Oecologia (2018) 187:1107–1118
https://doi.org/10.1007/s00442-018-4194-3

SPECIAL TOPIC

Soil carbon and nitrogen accumulation in residential lawns of the Salt Lake Valley, Utah

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Received: 22 August 2017 / Accepted: 5 June 2018 / Published online: 28 June 2018

Abstract
Urban lawn ecosystems are widespread across the United States, with fertilization rates commonly exceeding plant nitrogen (N) uptake rates. While urban soils have been shown to accumulate C and N over time, the long-term balance of N inputs and losses from lawn soils remains largely uncertain. We sampled residential lawn soils aged 7–100 years in the Salt Lake City metropolitan area as a means of inferring changes in total nitrogen (TN) content, organic carbon (OC) content, C:N ratio, and $\delta^{15}N$ of bulk soil over time. Core-integrated (0–40 cm) TN and OC stocks increased linearly by 2.39 g N m^{-2} $year^{-1}$ and 29.8 g OC m^{-2} $year^{-1}$ over the 100-year chronosequence. TN and OC percent were also negatively correlated with elevation. Multiple linear regression models including housing age and elevation as covariates, explained 68 and 62% of variability in TN and OC stocks respectively. $\delta^{15}N$ increased with housing age, soil depth, and clay content, suggesting N removal over time, especially in poorly drained soils. We quantified potential hydrologic and gaseous N losses over time by comparing observed N accumulation to different historic fertilization scenarios. Modeling and isotopic results suggest that, while soil N has accumulated over time, the majority of N added to lawns in the Salt Lake Valley over 50 years of fertilization was likely lost from surface soils via denitrification or leaching.

Keywords Stable isotopes · ^{15}N · Chronosequence · Urban · Turfgrass

Introduction

Soil organic matter (SOM) is an important sink for carbon (C) and nitrogen (N) globally. This is especially important in human-dominated landscapes, where excess N from fertilizer and atmospheric deposition contributes to greenhouse gas emissions, eutrophication, and degraded air quality (Gruber and Galloway 2008). In urban watersheds, excess anthropogenic N sources broadly consist of atmospheric deposition, fertilizer, and food import/sewage (Bernhardt et al. 2008). Lawn ecosystems are ubiquitous in urban areas, and have thus expanded with growing suburban and exurban land cover (Milesi et al. 2005; Robbins and Birkenholtz 2003). The majority of lawns in the United States is fertilized and irrigated (Polsky et al. 2014), and N fertilization rates often exceed plant N demand (Law et al. 2004; Fissore et al. 2012). At the watershed scale, N loading from lawn fertilizer is often equal to or greater than the N flux from wastewater in suburban regions (Baker et al. 2001; Groffman et al. 2004; Bernhardt et al. 2008; Hobbie et al. 2016). When fertilization rates exceed plant needs, excess N may accumulate in the soil, or be removed via denitrification (microbial removal) and leaching (hydrologic removal). Understanding the fate of N added to the urban landscape, as well as factors driving N storage is crucial for the management of water quality, air quality, and emissions of greenhouse gases (NO, N_2O).

Several studies have documented accumulation of C and N over time in urban lawns (Zhu et al. 2006; Lewis et al. 2006; Shi et al. 2006; Kaye et al. 2008; Raciti et al. 2011a) as well as higher C and N content of lawn soils compared to soils beneath adjacent native vegetation (Kaye et al. 2004; Zhu et al. 2006; Lewis et al. 2014). Soil

Communicated by Todd E. Dawson.

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N accumulates primarily as organic N, in association with organic C that builds up over time as homeowners irrigate and fertilize lawns (Raciti et al. 2008). Several studies have shown that soils under fertilized lawns consistently accumulate SOM rapidly for at least 30 years (Qian et al. 2003; Lewis et al. 2006; Raciti et al. 2011a), with mixed evidence of slower accumulation rates between 50 and 100 years after establishment (Qian et al. 2003; Shi et al. 2006). Assuming inputs remain the same, C and N accumulation rates in lawn ecosystems may slow as losses increase. This pattern has been documented in abandoned agricultural fields (Knops and Tilman, 2000) as well as old growth forests (Hedin et al. 1995), over longer timescales. Over time, turfgrass ecosystems may become net sources of N to groundwater and the atmosphere (Qian et al. 2003).

At the watershed scale, the accumulating soil organic N pool serves as an important sink for excess fertilizer. The future of this N pool is a source of uncertainty in regional N budgets (Baker et al. 2001; Raciti et al. 2011a; Hobbie et al. 2016). For accumulation to occur, N inputs must be greater than outputs; however, even small changes in the accumulation rate may be accompanied by increased N_2O emissions and/or groundwater N contamination. Several studies have shown that N outputs via leaching and gaseous fluxes are often low relative to N accumulation (e.g., Raciti et al. 2011b; Groffman et al. 2009; Baker et al. 2001; Wang et al. 2014). In Baltimore, MD, for instance, lawns with moderate fertilization (98 kg N ha^{-1} $year^{-1}$) accumulate 79% of annual N inputs, while leaching and gaseous losses account for just 13% of total fertilizer N inputs (Raciti et al. 2011b). N loss rates are subject to change over time as soils become N saturated but continue to receive elevated N inputs (Qian et al. 2003; Zhang et al. 2013). A recent study by Hobbie et al. (2016) estimated that ecosystem N accumulation accounted for between 8 and 12% of residential fertilizer inputs across several suburban watersheds in Minnesota. In addition, urban lawns often have elevated N_2O emissions compared with native landscapes (Kaye et al. 2004; Hall et al. 2008); however, this flux consistently accounts for less than 5% of fertilizer inputs, even under long-term high-fertilization scenarios (Hall et al. 2008; Zhang et al. 2013). N losses from leaching are more variable and subject to increase as soil N pools become saturated (Qian et al. 2003). The majority of reported N leaching rates account for less than 10% of fertilizer inputs (e.g., Raciti et al. 2011b; Wang et al. 2014; Hall et al. 2016) although recent studies from small suburban catchments in the midwestern United States estimate N leaching rates equivalent to between 20 and 40% of inputs despite relatively low fertilization rates of 15–36 kg N ha^{-1} $year^{-1}$ (Fissore et al. 2012; Hobbie et al. 2016; Nidzgorski and Hobbie 2016). Using the CENTURY model, Qian et al. (2003) suggest leaching rates can increase precipitously to 40–50% of fertilizer inputs if lawns become saturated with N over time.

Along with time since establishment, historical land use, soil properties, and socioeconomic factors can influence SOM density and accumulation in lawns. Prior agricultural land use has been shown to influence the standing stock and accumulation rate of SOM in both mesic (Raciti et al. 2011a) and arid cities (Lewis et al. 2006, 2014) regardless of soil age. Underlying natural factors such as elevation, climate, and soil texture are also important drivers of SOM across different ecosystems (Jobbágy and Jackson 2000) as well as across cities (Pouyat et al. 2002). Lawn management is a potentially important driver of heterogeneity in SOM and N cycling in urban landscapes as well. Fertilization, irrigation, and mowing frequency can significantly alter SOM accumulation rates over short timescales (Wang et al. 2014; Qian et al. 2003; Barton and Colmer 2006). Wang et al. (2014), for instance, found that gaseous N losses increased from 7.9 to 20.2% of total N inputs when fertilizer increased from 48 to 192 kg N ha^{-1} $year^{-1}$ along with increased irrigation. Organic C accumulation also increases with increased management intensity (i.e., fertilizer, mowing, irrigation, pesticides; Zirkle et al. 2011).

