

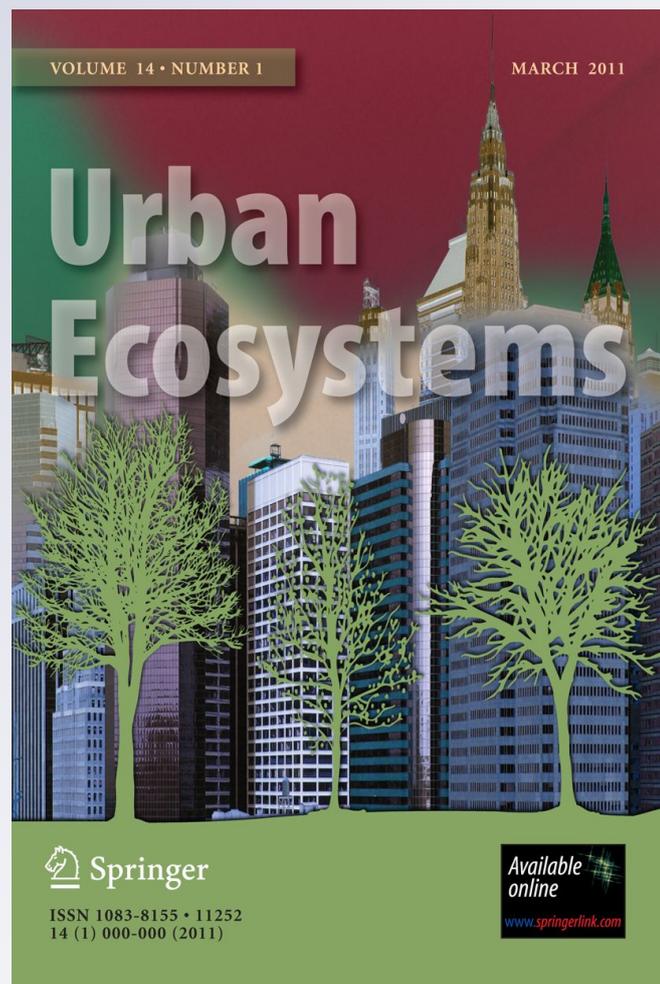
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The residential landscape: fluxes of elements and the role of household decisions

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Abstract We assessed biogeochemical cycling of elements through residential household landscapes to evaluate the importance of annual to decadal household-level decisions for element fluxes that contribute to urban and regional pollution. We combined a mailed survey, vegetation measurements, and allometric and biogeochemical models to estimate fluxes and accumulation of carbon (C), nitrogen (N), and phosphorus (P) in landscapes of 360 single-family homes in the Minneapolis-Saint Paul, Minnesota metropolitan area. Carbon inputs and accumulation were strongly influenced by the presence of trees on the property. Nitrogen inputs to the landscape exceeded estimated ecosystem demand for N on average by 51% and were dominated by N fertilizer application. Because Minnesota state law restricts the use of P fertilizer, pet waste was responsible for 84% of P inputs to the landscape. The results have implications for understanding sources of urban pollution and the potential flexibility (i.e., the potential for change) in household behaviors such as tree planting, fertilization, and pet waste management that contribute to such pollution.

Keywords Urban ecology · Carbon · Nitrogen · Phosphorus · Biogeochemistry · Turfgrass lawn

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Introduction

The household landscape represents a key feature in the residential areas of cities and in particular for single-family, detached homes. Residential landscapes (yards) have often been considered for their aesthetic value (Byrne 2005) and, in this context, they have been used as a proxy for lifestyle choices (Grove et al. 2006). More recently, residential landscapes, especially trees, have received attention regarding their role in ecosystem services, such as mitigating air pollution (Nowak et al. 2006), reducing nonpoint source pollution (Baker et al. 2008), and reducing energy use through shading and wind protection (Parker 1983). In the past decade there has been a growing body of work aimed at describing urban landscapes, including lawns, as complex biogeochemical systems in which fluxes of elements are affected by and result from coupled human-biophysical interactions (Kaye et al. 2005; Byrne and Grewal 2008).

Our previous work in the Upper Midwest of the United States estimated that landscape element input fluxes represent 9%, 25%, and 1% of the total household inputs of carbon (C), nitrogen (N), and phosphorus (P), respectively (Fissore et al. 2011). Although small relative to the average budget of an entire household, biogeochemical cycling through residential landscapes is environmentally relevant for several reasons. First, inputs of elements such as N and P to landscapes are potentially lost downstream or downwind, with detrimental ecosystem or human health consequences. Second, residential landscapes may accumulate elements in wood and soil, thereby mitigating losses or offsetting emissions from other household activities (e.g., fossil fuel CO₂ emissions; Qian et al. 2003; Fissore et al. 2011). Third, there is large flexibility associated with some fluxes that relate to household choices concerning landscape management, suggesting that households can change these behaviors (Fissore et al. 2011; Nelson et al. 2008). Knowledge regarding inputs and cycling of major elements to household landscapes and how human management decisions affect these processes would inform policies intended to reduce urban pollution.

Fluxes of elements through residential landscapes are largely influenced by human decisions made at the household scale (Nelson et al. 2008.), although large-scale factors such as atmospheric deposition and climate also play a role. Therefore, social factors potentially affect landscape biogeochemistry through their influence on landscaping decisions (Byrne and Grewal 2008; Nelson et al. 2008). Decisions that most directly affect the landscape ecosystem and its biogeochemical fluxes are made on various timescales, from decadal timescales, such as decisions regarding initial landscape contouring and the type and number of trees to plant, to annual timescales, such as decisions concerning rate and timing of fertilizer application, leaf and grass clipping management, and irrigation regime.

