note that variations among residential topsoil were quite high, so while soil homogenization may be occurring relative to native soils, there still is a large degree of variation in residential soils. More generally, Pouyat et al. (2015) note that while soil C and N are likely to converge/exhibit homogenization because they are associated with biogenic factors that are managed, properties more strongly controlled by soil parent material such as potassium and phosphorus are less likely to converge. A broader generalization about homogenization in residential ecosystems will require further investigation of what soil properties are or are not strongly influenced by management compared with other factors.

CONCLUSION

Residential development across the United States strongly influences soil C and N biogeochemistry. Driven by larger cultural trends in American residential development, the transformation of soil N and C cycle processes, moisture content, and pH is likely to have effects on multiple ecological variables, over multi-decadal timescales. For example, to the extent that plant biodiversity and animal biodiversity are adapted to and arise from local soil conditions, ecological change and homogenization of soil conditions may be a long-term, spatially extensive constraint on biodiversity that could be very slow or difficult to remediate. The homogenization of soil conditions that we observed here could cause an ecological cascade of effects on plant and animal components of these ecosystems (Frelich et al., 2019).

There is a clear need for a better understanding of soil C and N biogeochemical processes in residential ecosystems, and for developing management strategies to use this understanding to improve their environmental performance. For example, while residential lawns have a surprisingly high capacity to store C, particularly in drier climates, these benefits may be somewhat offset by concomitant increases in respiration. There is a clear need for detailed studies on C dynamics in residential soils so that lawns can be managed to improve C sequestration and reduce N hydrologic and gaseous losses to receiving ecosystems. Our results show clear stimulation of N cycle processes, and there is a need for further research to determine whether this stimulation results in increases in NO_3^- leaching to groundwater and emissions of nitrous oxide to the atmosphere.

Cultural landscaping practices clearly have important impacts on soil biogeochemical properties. That said, results are often different from what one might expect. Not all lawns are fertilized, and fertilizer effects on N and C cycling processes were not as strong as expected. Both irrigation and pesticide application had surprising effects on N transformations. There is a clear need for further coupled social–ecological research to understand the drivers and effects of these practices across the American residential macrosystem.

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CONFLICT OF INTEREST

The authors declare no conflict of interest.

DATA AVAILABILITY STATEMENT

Data are available from the EDI Data Portal: https://doi.org/10.6073/pasta/5683662180499904732e654e3869f3e6.

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SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

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