Where long-term monitoring data are not available, stable isotopes of N can serve as useful tracers of integrated N sources and cycling in ecosystems. Bulk soil $\delta^{15}N$ tends to become enriched over time and also with depth, as organic matter is mineralized and recycled and some portion of depleted $\delta^{15}N$ is lost from the ecosystem (Nadelhoffer and Fry 1988; Amundson et al. 2003). The physical process of leaching NO_3^- out of the soil column does not influence soil $\delta^{15}N$; however, each step of microbial conversions of organic matter to $NH_4^+$and NO_3^-, N_2 or N_2O favors lighter ^{14}N over ^{15}N (Mariotti et al. 1981), leading to $\delta^{15}N$-enrichment in bulk soil $\delta^{15}N$ soils over time (Nadelhoffer and Fry 1988). In cities, anthropogenic N inputs can influence soil $\delta^{15}N$. These include highly enriched wastewater (+ 5 to + 20‰), localized atmospheric NO_3^- deposition (− 8.5 to + 1.7‰, Hastings et al. 2013) and fertilizer (− 5 to + 5 $\delta^{15}N$ ‰, Kendall et al. 2007). Soil and plant matter N content varies from + 0.5 to + 12‰ $\delta^{15}N$ (Amundson et al. 2003), and may become $\delta^{15}N$ depleted if synthetic fertilizer dominates the TN pool (Trammell et al. 2016).

We sought to evaluate the magnitude and drivers of OC and TN accumulation, and potential for N losses, in the Salt Lake City metropolitan region over 100 years. Along with time since establishment, we also evaluated historical land cover, soil texture, elevation, and socioeconomic drivers of SOM accumulation. Assuming that residential soils have been fertilized and irrigated at similar rates over time, we hypothesized that if soils in this region have become N saturated, we would find a non-linear relationship between N content and household age. We also hypothesized that $\delta^{15}N$

would become more enriched over time if significant losses from the soil have occurred, whereas $\delta^{15}N$ would stay the same or decline if losses were small and inputs dominated by synthetic fertilizer. Assuming that income is a proxy for intensity of fertilization and irrigation inputs, we hypothesized that lawns in high-income areas would have larger pools of N and organic C as well as lower $\delta^{15}N$, compared with low-income homes of a similar age. We tested these hypotheses in Salt Lake City by sampling lawns across a 100-year chronosequence of housing age.

Methods

Site selection

Salt Lake City, UT, lies at the center of a major urban corridor between the Wasatch and Oquirrh Mountain Ranges. The mountains restrict urban development and the dispersion of atmospheric pollutants on both the east and west sides of the valley. The valley bottom is characterized by a semiarid climate with mean annual precipitation of 397 mm and mean average temperature of 11 degrees C (WorldClimate 2008). Despite unfavorable natural conditions, turfgrass and trees are abundant within the urbanized portions of the study area, where they are maintained and/or established through irrigation. Early writings describe the area as primarily grassland prior to European settlement in the 1840s, with trees generally confined to riparian corridors and upland shrubs being much less dominant than in modern times (Wakefield 1933).

Soil samples were collected from residential lawns between May and November 2007. Study sites consisted of 38 single-family residential parcels across Salt Lake County, built between 1900 and 2000. Sites were selected using the Salt Lake County parcel dataset (Salt Lake County Assessor 2008), which included information on the year that the current home was built. The year built for each home was used as a proxy for the date of transition to residential land use, and housing age was calculated by subtracting the year built from the year sampled (2007). Initial site selection was randomly stratified by decade of development after excluding parcels larger than 0.25 ha, and homeowner permission was granted prior to sampling each parcel.

Spatial information

We also determined the median income for the census block group surrounding each of our study parcels. We acquired block-group shapefiles for Salt Lake County and attached income data from the 2014 American Community Survey 5-year estimates. We overlaid this dataset on our map of sampled parcels and used a spatial join in ArcMap 10.3 GIS software to assign a median block income value to each parcel (U.S. Census Bureau 2016).

Land use prior to 1937 was determined based on aerial imagery of the Salt Lake Region (Utah State Geological Survey 2017). Images were georeferenced to a basemap of current aerial imagery using the georeferencing tool in ArcGIS 10.3. Prior land use categories (developed, wildland, or crops) were specified by visually inspecting imagery beneath each parcel in ArcMap.

Sampling methods

Two lawn soil cores were collected at each parcel using a 2-cm-diameter soil recovery probe to a depth of 40 cm (or less if stones larger than 2 cm were encountered) and sectioned into 10-cm increments. We collected a total of 75 cores across the 38 homes, and were able to recover a full core (0–40 cm) from 27 out of 75 cores. We used this subset of 27 for analyses related to whole-core C and N accumulation and interactive effects of sample depth and other drivers of soil C and N. Each 10-cm core segment was individually processed and analyzed. During sieving, increments were cleaned of visible roots using tweezers. Fine soil (< 2 mm), coarse soil (> 2 mm), and fine root fractions (< 2 mm diameter) were weighed. Two 5 g subsamples were isolated from air-dried and sieved soil through quartering (Tan 1996). One subsample was ground on a ball mill (MM200, Retch, Haan, Germany) for chemical analysis, and the other was dried in an 80 °C forced-air oven for 48 h to derive a moisture correction factor for soil texture and bulk density calculations.

Percent total carbon (C), percent TN, and $\delta^{15}N$ (‰) were measured on 30 (± 1) mg of air-dried and ground soil using an elemental analyzer coupled to an isotope ratio mass spectrometer (Finnigan MAT delta S, Thermo Electron Co, San Jose, CA) at the University of Utah Stable Isotope Facility (SIRFER). Soil reference material included in the analyses exhibited average standard deviations of 0.02, 0.00, and 0.4 for % total C, %TN, and $\delta^{15}N$ (‰), respectively ($n = 40$). Inorganic C was measured on 300 (± 10) mg of air-dried and ground soil via pressure calcimeter (Wagner et al. 1998, Sherrod et al. 2002), and OC was determined by difference. TN and OC concentrations were converted to mass per unit ground surface area for all full 40-cm cores by integrating samples from each 10-cm increment, using the following equation:

$$\mathrm{TN}\ (\mathrm{g\ m^{-2}}) = \%\mathrm{TN}/100 \times F \times (10\ \mathrm{cm}/D)/A \times 10{,}000, \tag{1}$$

where F is the fine soil fraction weight (g), D is the actual increment depth recovered (cm), and A is the cross-sectional area of the sampling probe tip (cm^2). The same protocol was used for OC, replacing TN in Eq. (1). Because this calculation does not include stones larger than 2 cm, it likely overestimates soil TN and OC contents in most cases.

Soil texture, pH and bulk density

Bulk density of each core increment was calculated as the oven-dry weight of rock and root-free soil divided by soil volume estimated from the sampling probe. We measured soil texture for each site on a composite sample of lawn soils (0–20 cm) using a rapid method developed by Kettler et al. (2001). Briefly, 10 g soil samples were each combined with 100 mL 3% sodium hexametaphosphate (HMP) solution and mechanically shaken for 2–3 h. Following particle dispersion, the sand fraction was isolated by rinsing the soil solution through a 0.053-mm sieve, and the silt fraction was isolated through sedimentation. Percent sand and silt were calculated using the oven-dried weight of the respective fraction and the oven-dried weight of the original sample. Subtracting percent sand and silt from 100 produced a value for percent clay. Organic matter was not removed prior to dispersion, which may slightly overestimate sand and underestimate clay.

Statistical analyses

We employed several statistical tests to address our hypotheses. First, to address the hypothesis that C and N stocks would increase logarithmically over time if soil C and N pools became saturated over time, and linearly if they had not yet saturated, we compared the fit of linear and logarithmic model of housing age versus TN and OC stock relationship. We determined the best model based on the highest r^2 and lowest AIC for both TN and OC stocks.

To test hypotheses about temporal, socioeconomic, physiological, and historic drivers of TN and OC accumulation both for whole-core C and N stocks as well as C and N concentrations at depth, we used a comprehensive model selection technique to determine the optimal combination of explanatory variables for each response variable. Response variables included TN g m^{-2} to 0–40 cm, OC kg m^{-2} to 40 cm, and %OC, %TN and $\delta^{15}N$ ‰, C:N ratio at each 10-cm depth interval. Covariates at the household-scale included housing age, % clay content, land use prior to 1937 (wild land, agricultural, or developed), elevation, median census block-group income, and interaction terms between depth interval and housing age, elevation and income. Interaction terms were included to test whether these household-scale drivers are influential at depth.