Because of its large potential environmental impact, turfgrass N fertilizer application is among the most studied management activities with effects on landscape biogeochemistry. Household landscapes have been suggested to be net retainers of N on average (Wollheim et al. 2005; Groffman et al. 2009), but N cycling in household landscapes is complex and strongly influenced by management practices (Qian et al. 2003; Groffman et al. 2009), along with physical factors such as slope, climate, and soil type (Baker et al. 2008). Consequently, the potential for household-to-household variability is high. Fertilizer application, of both N and P, is a common practice on residential lawns throughout the U.S., with varying amounts of fertilizer being applied among households (Law et al. 2004; Cheng et al. 2008; Fissore et al. 2011). In the past few years, legislation restricting the use of P fertilizer has been enacted at the local and state level throughout the United States, including Minnesota and Maine (Rosen and Horgan 2005; Ron Struss, Minnesota

Department of Agriculture, *pers. comm.*) as well as a number of cities and counties in several states. However, in most areas of the United States, there is no regulation of P fertilizer application to lawns.

If input fluxes of N and P through the household landscape are in excess of ecosystem accumulation (i.e., by plants and soil), there is the potential for losses of N and P to the surrounding environment. Such losses may contribute nutrients to runoff with potential negative implications for water quality in lakes and streams, to the atmosphere as reactive gases (in the case of N, Bijoor et al. 2008), and to groundwater as nitrate, posing a human health risk if consumed (Frank et al. 2005). Despite the high N retention potential of lawns, the mechanisms by which N is retained in urban ecosystems are poorly known (Wollheim et al. 2005). Furthermore, N retention capacity decreases over time following land development, increasing the risk of N losses from older residential landscapes (Petrovic 1990; Raciti et al. 2008). Nitrogen losses may also be exacerbated by chronic, high N fertilizer application, especially in conjunction with intense irrigation regimes.

In addition to fertilizer use, other components of landscape element budgets that are influenced by household decisions, such as pet waste and yard waste, likely affect landscape biogeochemistry, but to unknown degrees. While a number of studies have examined individual components of household landscape fluxes of N and P, such as the effect of atmospheric N deposition on turfgrass N cycling (Raciti et al. 2008) or the amount of P exported via runoff (Petrovic et al. 2005; Soldat et al. 2009), there are no comprehensive analyses of the biophysical and human drivers of household landscape element budgets and how they vary among households.

Here we assess the magnitude and variability of input and output fluxes of C, N, and P and accumulation for residential landscapes in Ramsey and Anoka counties in the Minneapolis–Saint Paul metropolitan area, Minnesota. We evaluate the drivers of these fluxes and investigate how household landscape management affects these fluxes. We hypothesize that C, N, and P input and output fluxes are driven by different factors and that household choices concerning landscape management have large effects on landscape biogeochemical fluxes and element accumulation in the landscape. Further, we hypothesize that there is wide variation in landscape management practices among households and the distribution of landscape elemental fluxes will be highly skewed. We focused on C, N, and P because of their overriding importance in contributing to climate change, atmospheric pollution, and water quality in urban and downstream or downwind ecosystems. We tested our hypotheses and estimated landscape elements fluxes with a combined approach that included information on household-specific landscape management practices with an on-the-ground vegetation assessment to develop flux estimates for 360 households.

Methods

Our study of residential landscapes focused on a sample of 360 single-family, detached, owner-occupied homes along a 55 km gradient of housing density (Hammer et al. 2004) in Anoka and Ramsey counties in Minnesota, with the city of Saint Paul at the urban core (Fig. 1). These 360 households were randomly selected from a pool of 1,517 households that responded to a survey mailed to 15,000 households in April 2008 and that also gave us permission to measure vegetation on their property (Fissore et al. 2011). Our survey design was based on a completely randomized sampling of owner-occupied, single-family houses in proportion to housing density. Our approach integrated mail survey-based information that was relevant for estimating landscape fluxes of C, N, and P; field measurements of