All statistical analyses were carried out using R programming language (R Core Team 2017). The model selection procedure involved first testing all possible combinations of covariates, using the 'bestglm' package in R, which determines the combination of covariates with the lowest AIC (McLeod and Xu 2017). AIC is a measure of both the model fit and complexity, and is used in model selection to reduce over-fitting.

After selecting the optimal covariates, we applied a linear mixed effects model approach to account for pseudoreplication because our sampling design was nested (i.e., two core samples at each household and four depth increments per core). Using the package 'nlme' in R (Pinheiro et al. 2017), we included a random effect for cores nested within a household (i.e., core/household), regardless of whether it improved model fit. Covariates selected using the 'bestglm' procedure were included as fixed effects, and we used the stepAIC() function from the package 'MASS' in R (Venables and Ripley 2002) to perform a backward stepwise regression test determining whether or not this selection of covariates still resulted in the lowest AIC after including random effects. We scaled each variable and covariate to have a mean of 0, and report scaled model parameter estimates in Table 1.

We tested the final model for each response variable for linear model assumptions. We plotted the standardized residuals versus fitted values to inspect each model for homoscedasticity and linearity. We tested for multivariate normality by plotting a histogram and a Q–Q plot of residuals for each response variable. We tested for multicolinearity among covariates in the final model for each response variable using the variance inflation factor (VIF) in R package 'car' (Fox and Weisberg 2011), and removed covariates with VIF > 10. Ideally, VIF should be below 3 to eliminate the presence of multicolinearity; however, 10 is also a suitable cutoff when models include interaction terms (Zuur et al. 2010). Out of the six response variables modeled, only the model for $\delta^{15}N$ included a covariate with VIF > 10 (elevation x depth increment interaction). We removed this covariate for the final model of $\delta^{15}N$. Additionally, VIF was < 3 for all covariates in the remaining models, except depth increment x housing age.

Certain covariates in our dataset were intrinsically correlated due to the structure and demographics of Salt Lake City. To assess all possible correlations, independent of the mixed effects model selection, we also calculated Pearson's correlation coefficient for all covariates (Fig. 4).

Modeling historic soil N loading

We found that a linear model of soil TN stock versus parcel age fit slightly better than logarithmic with a difference in AIC of 2.2 and 0.07 difference in r^2. This supports our hypothesis that N has accumulated in soils during the 100-year history captured by our chronosequence. We inferred an accumulation rate based on the slope of the following relationship:

$$\mathrm{TN}_y = mY_y + b, \tag{2}$$

Table 1 Standardized linear regression coefficient (coef) and partial *r*-squared values for individual parameters in multiple linear regressions for whole-core (0–40 cm) TN and OC, as well as %OC, %TN, δ^{15}N (‰), and C:N ratio by depth increment

Covariate tested	Value	TN g m^{-2} 0–40 cm	OC g m^{-2} 0–40 cm	TN %	OC %	d15 N	C:N ratio
Housing age	coef	0.48	0.48	0.68	0.75	0.59	− 0.03
	r^2	0.38	0.36	0.23	0.30	0.15	0.05
Depth increment	coef	na	na	− 0.37	ns	0.50	− 0.63
	r^2	na	na	0.08	ns	0.11	0.04
Income	coef	ns	ns	ns	ns	ns	0.00
	r^2	ns	na	ns	ns	na	0.10
Elevation	coef	− 0.51	− 0.44	− 0.20	ns	na	0.02
	r^2	0.41	0.32	0.10	na	na	0.12
Percent clay	coef	ns	ns	ns	ns	0.38	0.09
	r^2	na	na	na	na	0.27	0.06
Prior land use: agricultural	coef	ns	ns	ns	ns	ns	ns
	r^2	na	na	na	na	na	na
Prior land use: developed	coef	ns	ns	ns	0.55	ns	2.73
	r^2	na	na	na	0.07	na	0.15
Prior land use: wildland	coef	ns	ns	ns	− 0.31	ns	0.60
	r^2	na	na	na	0.03	na	0.01
Age × depth interaction	coef	ns	ns	− 0.48	− 0.88	− 0.38	0.01
	r^2	na	na	0.07	0.52	0.03	0.03
Income × depth interaction	coef	ns	ns	ns	ns	ns	ns
	r^2	na	na	na	na	na	na
Elevation × depth interaction	coef	ns	ns	ns	ns	ns	ns
	r^2	na	na	na	na	na	na
Full model	r^2	0.68	0.62	0.66	0.61	0.51	0.27
	AIC	54.77	57.24	311.43	318.04	340.12	647.78
	dAIC	− 5.66	− 4.66	− 8.80	10.39	0.68	223.78

NA denotes parameters that were not included in the model and NS denotes covariates removed during stepwise linear regression model selection. The full-model AIC and *r* squared values describe the overall model fit for each covariate, and dAIC describes the difference in AIC between a linear model with all available covariates and the final model

where TN_y is the standing stock of TN (g N m^{-2}) in year *y*, *m* is the slope of the relationship between housing age and TN, Y_y is the age of a hypothetical lawn in a given year, and *b* is the standing stock of TN in year 0.

To evaluate whether the current standing stock of lawns would have required fertilizer inputs, and to bracket potential losses of fertilizer N to the atmosphere and/or groundwater, we developed a simple model of N accumulation and compared it with different potential historic N input scenarios. We estimate the historic TN stock of a 100-year-old lawn using Eq. (2), and evaluated the difference between measured and cumulative N input for five scenarios. For each, we assumed that atmospheric deposition has continued at a constant rate of 1.2 g N m^{-2} year^{-1} based on contemporary regional N deposition measurements (Fenn et al. 2003) for the 100-year time series (1907–2007). While this is a first-order approximation that makes simplifying assumptions, the selection of N deposition rates does not strongly influence the results because even high rates of N deposition are very small compared with recommended lawn fertilization rates. We also assumed that atmospheric deposition was the only N input to residential lawns prior to widespread implementation of the Haber–Bosch process. Fertilization levels were chosen based on the low (5 g N m^{-2} year^{-1}) and high (30 g N m^{-2} year^{-1}) range of recommendations from the Utah State extension service (Sagers et al. 1990), and fertilization scenarios involved synthetic fertilizer application beginning in 1950 and continuing constantly until the present day. In scenarios with clippings removed, we assumed that this removal constituted 25% of fertilizer inputs, based on the mean N removal rate from clippings in Wang et al. (2014). These assumptions facilitate a first-order approximation of potential N losses over time. We modeled the cumulative N load on an annual time step for each of the five scenarios following the below equation:

$$TN_y = Y_p \times L_p + Y_{HB} \times L_{HB}, \tag{3}$$

where TN_y is the predicted TN stock in year (*y*), Y_p is the number of years before 1950 (before Haber–Bosch), L_p is the annual net loading rate (g N m^{-2} year^{-1}) before 1950, Y_{HB}

is the number of years after 1950, and L_{HB} is the post-1950 loading rate for that scenario.

Results

Correlations with OC, TN and δ^{15}N, C:N

Bulk soil %N ranged from 0.02 to 0.74% across all sites and 10-cm depth increments. Soil %OC ranged from 0.14 to 7.53% across the same subset of samples. Whole-core TN density (sampled to 40 cm) ranged from 83.2 to 499.7 g N m^{-2} and OC density ranged from 0.84 to 6.02 kg C m^{-2}.

A multiple linear regression explained 68% of the variability in whole-core TN density (g N m^{-2}, Table 1). The two covariates remaining after stepwise model selection included housing age (partial $r^2 = 0.38$) and elevation (partial $r^2 = 0.41$). Housing age was positively correlated with TN density and elevation was negatively correlated (Table 1). The multiple linear model of soil OC density (g C m^{-2}) had an overall r^2 of 0.62, with 36% variability attributed to housing age, and 32% attributed to elevation.

Soil %N and %OC both decreased significantly with core depth and increased with housing age (Table 1; Fig. 1). Multiple linear models of %TN and %OC by 10 cm depth interval resulted in a significant interaction between soil age and depth for both dependent variables (Table 1). Differences related to age were greatest between shallow (0–10 cm) core sections, and soils of different ages became more similar with depth (Fig. 1). %OC and %TN were also both negatively correlated with elevation. Soil δ^{15}N increased with both depth and age. As with %N and %OC, the greatest differences in δ^{15}N related to age were found in the upper 10 cm of soil. We were only able to explain 13% of variability in C:N ratio with the available covariates. C:N was positively, but weakly correlated with elevation (partial $r^2 = 0.05$), % clay (partial $r^2 = 0.06$), and negatively with depth increment (partial $r^2 = 0.037$). Interaction between housing age and depth interval explained an additional 3% and housing age along explained less than 1%.

Correlations among covariates

When checking for correlations among covariates, we found a strong positive correlation between median income and elevation (Fig. 4) and a moderate negative correlation between elevation and % clay. These relationships did not compromise the fit or assumptions of our final linear models because we accounted for potential variance inflation based on the VIF. Correlations among covariates do influence our interpretation of model parameter selection, however, as positively correlated variables (elevation and income) may have overlapping influence on response variables.

Temporal trends in OC and TN accumulation and potential losses

Housing age was significantly ($p < 0.001$) positively correlated with whole-core OC and TN stocks (g m^{-2}, Table 1; Fig. 2). The slopes of these relationships correspond with potential average accumulation rates of 2.39 g N m^{-2} year^{-1}

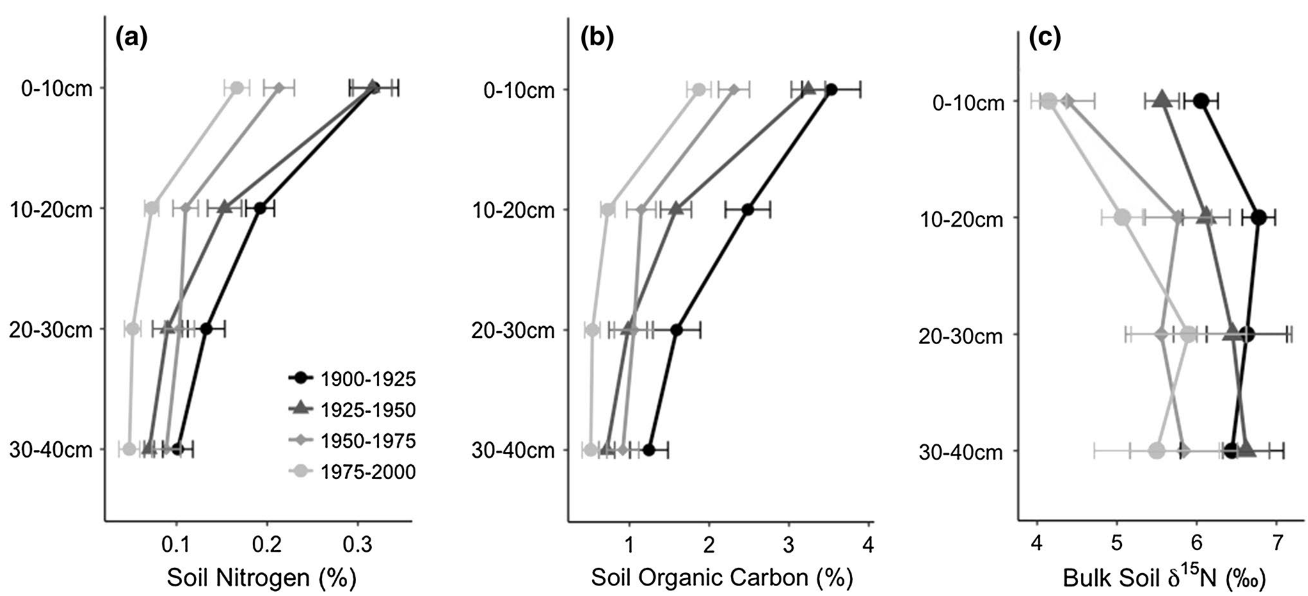

Fig. 1 Depth profiles of **a** %N, **b** %OC, and **c** δ^{15}N (‰) in residential lawn soils. Shades of gray signify the age of housing units where soils were sampled, grouped by quarter century. Error bars indicate one standard deviation from the mean by quarter and depth increment

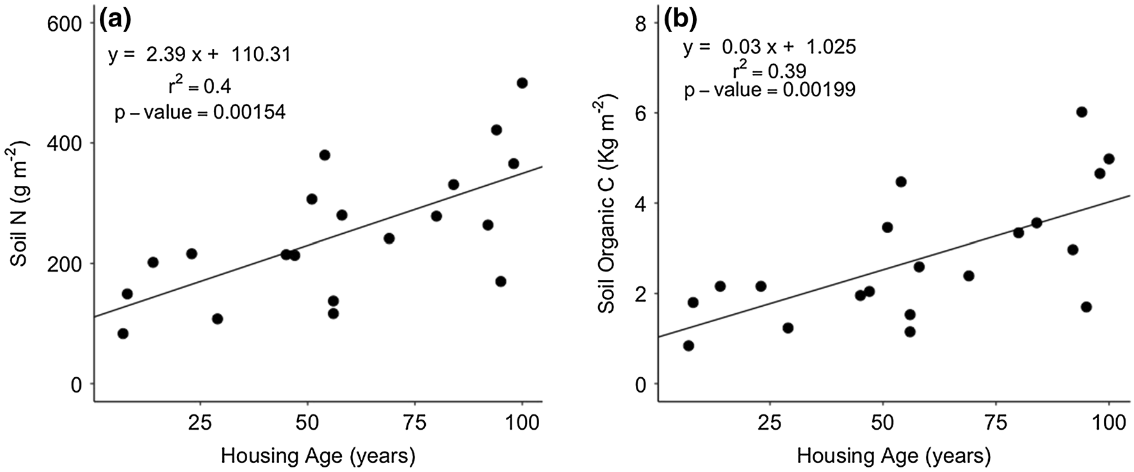

Fig. 2 Scatterplot of **a** bulk soil TN **b** bulk soil OC vs. housing age. Each point indicates average bulk TN or OC of two 0–40-cm cores at each site. Significant ($p < 0.001$), positive relationships exist between housing age and both OC and TN contents, with r^2 values of 0.38 and 0.40, respectively

and 29 g C m^{-2} year^{-1} (Fig. 2). Our observed TN accumulation rate is substantially lower than the two recommended fertilizer application rates of 5 and 29 g N m^{-2} year^{-1}. The cumulative N inputs under the two different fertilization scenarios exceeded observed TN in lawns by 153 and 1544 g N m^{-2} after 57 years of simulated fertilization application (years 1950–2000 of Fig. 3). N inputs in the high-fertilization scenario surpassed observed soil TN stocks after 1 year, while in the low-fertilization scenario N inputs surpassed stocks in 14 years (Fig. 3). When 25% of