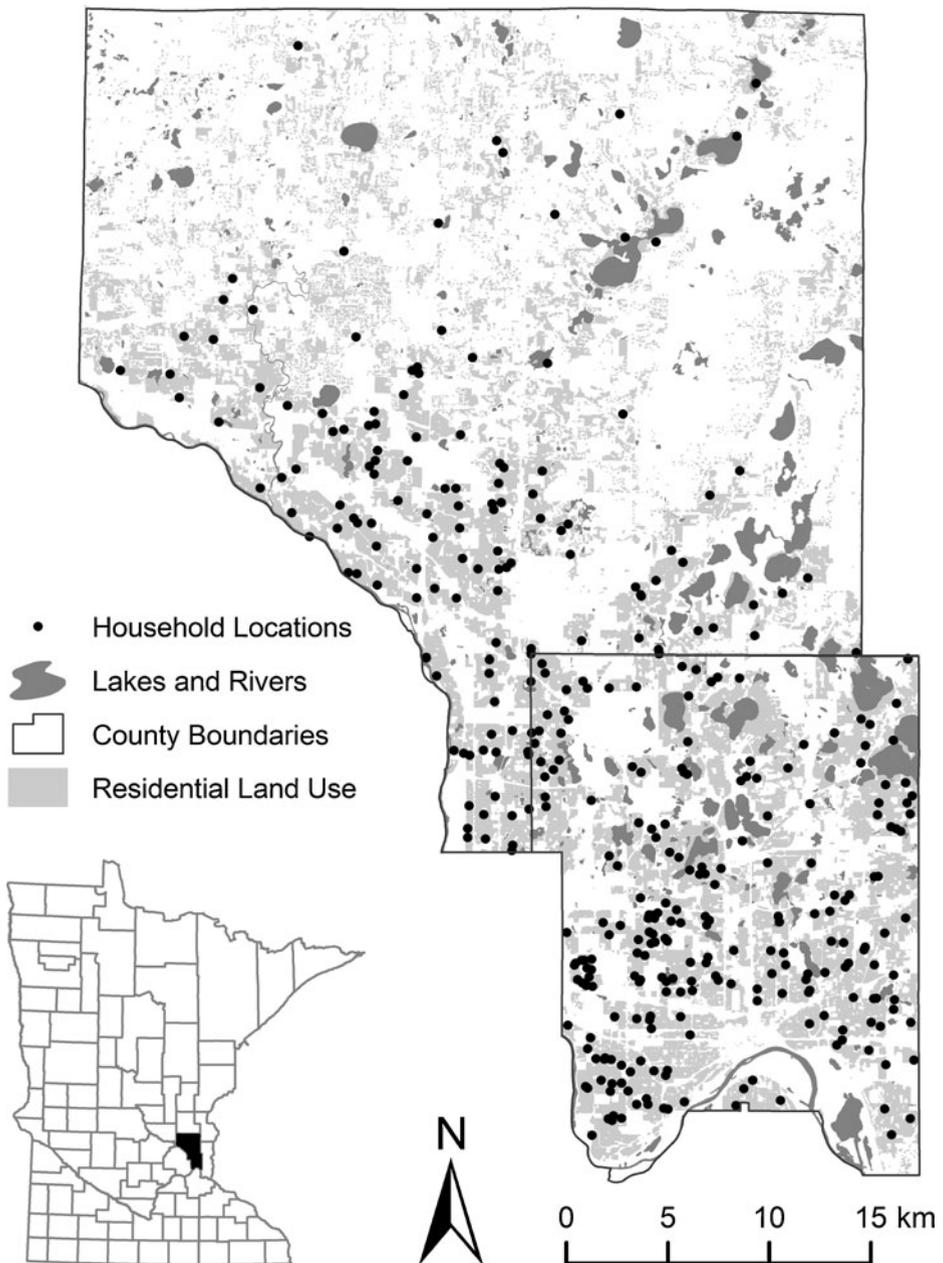


Fig. 1 Location of the 360 surveyed households in Ramsey and Anoka counties, Minnesota. Each dot represents a single household. Specific household location was randomly shifted to protect the anonymity of survey respondents. Data source: ESRI, MN DNR, Metropolitan Council; TCHEP. Projection: UTM Zone 15N NAD 83. Shaded areas represent residential land use as of 2005

trees; available data and model output from the literature; and computational tools that are described in detail in Fissore et al. (2011). We chose to estimate fluxes using some

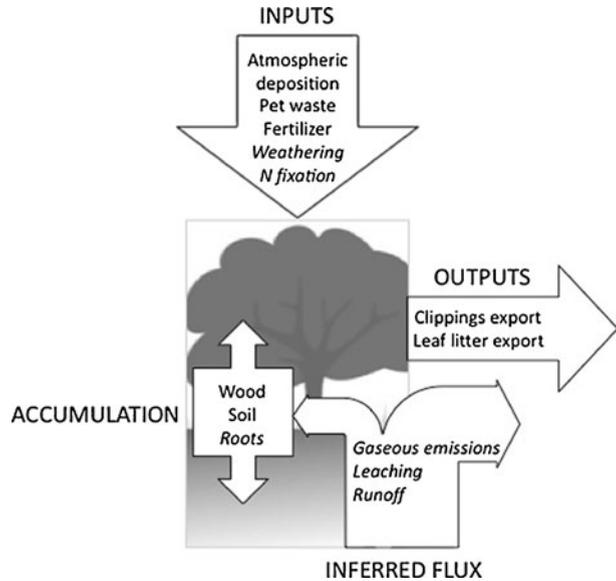
simplifying assumptions so that we could quantify fluxes for a large number of households in order to (1) assess household-to-household variability and (2) relate fluxes to socioeconomic factors (Nelson et al. 2008). Given the large number of households assessed, making on-the-ground measurements of landscape fluxes was not feasible.

Some of the questions in the mail survey (Nelson et al. 2008) specifically targeted households' landscape management practices, such as fertilizer application, irrigation practices, and disposal of lawn clippings and leaves. In addition, we asked questions concerning the numbers and sizes of pets (dogs or cats), as dog waste contributes to landscape element cycling (Fissore et al. 2011). Field measurements of vegetation were conducted between May and August of 2008 and included those parameters required by the Urban Forest Effects (UFORE, Nowak et al. 2008) model to estimate tree net primary production (NPP). The measurements included tree diameter at breast height, total height, height of first branching, canopy light exposure, percent of canopy missing, and dieback (as percent dead branches). Vegetation measurements were conducted for the entire parcel for the majority of properties. For a limited number of properties that were larger than 0.1 ha and included large wooded areas (48 households or 13% of the total), we sampled five random, 8-m radius, non-overlapping subplots per hectare and then scaled the vegetation measurement to the total parcel area (excluding buildings and driveways). Data from field vegetation assessments were recorded in a relational database and analyzed by the USDA Forest Service using the UFORE model.

All gathered information was entered into the "landscape" component of the Household Flux Calculator (HFC), a computational tool we developed that estimates fluxes and accumulation of C, N, and P through households, including their landscapes (Fissore et al. 2011). Briefly, the HFC accounts for inputs of C, N, and P to the landscape via NPP of tree leaves, tree wood, and turfgrass; atmospheric deposition (N and P); fertilizer application (N only, as Minnesota law restricts P fertilizer use on lawns); dog excreta (C, N, and P); and gasoline to fuel lawnmowers (C). Fluxes leaving the household landscape include leaf litter and soil organic matter decomposition (C), grass clipping and leaf litter removal (C, N, and P), dog feces decomposition (C), dog feces disposal (C, N, and P), and gasoline combustion in lawnmowers (C). All C that does not accumulate in the landscape is assumed to eventually leave the system as CO₂ to the atmosphere. The HFC allows accumulation of elements in soil and wood, and the difference between inputs and the sum of accumulation plus known exports (as leaves and lawn clippings) represents elements in excess or deficit of ecosystem demand (some of which presumably leave the household landscape as runoff, volatilization, nitrification, denitrification, etc.; Fig. 2). With the available data we were unable to partition such flux among potential pathways; thus we combined these into a single term, hereafter called the "inferred" N or P flux (Fig. 2, Table 1). This flux can be positive or negative, depending on whether inputs to the landscape are, respectively, more or less than accumulation in soil and wood plus known exports through leaf or lawn clippings removal.

The HFC uses the number of fertilizer application events per year reported by survey respondents to calculate annual mass of N applied, assuming that each application occurs at the rate of 48.9 kg N ha⁻¹ per application event (i.e., the nominal 1 lb per 1,000 square feet application rate recommended on retail fertilizer packages) and that fertilizer is evenly applied to the entire landscape area. For those households that rely on lawn care companies to fertilize their lawns, the HFC applies the value for the most commonly used company among the surveyed households, corresponding to 159 kg N ha⁻¹ year⁻¹ (TrueGreen®, *pers. comm.*), scaled to the landscape area. In addition, we used available data concerning gasoline combustion by lawn mowers (Christensen et al. 2001) and local data on N and P in

Fig. 2 Schematic representation of landscape fluxes of nitrogen and phosphorus, as accounted for in the Household Flux Calculator (HFC, Fissore et al. 2011). Flux components in *italics* were not individually assessed, either through direct measurements or calculation, in the HFC (e.g. P inputs due to weathering, but could be potentially relevant for the total household landscape element budget and for understanding households' impact on environmental biogeochemistry. Inferred fluxes of nitrogen and phosphorus include gaseous losses, leaching, and runoff. In the HFC, they were calculated as the difference between input and output fluxes after taking into account soil and tree wood accumulation



atmospheric deposition from the nearest available monitoring sites (N data from NADP Cedar Creek 2008, and P data from Barr Engineering Inc 2004).

For the tree contribution to landscape element fluxes, we did not estimate gross primary production and plant respiration as explicit C fluxes. Rather, we calculated leaf NPP ($\text{kg C m}^{-2} \text{ year}^{-1}$) from UFORE's (Nowak et al. 2008) leaf biomass output by assuming that annual litterfall = leaf NPP = leaf biomass (kg m^{-2})/leaf lifespan (year) (leaf lifespan from the Glopnet database, Wright et al. 2004). We used species-specific leaf N and P concentrations (Wright et al. 2004) along with nutrient resorption equations from Kobe et al. (2005) to determine litter N and P concentrations, and then we multiplied litter nutrient concentrations by litterfall to determine litterfall N and P fluxes. For leaf litter remaining on site, we assumed that litterfall was in equilibrium with litter decomposition such that all litterfall C was matched by equivalent respiration each year, and all litter N and P entered the soil and contributed to internal recycling. In our model, disposal of leaves outside the property caused all elements to exit the system, representing a net export. We calculated wood NPP ($\text{kg C m}^{-2} \text{ year}^{-1}$) from UFORE output and the N and P in wood production based on average wood C:N and C:P stoichiometry (Rodin and Bazilevich 1967). Because of lack of necessary information, we ignored C, N, and P fluxes associated with tree root NPP, removal of woody yard waste, and decomposition of dead wood (e.g. fallen branches). As a result we likely have underestimated tree nutrient uptake and therefore overestimated inferred fluxes of N and P and losses of C from wood removal and decomposition.

Turfgrass C fluxes (lawn NPP, heterotrophic respiration, and clippings removal fluxes) were estimated from the Biome-BGC ecosystem process model run for turfgrass in Minneapolis, Minnesota (Milesi et al. 2005, C. Milesi, *pers. comm.*). From the model output, we generated response surfaces for C fluxes as a function of fertilization rate (separately for clippings removed or left in place), and further modified model-estimated C fluxes based on irrigation practices (see Fissore et al. 2011). We estimated turfgrass C, N, and P concentrations from published and unpublished values (Jiang et al. 2000; Kopp and Guillard 2002; Kussow 2004, J. Oleksyn and P. B. Reich, unpublished data) and calculated N and P export fluxes associated with lawn clippings removal by dividing annual clippings

Table 1 Input, output, accumulation, and inferred carbon, nitrogen, and phosphorus fluxes calculated for 360 households in the Minneapolis–Saint Paul, Minnesota metropolitan area. Inferred fluxes of nitrogen and phosphorus include multiple component fluxes that were not estimated separately (Fig. 2)

		Inputs			Outputs			Accumulation			Inferred				
		Mean	s.e.	Component	Mean	s.e.	Component	Mean	s.e.	Component	Mean	s.e.	Component	Mean	s.e.
		kg element household ⁻¹ year ⁻¹													
Component		Mean	s.e.	Component	Mean	s.e.	Component	Mean	s.e.	Component	Mean	s.e.	Component	Mean	s.e.
a. Carbon	Grass NPP	315.8	29.8		269.4	25.3	Soil	37.4	4.4						
	Tree wood NPP	283.6	36.8	Soil heterotrophic respiration	53.9	10.3	Tree wood	283.6	36.8						
	Leaf biomass NPP	164.2	20.2	Leaf export	110.3	18.3									
	Dog excreta	0.4	0	Leaf decomposition	0.4	0									
b. Nitrogen				Dog excreta decomposition	9.0	2.4									
	Atmospheric deposition	1.5	0.2	Lawn clipping export	1.0	0.2	Soil	3.9	0.4	multiple	7.4	1.0			
	Fertilizer application	11.5	1.2	Leaf export	0.5	0.1	Tree wood	1.7	0.2						
	Dog excreta	1.4	0.2	Lawn clipping export											
c. Phosphorus	Atmospheric deposition	3.6e ⁻²	4.1e ⁻³	Leaf export	7.0e ⁻²	1.0e ⁻²	Soil	0.6	6.0e ⁻²	multiple	-0.7	0.1			
	Dog excreta	0.2	2.0e ⁻²	Grass clipping export	8.0e ⁻²	2.0e ⁻²	Tree wood	0.2	2.0e ⁻²						

C production by turfgrass C:N or C:P ratios. Carbon accumulation in landscape soil was estimated as the difference between NPP (minus clippings C exports, if any) and heterotrophic respiration (Milesi et al. 2005, C. Milesi, *pers. comm.*). Associated accumulation of N and P in soils was estimated from literature soil C:N and C:P values for turfgrass systems (Elliott 1986; Horgan et al. 2002).