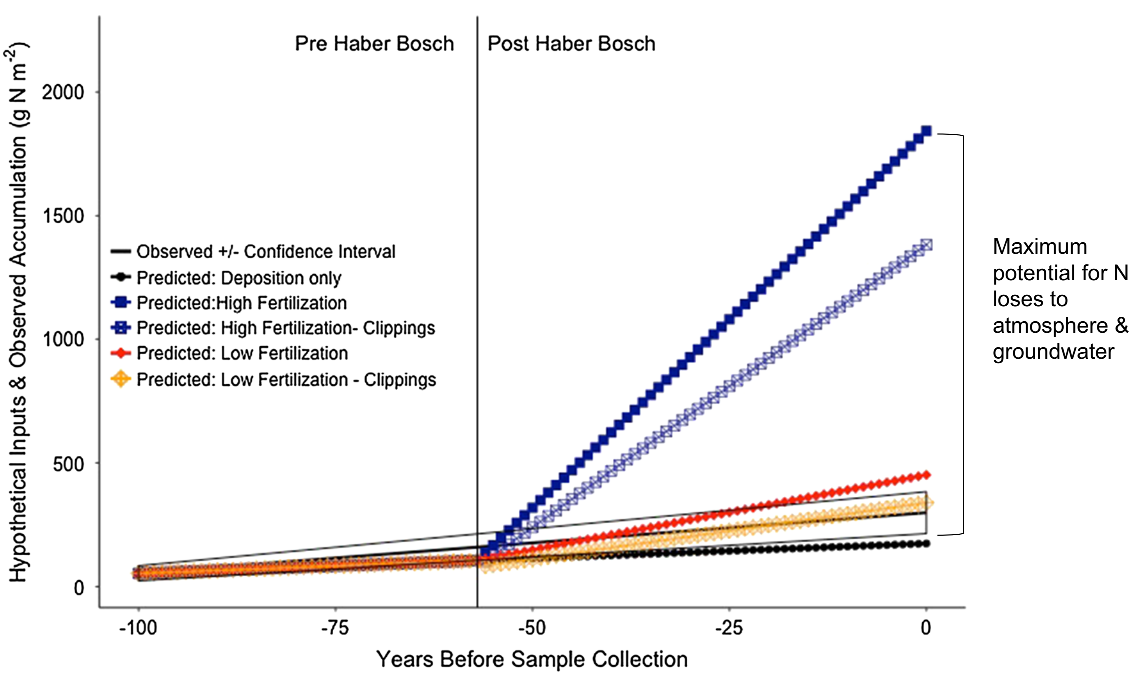

Fig. 3 Comparing the expected change in TN stocks over time with cumulative TN loading for a given housing age. Different symbols denote TN accumulated in the soil under the following scenarios: Solid lines: the observed pattern from the local chronosequence ± 95% confidence interval, filled circles: atmospheric N deposition inputs only (1.2 g N m^{-2} year^{-1}), red filled diamonds: low fertilization rate dose (5 g N m^{-2} year^{-1}), red open diamonds: low fertilization with clippings removed, blue filled squares: high dose of N fertilizer (29 g N m^{-2} year^{-1}), blue open squares: high fertilization with clippings removed. Grass clippings were assumed to constitute a constant 25% of fertilizer addition for both high and low scenarios. This figure is available in color in the online version of this journal

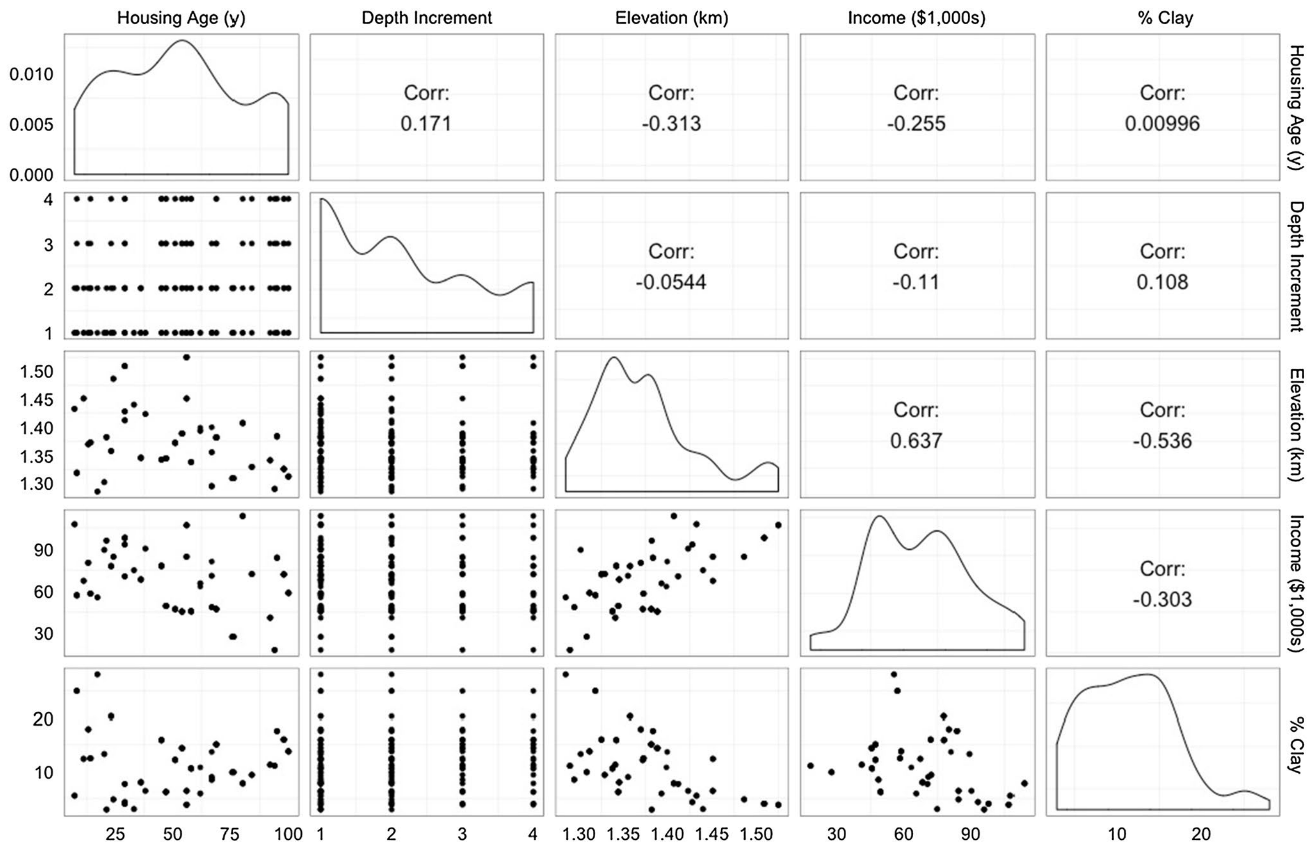

Fig. 4 Correlation matrix among all predictor variables used in multiple linear regression models. Values represent Pearson's correlation coefficient

fertilizer was removed as grass clippings, the higher fertilization N inputs took 4 years to surpass observed TN stocks and the lower scenario took 38 years to surpass observed TN stocks. The offset between cumulative loading and observed N stocks can be interpreted as potential high and low estimates of combined N losses to groundwater and/or the atmosphere over this time period.

Discussion

Lawn fertilizer is a significant N input to urban watersheds; however, the fate of this N source has been shown to vary widely across studies (Groffman et al. 2004; Hale et al. 2014; Bernhardt et al. 2008). Because lawns cannot accumulate organic matter indefinitely, and hydrologic and gaseous N losses from lawn soils represent a threat to water quality and greenhouse gas emissions, respectively, it is important to understand the potential for changing SOM dynamics over time. Assuming N inputs remain relatively constant (or increase) over time, decreases in N accumulation can be attributed to increased N losses. Maintaining this assumption, we hypothesized that the relationship between housing age and C and N stocks would fit a logarithmic function if SOM had become saturated within the 100-year timeframe, and a linear function if SOM had not yet reached saturation in lawns. Surprisingly, we found that N stocks have accumulated linearly over 100 years (Fig. 1; Table 1). Trends of increasing δ^{15}N with soil age and depth (Table 1; Fig. 2) demonstrate that significant N losses have likely occurred under virtually all reasonable scenarios of historical N inputs. Contrary to numerous prior studies in urban lawns, which show relatively low N loss compared with accumulation rates (Raciti et al. 2011a; Wang et al. 2014; Hobbie et al. 2016), our results suggest that a portion of lawn N inputs may be lost to the atmosphere or groundwater prior to lawn N stocks becoming fully saturated. The magnitude of total N losses—both gaseous and hydrologic—can be estimated by comparing current standing stocks of N to cumulative N inputs under different fertilization scenarios (Fig. 3).

C and N accumulation and density

Our soil C and N accumulation rates are on the low end of the range of prior studies in humid regions, and high compared with more arid soils. TN and OC have accumulated in lawns of Salt Lake City region at a rate of 2.39 g N m^{-2} $year^{-1}$ and 29 g C m^{-2} $year^{-1}$, respectively. These rates are low compared with N and OC accumulation in Baltimore 8.3 g N and 82 g C m^{-2} $year^{-1}$ (Raciti et al. 2011a), but high compared with N accumulation in Phoenix (1.2–1.9 g N m^{-2} $year^{-1}$; Lewis et al. 2006; Kaye et al. 2008). Wang et al. (2014) measured a range of N immobilization rates (2.8–8.7 g N m^{-2} $year^{-1}$) in the Los Angeles area, with the lowest immobilization rates occurring under low fertilizer and irrigation regimes.