As described above, the HFC allows for C, N, and P accumulation in soil and wood (Fig. 2). If N and P inputs exceeded ecosystem demand (nutrient uptake by wood and soils plus any exports as tree leaves or lawn clippings), inferred fluxes were positive, and we assumed that excess nutrients were lost, likely through a number of unspecified pathways, including runoff, leaching, and gaseous emissions (N only). If, on the other hand, N and P inputs were insufficient to meet ecosystem uptake requirements, inferred fluxes were negative, and we assumed no net loss.

In addition, we made the following simplifying assumptions for the sake of estimating biogeochemical fluxes for a large number of households. We assumed that: (1) the tree portion of household landscapes always accrued C in wood while turfgrass lawns lost or gained C, depending on management (Milesi et al. 2005) (i.e. landscapes were never at steady state); (2) there were no inputs from N fixation, so we may have underestimated inputs in yards with significant symbiotic N fixers (e.g., *Trifolium repens*); (3) nutrient retention was independent of site characteristics such as slope and soil texture; (4) C:N and C:P ratios of soils and plant organs within species were invariant; (5) a fixed percentage (40%) of dog excreta ended up in the landscape (Swann 1999) and only in the dog owners' landscape, and cats used litter boxes exclusively; and (6) weathering inputs of P to the landscape are nil.

Statistical analysis

We used regression analyses to assess the effects of landscape size, number of trees, and tree density on C inputs and accumulation and to explore the relationship between N fertilizer application and inferred N flux. We fit two continuous distributions to element fluxes, the Gaussian (normal) and the gamma distribution, and calculated the skewness. We tested the goodness-of-fit of flux distributions using the Shapiro-Wilk test for the normal distribution and Cramér-von Mises W^2 for the gamma distribution. We used paired t-tests to assess whether there were significant differences between inferred P fluxes when pet waste was included or not and differences in landscape size and tree cover. For our analyses we used JMP-SAS version 7 (SAS Institute Inc., Cary, CA), and in all cases significance was assessed with $\alpha=0.05$.

Results

On average, among our sample of 360 household landscapes, landscape C input fluxes were dominated by plant NPP, N input fluxes by fertilizer application, and P input fluxes by pet waste and atmospheric deposition (Table 1). Dominant output fluxes included heterotrophic respiration in the case of C, inferred losses for N, and leaf and lawn clipping removal for P. Accumulation occurred in tree wood and soil for all elements and averaged 42%, 38%, and 350% of landscape inputs for C, N, and P, respectively.

Total C input fluxes were dominated by plant NPP, with minor (less than 1%) contributions from pet waste and gasoline used by lawn mowers. Total C input to the landscape averaged 765 kg C household⁻¹ year⁻¹ (Table 2) and was significantly and

Table 2 Main statistical parameters describing distributions of total carbon, nitrogen, and phosphorus input fluxes through 360 household landscapes, expressed on a per household and on an areal basis

Element flux	Total C input		Total N input		Total P input		
	kg C household ⁻¹ year ⁻¹	kg C m ⁻² year ⁻¹	kg N household ⁻¹ year ⁻¹	g N m ⁻² year ⁻¹	kg P household ⁻¹ year ⁻¹	mg P m ⁻² year ⁻¹	
Mean	765	0.51	14.4	11.5	0.22	289.8	
Median	355	0.47	8.12	8.6	0.03	24.7	
S.D.	1466	0.23	25.8	7.7	0.38	641.3	
Skewness	4.7	1.6	6.7	0.6	3.3	3.8	
Normal goodness-of-fit	<i>p</i>	<0.00001	<0.001	<0.00001	<0.0001	<0.00001	<0.00001
Gamma distribution parameters							
	α	0.8636	5.2278	0.7794	1.5034	0.5697	0.4350
	σ	885.9020	0.0961	18.4270	0.0076	0.4585	0.0007
Gamma goodness-of-fit	<i>p</i>	<0.001	>0.25	0.0015	<0.001	<0.001	<0.001

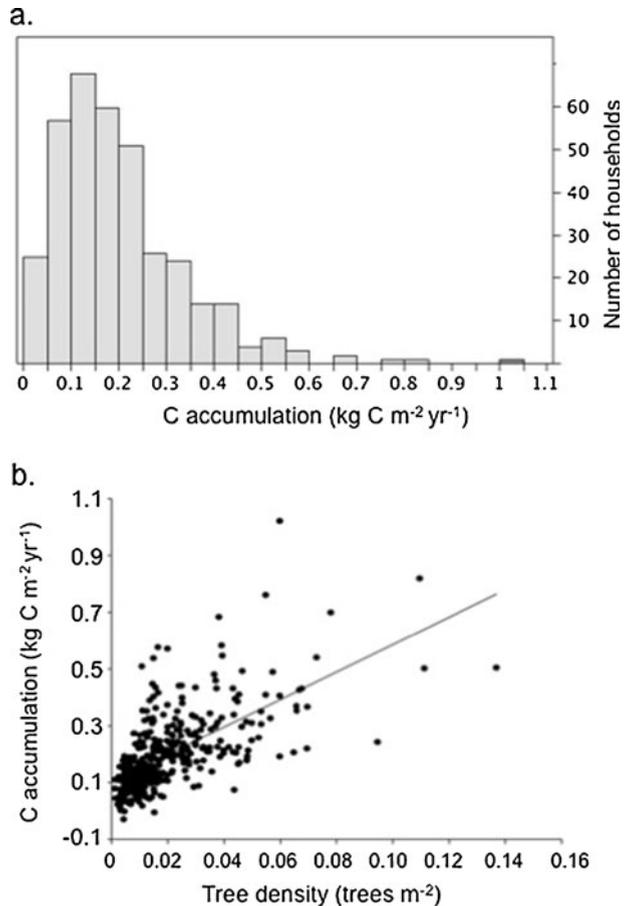
positively related to the property size ($p < 0.0001$, $R^2 = 0.71$) and to the number of trees on the property ($p < 0.0001$, $R^2 = 0.84$), which also increased with increasing property size ($p < 0.0001$, $R^2 = 0.55$). Lawn size, number of trees, and tree density all varied greatly among households (Table 3). Total landscape C inputs were highly skewed (Table 2); however, on an areal basis, C inputs to the household landscape were nearly normally distributed (Table 2), indicating that property size was largely responsible for the skewness of total C input fluxes. Outputs of C occurred mainly as CO₂ emitted to the atmosphere through heterotrophic respiration from soil and decomposing leaf litter that was left on site (Table 1), whereas leaf and lawn clipping removal combined represented approximately 14% of total output fluxes. Carbon accumulation in the landscape (calculated as the difference between input and output fluxes) was significantly and positively related to property size ($p < 0.0001$) and number of trees on the property ($p < 0.0001$). On an areal basis, C accumulation was skewed to the left (Table 2, Fig. 3a) and was significantly and positively related to tree density (Fig. 3b).

Total landscape N input fluxes averaged 14.4 kg N household⁻¹ year⁻¹ (11.5 g N m⁻² year⁻¹) and were significantly skewed (Table 2). These fluxes were dominated by N fertilizer application, which accounted for about 80% of total N input fluxes, followed by atmospheric deposition and pet waste, each representing approximately 10% of total input fluxes (Table 1). Because we assumed that N fertilizer was applied homogeneously to the

Table 3 Main biophysical characteristics of household landscapes in the Minneapolis–Saint Paul metropolitan area ($n = 360$)

	Mean	s.e.	Minimum	Maximum
Lawn size (m ²)	1457	164	162	37555
Number of trees	40	6	1	1057
Tree density (trees ha ⁻¹)	205	10	12	1368

Fig. 3 a. Distribution of landscape carbon accumulation (per unit area) and b. Relationship between landscape carbon accumulation and number of trees on the property



entire property, total N fertilization was directly related to property size in our calculations. The distribution of N fertilizer application rates was significantly skewed among households, with 28% of the households reporting that they did not apply fertilizer at all (Fissore et al. 2011). On average, household rates of N fertilizer applications were approximately eight times those estimated for wet and dry atmospheric N deposition (Table 1). There was no correlation between the rate of N fertilizer applied and the export fluxes associated with lawn clipping and leaf removal. Nitrogen fertilizer application on average exceeded those exports summed with N accumulation in wood and soil, resulting in a positive inferred N flux that was 82% of total output fluxes and 51% of inputs. Over two-thirds of landscape N accumulation occurred in soil, and the remainder accumulated in wood (Table 1). Landscape N accumulation represented approximately 38% of total N inputs.

Given the statewide P fertilizer restriction in Minnesota, we assumed that homeowners were not applying P fertilizer to their lawns. Inputs of P to the household landscape were estimated to be $0.2 \text{ kg P household}^{-1} \text{ year}^{-1}$ (Table 2) and were dominated by pet waste, which accounted for 84% of total landscape P input fluxes, with atmospheric P deposition accounting for the rest (Table 1). We assumed that atmospheric deposition was uniformly distributed across the entire property; therefore larger properties had higher total P input

fluxes, on a household basis. Seventy-six percent of total estimated P accumulation occurred in soil, with the remainder in wood. In contrast to N, after accounting for P exported as lawn clippings and leaves, we estimated a large negative inferred P flux, indicating that P inputs are not in excess of ecosystem demand, and that uptake from P stored in soil plus weathering inputs likely are necessary to meet annual plant demand for this element.

Landscape management practices such as number of trees, N fertilization rates, irrigation regime, and leaf and lawn clipping removal (Tables 3, 4 and 5) varied considerably among households. As hypothesized, management practices affected element fluxes. Among our 360 households, irrigation practices were largely related to fertilization practices, as zero fertilization (28% of households) corresponded primarily to no or low irrigation, whereas use of fertilizer was typically associated with more frequent irrigation (Table 4). Conversely, of households that applied fertilizer, only 14% used irrigation rarely or never, compared to 49% for those that did not apply N fertilizer (Table 4).

In addition to N fertilizer, landscapes recycled N from leaf litter and lawn clippings left on site, which can contribute to the excess landscape N (Table 1) already mentioned. Of the households that left both lawn clippings and leaves on site and that also fertilized (29% of total), all showed N to be in excess of ecosystem demand. However, landscape management practices alone were not sufficient to predict where N loss would occur. For example, among those households that did not apply N fertilizer, 55% showed N to be in excess of ecosystem demand, resulting in positive inferred N fluxes, and presumably a net loss of N from the household landscape. Among these households showing excess N, almost all left grass clippings on site. Also, of households with estimated positive inferred N flux (i.e., N loss), 39% had at least one dog compared to only 27% in the households with no estimated excess N. Notably, households showing positive inferred N fluxes had lower tree cover than those with no excess N (on average 16 versus 57 trees per household). Other variables could have influenced the N losses inferred using our mass balance approach, but it was not possible to identify them with the approach used here.