While our soil C and N accumulation rates are on the low end of the range of prior studies, the apparent lack of saturation of the C and N pools was still somewhat surprising, given that fertilizer N inputs have likely exceeded plant nutrient demands historically (Fig. 3). Mechanistic models predict rapid organic matter accumulation in the early years following lawn establishment and slower accumulation later (Qian et al. 2003). Shi et al. (2006) found increasing N content between 3-, 8-, 25- and 97-year-old lawn ecosystems, with the greatest apparent accumulation rate in younger lawns. Qian et al. (2003) also predicted that the N accumulation rate would slow over time, from 6.6 to 1.4 g N m^{-2} $year^{-1}$ between years 1–30 and years 60–100, respectively. The difference between expected and observed patterns of N accumulation may be explained partially by heterogeneity in lawn management, historic land use, and physical characteristics of soil. Additionally, it is conceivable that older homes were built on soils with higher initial SOM, if initial settlements were focused on areas with better farming. This might hide a logarithmic trend that would become apparent if individual properties were tracked through time.

As with N accumulation rates, N density fell within the high and low range of prior studies in arid cities, and was low compared with humid region lawn soils. The mean N density in our study (251 g N m^{-2}) is comparable to the highest N densities measured in Phoenix (Lewis et al. 2006). Given the slow rate of N accumulation, it is possible that soils in this region have the potential to retain more N if given more time. The maximum N density of surface soils (0–40 cm) measured in our study, of 499 g N m^{-2}, was similar to the maximum predicted by Qian et al. (2003) for soils in Colorado measured to 60 cm. However, our linear regression model (Fig. 1) predicts N density of ~ 300 g N m^{-2} after 100 years. Some of the discrepancy across studies is clearly due to the depth of the soil profile sampled (i.e., 1-m depth in Baltimore; Raciti et al. 2011a; 40 cm in Salt Lake City, and 60 cm in Colorado; Qian et al. 2003). Differences in climate are apparent when comparing surface soils (0–10 cm). Raciti et al. (2011a) measured high average surface N stocks (170 N m^{-2}) in Baltimore, compared with arid and semiarid cities of Phoenix (90–110 g N m^{-2}; Lewis et al. 2006; Kaye et al. 2008), and Salt Lake City (93 g N m^{-2}; this study). Surface soils in Denver (0–15 cm) stand out among these studies, however, with 465 g N m^{-2} (Kaye et al. 2004). Prior land use is also an important consideration. Lawns with significant N accumulation in Baltimore were established on formerly agricultural soils, with deep profiles and high capacity for organic N sorption. We did not find an effect of prior land use (agricultural versus wildland) on N content in Salt Lake City, however, despite widespread alkaline soils, which can contribute to high N sorption.

Surface soil OC densities in our study ranged from 0.4 to 6.0 kg C m^{-2} across lawns between 7 and 100 years post-development (0–40 cm). These values are low compared with the range of soil C stocks measured in residential soils across the United States (Pouyat et al. 2006, 8.3–10.8 kg C m^{-2}). However, that study only included one city in the western U.S. (Oakland, CA), which also had the lowest C density to 1 m depth. Our results are more comparable to measurements in Phoenix, AZ, which ranged from 0.8 to 2.0 kg C m^{-2}, with higher OC stocks in previously agrarian lawns. While our areal densities are biased on the low side due to sampling depth, OC density of surface soils (0–10 cm) in our study is high compared with surface OC density in desert soils and xeric lawns measured by Kaye et al. (2008) of 0.45–0.5 kg C m^{-2}.

Prior studies of urban lawn ecosystems have reported significantly higher OC densities than adjacent native ecosystems (Kaye et al. 2004; Qian and Follett 2002; Pouyat et al. 2009). Pouyat et al. (2009) found that shallow soils under urban turfgrass in Baltimore, MD, and Denver, CO, were both 2.5- and 2-fold higher than adjacent native soil OC stocks, despite an absence of climatological differences. Intensive management of urban soils may overcome natural constraints on OC accumulation such as climate, parent material, and time, leading to convergence in soil properties across cities as lawns age (Lewis et al. 2006, 2014; Pouyat et al. 2009).

Physical and socioeconomic correlations with soil N and C content

While housing age was the most effective predictor of both TN and OC contents in residential lawns, we hypothesized that additional factors related to physical properties, land cover history, and socioeconomic factors would also affect organic matter accumulation in soils. Given prior evidence of correlations between lawn management and socioeconomic status (Groffman et al. 2016), as well as evidence that high N inputs are linked to higher N storage (Wang et al. 2014), we expected to find higher TN stocks in high-income lawns, assuming that wealthy households use more fertilizer. We did not find this to be the case, as income was not a significant predictor of TN stocks or TN percent at depth (Table 1). TN was negatively correlated with elevation, however, and we found a strong positive correlation between income and elevation in Salt Lake City (Fig. 4). This correlation between elevation and income (i.e., wealthier neighborhoods at higher elevations) complicates interpretation of elevation-alone driving changes. TN was unrelated to other natural or historic factors (soil texture, prior land use, Table 1).

The negative direction of the TN versus elevation relationship remains surprising. There could be several reasons for this trend. First, since there was a 300-m variation in elevation across our study sites, consistently lower temperatures and or lower historical primary productivity related to soil properties at higher elevations could drive lower soil organic matter accumulation overall. This hypothesis is supported by the negative relationship between OC stocks and elevation (Table 1). Another possibility is that N losses are greater at higher elevation due to differences in soil drainage. We found a moderate negative correlation between elevation and % clay (Fig. 4), which could be related to greater hydrologic N losses from high-elevation soils. There could be differences in N inputs to the soil along elevation gradients, due to differences in fertilizer or atmospheric deposition. We expected to find higher N stocks in high-elevation lawns, however, given correlations between elevation and income, and prior research on management intensity and income. If fertilization is instead equal across lawns, differences in N deposition between low-elevation areas could explain higher N inputs to low-elevation lawns. Hall et al. (2016) measured relatively constant dissolved N deposition rates across the Salt Lake Valley; however, studies in other regions have demonstrated highly variable N deposition within cities (Decina et al. 2017), and it is difficult to rule out significant spatial variability in N pollution in this region without targeted sampling of high- versus low-income areas (Cobley et al. in revision).

N isotopes provide further insight into spatial and temporal variability in N content. $\delta^{15}N$ was strongly predicted by housing age, soil depth, percent clay, and interactions between age and depth. We expected to find a negative relationship between $\delta^{15}N$ and income, if synthetic fertilizer dominated inputs and minimal losses of N have occurred over time. While there is a moderate, negative correlation between these variables (Fig. 4), income was not a significant predictor of $\delta^{15}N$ when housing age and soil texture were taken into account.

Carbon accumulation followed a similar pattern to N, with the same predictors (housing age, elevation) being identified as important; however, different drivers of OC emerged at depth. Along with soil depth and housing age, variability in OC was predicted best by historic land use (Table 1). Unlike TN, soil OC was significantly lower in soils previously under native vegetation (i.e., wildland) compared with soils previously in agriculture.

N losses over time: evidence and potential mechanisms

While fertilized soils accumulate N over time, immobilization alone cannot explain the fate of historic fertilization in our study. We compared soil N accumulation with five scenarios of historic N inputs for a hypothetical 100-year-old lawn to bracket the potential magnitude of hydrologic and gaseous N losses in this region. Results from this scenario-based model demonstrate that (1) current N stocks are not feasible without some fertilizer input, and (2) there is potential for significant N losses throughout the 50-year period if any but the lowest recommended application rates of fertilizer had been applied (Fig. 3). Patterns in $\delta^{15}N$ enrichment with age, soil depth, and interaction between these two parameters provide additional evidence that significant N losses have taken place over time (Fig. 1). The pattern of increased $\delta^{15}N$ with soil age and depth is also widespread in natural systems, as $\delta^{15}N$-depleted losses (via NO_3^- leaching or denitrification) enrich the remaining soil pool (Amundson et al. 2003). One key difference with natural systems is that N inputs in natural systems are generally very low compared with fertilized lawns, and more sensitive to isotope effects through recycling of soil organic matter. In highly fertilized systems, N inputs exceed plant needs, reducing the necessity for internal recycling of N.