Dog waste contributed to inputs of N and, more significantly, P to the landscape. About 30% of the households had at least one dog and dog waste contributed to 10% and 84% of total N and P input fluxes, respectively, among all 360 households of this study. When considering only households with dogs, dog waste represented 23% and 93% of total household N and P input fluxes, respectively. Dog waste input had an effect on N and P inferred fluxes (Fig. 4), with presence of dogs resulting in greater excess N flux (i.e., greater inferred N flux) and reduced negative inferred P fluxes.

Table 4 Irrigation regime for landscapes not fertilized and fertilized with nitrogen. Values represent % of total within each group (fertilized or not-fertilized)

Irrigation regime	Not-fertilized (% of total)	Fertilized
Rarely/never	49	14
Occasionally	45	47
Regularly	6	39

Irrigation regimes were self-reported by homeowners through a mailed survey. Frequency expressed as follows: *Regularly*; once or more per week; *Occasionally*; when grass is dry

Table 5 Leaf litter and lawn clipping management practices among the 360 sampled households

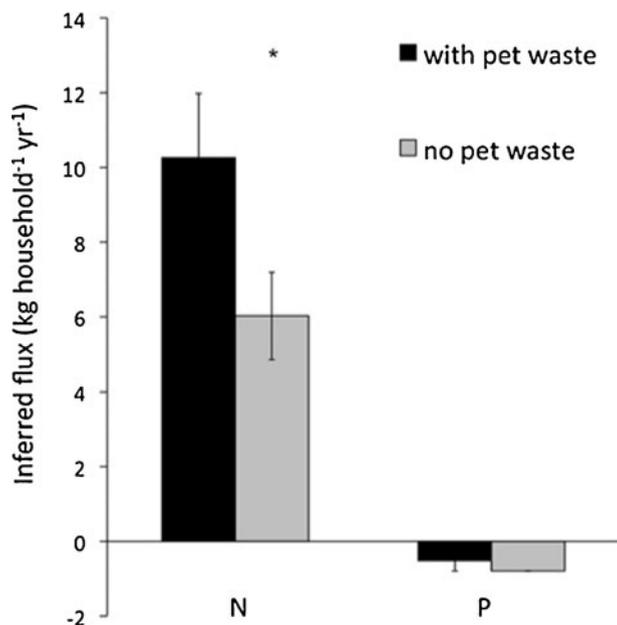
	% of Total households
Lawn clippings left on site	85
Lawn clippings removed	15
Leaves left on site	42
Leaves removed from entire property	41
Leaves removed from part of property	16
Both leaves and lawn clippings removed	11
Both leaves and lawn clippings left on site ^a	57

^a Includes households that remove leaves from only part of the property

Discussion

As hypothesized, biogeochemical fluxes of C, N, and P were each driven by different specific factors, which in turn were affected by household choices regarding landscape management. The abundance of trees was the most important factor affecting landscape C inputs and accumulation. By contrast, landscape N inputs were dominated by lawn fertilizer, while the soil, rather than trees, was most important for N accumulation. Landscape P inputs in the state of Minnesota, where legislation restricts P fertilizer use, were dominated by pet waste, and, similar to N, most P accumulated in soil. The skewed nature of input flux distributions indicated large household-to-household variability and a disproportionate contribution of a small number of households to total fluxes across all households, suggesting that changes in household-specific activities can have a large effect on household landscape biogeochemistry.

Fig. 4 Inferred fluxes of N and P for households that did or did not have pet waste. An asterisk indicates a significant difference between the two categories of households



The household choice to plant trees or to retain existing trees on the property at the time it was developed, rather than opting for a turfgrass-dominated landscape, strongly influenced total C input fluxes through the landscape. Larger properties had more trees and, consequently, higher total C inputs on a per household basis. Yet, on a per-area basis, C input was nearly normally distributed, indicating similar tree densities among the households' landscape space. Our estimates for landscape C inputs among 360 households covering an urban to exurban region are on the same order but slightly lower than those reported by Jo and McPherson (1995) for 50 residential units within the city of Chicago, Illinois.

Households likely make decisions regarding tree planting for a variety of reasons, both socioeconomic and related to how they value the variety of benefits provided by trees. For example, evaluating urban trees, residents of the most populated metropolitan areas in the United States valued the benefits of tree shade that provided cooling in the urban environment (Lohr et al. 2004). Thus, homeowners may consider the purchase of a house with mature trees a benefit that they want to protect (Theriault et al. 2002). Homeowners may also plant their own trees, guided by personal aesthetic choices and typically trees have been estimated to add to property value (Anderson and Cordell 1988; Sander et al. 2010). Our results indicate that these choices related to planting or removing household trees have a significant long-term (decadal) effect on landscape biogeochemistry.

Landscape output fluxes of C were dominated by soil processes (heterotrophic respiration from soil and decomposing leaf litter), and these fluxes were affected by household choices concerning vegetation cover and property size. Larger yards not only had more trees on average, but also had more lawn area. The much smaller C output flux resulting from household export of plant material (lawn clippings and leaf litter) was influenced by household choices made at short (annual) timescales and accounted for a small fraction of total landscape C input and output fluxes. Nevertheless, removing plant material reduced C return to soil and decreased its accumulation rate (Qian et al. 2003; Milesi et al. 2005), with effects also on the cycling and accumulation of N and P in soil.

Trees contribute in part to offset increasing atmospheric CO₂ due to anthropogenic emissions from fossil fuel combustion. However, our previous study (Fissore et al. 2011) has shown that households' emission offset by residential landscape through accumulation of C in soil and trees is minimal (on average 5% of household emissions). The small C emissions offset by residential landscape in the Upper Midwest of the United States suggests that, for cities in climates less conducive to supporting high tree biomass, the potential for residential landscapes to offset household fossil fuel C emissions would be even lower. In addition to C accumulation in their biomass, urban trees can carry additional environmental and energy saving co-benefits by shading houses hence reducing energy use (Simpson and McPherson 1998; Akbari 2002).