The relative importance of immobilization compared with losses via leaching or denitrification has implications for downstream impacts of fertilizer. Patterns in $\delta^{15}N$ provide insights into the potential loss mechanism of N from soils. While quantifying the relative contribution of leaching vs. gaseous losses is beyond the scope of this study, $\delta^{15}N$ patterns suggest denitrification may be the dominant source. The most compelling evidence for this is the positive correlation between $\delta^{15}N$ and clay content in soils (Table 1). Since clay content is related to soil drainage, N losses are likely dominated by denitrification in the high-clay soils of our study (Groffman and Tiedje, 1989; Silver et al. 2000). N leaching may have a greater influence on N losses in well-drained soils of our study area. $\delta^{15}N$ enrichment of bulk soils may result from either hydrologic or gaseous losses; however, gaseous losses tend to have a larger effect. The process of leaching does not alone influence isotopic enrichment; nitrification produces water-soluble NO_3^-, which is $\delta^{15}N$ depleted compared with the SOM pool. The isotope effect due to gaseous losses is much greater than that of leaching because denitrification removes N from the already $\delta^{15}N$-depleted NO_3^- pool (Mariotti et al. 1981).

Conclusions

Soil organic matter accumulation is an important N sink in urban watersheds (Baker et al. 2001; Raciti et al. 2011a; Hobbie et al. 2016). Unlike numerous studies showing that N and OC accumulation stops or slows after 30–50 years post-development (Qian et al. 2003; Shi et al. 2006), we document relatively low but consistent N and OC accumulation in residential lawn soils over a 100-year period in the Salt Lake City metropolitan region. N accumulation at this rate would not be possible without some level of fertilization inputs beyond current levels of atmospheric deposition (Fig. 3); however, fertilization at the highest recommended levels for this region would have likely resulted in significant N losses given the current standing stocks of bulk soil N. Isotopic evidence suggests that (1) significant losses have taken place over time, (2) socioeconomic factors affecting lawn care influence N and C accumulation rates, and (3) denitrification is likely an important mechanism for N losses, especially in clay-rich soils in this region. Recent studies support this conclusion, having demonstrated enhanced denitrification capacity in irrigated lawn soils (Hall et al. 2016).

Acknowledgements The authors would like to thank the 40 homeowners who allowed us access to sample their lawns, as well as Bert Granberg and La'Shaye E. Cobley for assistance with spatial analysis of land cover and census data. Funding was provided by EF-0120142, EF-1241286, and EPS 1208732 from the U.S. National Science Foundation.

Author contribution statement RMS performed statistical analyses and wrote the manuscript; JCW, JE, and PD conceived the study, carried out field and laboratory measurements, and contributed to writing the methods and discussion sections; and DEP provided insights into analysis and interpretation of the data.

References

Amundson R, Austin AT, Schuur EAG et al (2003) Global patterns of the isotopic composition of soil and plant nitrogen. Global Biogeochem Cycles 17:31\1–31\10. https://doi.org/10.1029/2002gb001903

Baker LA, Hope D, Xu Y et al (2001) Nitrogen balance for the Central Arizona-Phoenix (CAP) ecosystem. Ecosystems 4:582–602. https://doi.org/10.1007/s10021-001-0031-2

Barton L, Colmer TD (2006) Irrigation and fertiliser strategies for minimising nitrogen leaching from turfgrass. Agric Water Manag 80:160–175. https://doi.org/10.1016/j.agwat.2005.07.011

Bernhardt ES, Band LE, Walsh CJ, Berke PE (2008) Understanding, managing, and minimizing urban impacts on surface water nitrogen loading. Ann N Y Acad Sci 1134:61–96. https://doi.org/10.1196/annals.1439.014

Decina SM, Templer PH, Hutyra LR et al (2017) Variability, drivers, and effects of atmospheric nitrogen inputs across an urban area: emerging patterns among human activities, the atmosphere, and soils. Sci Total Environ 609:1524–1534. https://doi.org/10.1016/j.scitotenv.2017.07.166

Fenn ME, Haeuber R, Tonnesen GS et al (2003) Nitrogen emissions, deposition, and monitoring in the Western United States. Bioscience 53:391–403. https://doi.org/10.1641/0006-3568(2003)053[0391:NEDAMI]2.0.CO;2

Fissore C, Hobbie SE, King JY et al (2012) The residential landscape: fluxes of elements and the role of household decisions. Urban Ecosystems 15:1–18. https://doi.org/10.1007/s11252-011-0189-0

Fox J, Weisberg S (2011) An {R} companion to applied regression, 2nd edn. Sage, Thousand Oaks. http://socserv.socsci.mcmaster.ca/jfox/Books/Companion

Groffman PM, Tiedje JM (1989) Denitrification in north temperate forest soils: spatial and temporal patterns at the landscape and seasonal scales. Soil Biol Biochem 21:613–620

Groffman PM, Law NL, Belt KT et al (2004) Nitrogen fluxes and retention in urban watershed ecosystems. Ecosystems 7:393–403. https://doi.org/10.1007/s10021-003-0039-x

Groffman PM, Williams CO, Pouyat RV et al (2009) Nitrate leaching and nitrous oxide flux in urban forests and grasslands. J Environ Qual 38:1848–1860. https://doi.org/10.2134/jeq2008.0521

Groffman PM, Grove JM, Polsky C et al (2016) Satisfaction, water and fertilizer use in the American residential macrosystem. Environ Res Lett 11:34004. https://doi.org/10.1088/1748-9326/11/3/034004

Gruber N, Galloway JN (2008) An earth-system perspective of the global nitrogen cycle. Nature 451:293–296. https://doi.org/10.1038/nature06592

Hale RL, Turnbull L, Earl S et al (2014) Sources and transport of nitrogen in arid urban watersheds. Environ Sci Technol 48:6211–6219. https://doi.org/10.1021/es501039t

Hall SJ, Huber D, Grimm NB (2008) Soil N_2O and NO emissions from an arid, urban ecosystem. J Geophys Res Biogeosci 113:1–11. https://doi.org/10.1029/2007jg000523

Hall SJ, Baker MA, Jones SB et al (2016) Contrasting soil nitrogen dynamics across a montane meadow and urban lawn in a semi-arid watershed. Urban Ecosyst 19:1083–1101. https://doi.org/10.1007/s11252-016-0538-0

Hastings MG, Casciotti KL, Elliott EM (2013) Stable isotopes as tracers of anthropogenic nitrogen sources, deposition, and impacts. Elements 9:339–344. https://doi.org/10.2113/gselements.9.5.339

Hedin LO, Armesto JJ, Johnson AH (1995) Patterns of nutrient loss from unpolluted, old-growth temperate forests: evaluation of biogeochemical theory. Ecology 76:493–509. https://doi.org/10.2307/1941208

Hobbie SE, Finlay JC, Janke BD et al (2016) Contrasting nitrogen and phosphorus budgets in urban watersheds and implications for managing urban water pollution. Proc Natl Acad Sci 114:4177–4182. https://doi.org/10.1073/pnas.1618536114

Jobbágy EG, Jackson RB (2000) The vertical distribution of soil organic carbon and its relation to climate and vegetation. Ecol Appl 10:423–436. https://doi.org/10.1890/1051-0761(2000)010[0423:TVDOSO]2.0.CO;2

Kaye JP, McCulley RL, Burke IC (2004) Carbon fluxes, nitrogen cycling, and soil microbial communities in adjacent urban, native and agricultural ecosystems. Glob Change Biol 11:575–587. https://doi.org/10.1111/j.1365-2486.2005.00921.x

Kaye AJP, Majumdar A, Gries C et al (2008) Hierarchical bayesian scaling of soil properties across urban, agricultural, and desert ecosystems. Ecol Appl 18:132–145

Kendall C, Elliott EM, Wankel SD (2007) Tracing anthropogenic inputs of nitrogen to ecosystems. In: Michener R, Lajtha K (eds) Stable isotopes in ecology and environmental science, 2nd edn. Blackwell Publishing, pp 375–449

Kettler TA, Doran JW, Gilbert TL (2001) Simplified method for soil particle size determination to accompany soil-quality analyses. Soil Sci Soc Am J 65:849–852

Knops JMH, Tilman D (2000) Dynamics of soil nitrogen and carbon accumulation for 61 years after agricultural abandonment. Ecology 81:88–98. https://doi.org/10.1890/0012-9658(2000)081[0088:DOSNAC]2.0.CO;2

Law NL, Band LE, Grove M (2004) Nitrogen input from residential lawn care practices in suburban watersheds in Baltimore County, MD. J Environ Plan Manage 47:737–755. https://doi.org/10.1080/0964056042000274452

Sagers LA "Fertilizing Lawns" (1990) All Archived Publications. Paper 716. http://digitalcommons.usu.edu/extension_histall/716

Lewis DB, Kaye JP, Gries C et al (2006) Agrarian legacy in soil nutrient pools of urbanizing arid lands. Glob Change Biol 12:703–709. https://doi.org/10.1111/j.1365-2486.2006.01126.x

Lewis DB, Kaye JP, Kinzig AP (2014) Legacies of agriculture and urbanization in labile and stable organic carbon and nitrogen in Sonoran Desert soils. Ecosphere. https://doi.org/10.1890/ES13-00400.1

Mariotti A, Germon J, Hubert P et al (1981) Experimental determination of nitrogen kinetic isotope fractionation: some principles; illustration for the denitrification and nitrification process. Plant Soil 62:413–430. https://doi.org/10.1007/BF02374138

McLeod AI, Xu C (2017) bestglm: Best Subset GLM. R package version 0.36. https://CRAN.R-project.org/package=bestglm

Milesi C, Running SW, Elvidge CD et al (2005) Mapping and modeling the biogeochemical cycling of turf grasses in the United States. Environ Manage 36:426–438. https://doi.org/10.1007/s00267-004-0316-2

Nadelhoffer KJ, Fry B (1988) Controls on natural nitrogen-15 and carbon-13 abundances in forest soil organic matter. Soil Sci Soc Am J 52:1633–1640

Nidzgorski DA, Hobbie SE (2016) Urban trees reduce nutrient leaching to groundwater. Ecol Appl 26:1566–1580. https://doi.org/10.1002/15-0976

Pinheiro J, Bates D, DebRoy S, Sarkar D, R Core Team (2017) nlme: linear and nonlinear mixed effects models. R package version 3.1-131. https://CRAN.R-project.org/package=nlme

Polsky C, Grove JM, Knudson C et al (2014) Assessing the homogenization of urban land management with an application to US residential lawn care. Proc Natl Acad Sci USA 111:4432–4437. https://doi.org/10.1073/pnas.1323995111

Pouyat RV, Groffman PM, Yesilonis I, Hernandez L (2002) Soil carbon pools and fluxes in urban ecosystems. Environ Pollut 116:S107–S118. https://doi.org/10.1016/S0269-7491(01)00263-9

Pouyat RV, Yesilonis ID, Nowak DJ (2006) Carbon storage by urban soils in the United States. J Environ Qual 35:1566–1575. https://doi.org/10.2134/jeq2005.0215

Pouyat RV, Yesilonis ID, Golubiewski NE (2009) A comparison of soil organic carbon stocks between residential turf grass and native soil. Urban Ecosyst 12:45–62. https://doi.org/10.1007/s11252-008-0059-6

Qian Y, Follett RF (2002) Assessing soil carbon sequestration in turfgrass systems using long-term soil testing data. Agron J 94:930–935. https://doi.org/10.2134/agronj2002.9300

Qian YL, Bandaranayake W, Parton WJ et al (2003) Long-term effects of clipping and nitrogen management in turfgrass on soil organic carbon and nitrogen dynamics: the CENTURY model simulation. J Environ Qual 32:1694–1700. https://doi.org/10.1108/17506200710779521

Raciti SM, Groffman PM, Fahey TJ (2008) Nitrogen retention in urban lawns and forests. Ecol Appl 18:1615–1626

Raciti SM, Groffman PM, Jenkins JC et al (2011a) Accumulation of carbon and nitrogen in residential soils with different land-use historires. Ecosystems 14:287–297

Raciti SM, Burgin AJ, Groffman PM et al (2011b) Denitrification in suburban lawn soils. J Environ Qual 40:1932–1940. https://doi.org/10.2134/jeq2011.0107

Robbins P, Birkenholtz T (2003) Turfgrass revolution: measuring the expansion of the American lawn. Land Use Policy 20:181–194. https://doi.org/10.1016/S0264-8377(03)00006-1

Salt Lake County Assessor (2008) Salt Lake County parcel database. http://slco.org/assessor/data/, Updated: 22 May 2017. Last Access: 31 July 2017

Sherrod LA, Dunn G, Peterson GA, Kolberg RL (2002) Inorganic carbon analysis by modified pressure-calcimeter method. Soil Sci Soc Am J 66:299–305

Shi W, Muruganandam S, Bowman D (2006) Soil microbial biomass and nitrogen dynamics in a turfgrass chronosequence: a short-term response to turfgrass clipping addition. Soil Biol Biochem 38:2032–2042. https://doi.org/10.1016/j.soilbio.2006.01.005

Silver WL, Neff J, McGroddy M et al (2000) Effects of soil texture on belowground carbon and nutrient storage in a lowland Amazonian forest ecosystem. Ecosystems 3:193–209. https://doi.org/10.1007/s100210000019

R Core Team (2017) R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/

Trammell TLE, Pataki DE, Cavender-Bares J et al (2016) Plant nitrogen concentration and isotopic composition in residential lawns across seven US cities. Oecologia 181:271–285. https://doi.org/10.1007/s00442-016-3566-9

United States Census Bureau, American Community Survey, Five-year Estimates of Median Household Income at the Block Group Scale, http://www.census.gov. Accessed 31 March 2016

Utah State Geologic Survey (2017) Areal imagery collection. https://geodata.geology.utah.gov/imagery/. Accessed 14 June 2017

Venables WN, Ripley BD (2002) Modern applied statistics with S, 4th edn. Springer, New York. ISBN: 0-387-95457-0

Wagner SW, Hanson JD, Olness A, Voorhees WB (1998) A volumetric inorganic carbon analysis system. Soil Sci Soc Am J 62:690–693

Wakefield JH (1933) A study of the plant ecology of the Salt Lake and Utah Valleys before the Mormon immigration. MS thesis, Brigham Young University, p 56

Wang W, Haver D, Pataki DE (2014) Nitrogen budgets of urban lawns under three different management regimes in southern California. Biogeochemistry 121:127–148. https://doi.org/10.1007/s10533-013-9942-1

WorldClimate (2008) http://www.worldclimate.com, Updated 12 Sep 2003. Accessed 2008

Zhang Y, Qian Y, Bremer DJ, Kaye JP (2013) Simulation of nitrous oxide emissions and estimation of global warming potential in turfgrass systems using the DAYCENT model. J Environ Qual 42:1100. https://doi.org/10.2134/jeq2012.0486

Zhu WX, Hope D, Gries C, Grimm NB (2006) Soil characteristics and the accumulation of inorganic nitrogen in an arid urban ecosystem. Ecosystems 9:711–724. https://doi.org/10.1007/s10021-006-0078-1

Zirkle G, Augustin B, Lal R (2011) modeling carbon sequestration in home lawns modeling carbon sequestration in home lawns. J Environ Qual 46:808–814

Zuur A, Ieno EN, Elphick CS (2010) A protocol for data exploration to avoid common statistical problems. A protocol for data exploration to avoid common statistical problems 1:3–14