As hypothesized, N input fluxes were dominated by N fertilizer additions. Nitrogen fertilizer use is a widespread practice in households in the U.S. (Law et al. 2004) that occurs with a wide range of application rates (Petrovic 1990; Osmond and Hardy 2004). Among our surveyed 360 households, N fertilizer application rates similarly were variable, with a few households contributing disproportionately to total N fertilizer inputs among all households, and about one-third of all households reporting that they did not apply fertilizer at all (Fissore et al. 2011). These results are consistent with findings from studies in North Carolina (Osmond and Hardy 2004) and Maryland (Law et al. 2004). The large range in N fertilizer application regime reflects a highly variable, household-specific decision that is made annually, indicating that there is the potential for large flexibility (i.e. potential for change) around this behavior.

In theory, N fertilization is intended to support biotic demand for N, in part caused by removal of N in exported plant material. However, our study suggests that homeowners may not consider the removal of N in leaves and lawn clippings when deciding whether and how much to fertilize, as there was little correspondence between these two behaviors. Rather, homeowners' decisions about whether to fertilize are related to their attitudes, norms, and values (Nelson et al. 2008) and to the widespread idea that fertilizing will result in healthier and greener lawns (Nelson et al. 2008; see also Cheng et al. 2008). However, adding N fertilizer to achieve what is a personal perception of 'healthier' or 'greener' lawn might not coincide with meeting actual plant demands for N, especially if lawn clippings and leaf management are not considered as well. In fact, similar lawn quality (expressed as grass color, etc.) can be achieved by using one less fertilizer application per year if clippings are left on site instead of being removed (Guillard and Kopp 2004; Heckman et al. 2000; B. Horgan, *pers. comm.*).

The largest output flux of N from the landscape occurred via "inferred" fluxes, the term we use to indicate the unpartitioned flux of N that was in excess (or shortage) of ecosystem demand (i.e., the N required to stoichiometrically match C accumulation in wood and soil, Fissore et al. 2011). On average, N inputs exceeded ecosystem demand, meaning that, if this N was not retained in the soil, it likely was lost via gaseous losses associated with nitrification/denitrification processes or via runoff to surface waters or leaching to groundwater, with potentially negative effects on water quality (Qian et al. 2003). Losses as nitrate leaching (NO_3^-) and nitrous oxide flux (N_2O) have been directly measured to range from 0.05 to 4.1 $\text{g N m}^{-2} \text{ year}^{-1}$ for NO_3^- and 0.05 to $>0.3 \text{ g N m}^{-2} \text{ year}^{-1}$ for N_2O (Groffman et al. 2009). Our estimated 7.4 $\text{kg N household}^{-1} \text{ year}^{-1}$ inferred N flux—which includes these fluxes, among others—corresponded to 0.06 $\text{g N m}^{-2} \text{ year}^{-1}$, placing it at the lower end of previously reported values. Our model may underestimate actual N losses from the residential landscape, for example because of N losses that occur during times when inputs are high relative to uptake (early spring, late autumn) or because our approach ignored losses that might have resulted from high irrigation or rainfall events.

Removal of lawn clippings and leaf litter represented a much smaller output flux, but each was important for landscape N cycling because, together with inputs as fertilizer, they had a large effect on the magnitude of the inferred flux of N. In addition, these management practices were the result of direct household choices made on an annual time scale and, thus, may be relatively flexible. Many have suggested that current N fertilizer application rates can be reduced and ecosystem demand for N can be met by returning clippings to lawns (Heckman et al. 2000; Kopp and Guillard 2002) and mulching leaves into the lawn, thereby reducing the risk for high N losses from the landscape. Indeed, in our study all households that applied N fertilizer and also left grass clippings and leaves to decompose on site showed N to be in excess of ecosystem demand. Even though we observed a positive relationship between N fertilizer application rates and inferred N fluxes, N fertilization alone was not sufficient to explain N in excess of ecosystem demand. In fact, about half of the households that did not apply N fertilizer still showed N to be in excess of ecosystem demand. These positive inferred N fluxes appeared to be mainly related to a lower abundance of trees (hence lower N accumulation in wood) than in those landscapes in which N was lower than ecosystem demand. Also, this excess N partly was due to greater inputs of pet waste and lower grass clipping removal. These findings suggest that understanding of N cycling through the residential landscape is far from complete, and more information is needed concerning drivers of input and output fluxes of N in addition to fertilizer application. For example, N inputs would have been even higher if we had estimated N inputs from biological N fixation. In a subset of 137 of our 360 household landscapes that were surveyed for plant species

composition, 60% of yards contained *Trifolium repens* (white clover) (Knapp et al., *Phylogenetic and functional characteristics of household yard florals and their changes along an urbanization gradient*, accepted for publication; Cavender-Bares et al. *pers. comm.*), which could be contributing significant N inputs via fixation.

Landscapes have the capacity to accumulate N, mostly in soil, and N accumulation is related to the potential for soil to build up organic C. Our estimates for N accumulation in soil were derived from models of turfgrass NPP and heterotrophic respiration (Milesi et al. 2005) combined with assumptions regarding C:N stoichiometry of soils, and were dependent on landscape management practices of N fertilization, irrigation, and plant material removal, with greater net ecosystem C accumulation associated with increasing fertilization and adequate water supply. High N retention occurs under moderate management conditions, such as low fertilization and moderate irrigation (Raciti et al. 2008; Groffman et al. 2009). However, lower soil N retention (i.e., greater losses) has been shown to result from excessive landscape irrigation (Guillard and Kopp 2004; Groffman et al. 2009), especially when soils are sandy (Sharma et al. 1996). While our model accounted for the soil type in our study region, our estimates of N losses did not account for extreme irrigation or precipitation events, which may have caused us to underestimate N losses from the landscape as inferred N fluxes. As time since lawn establishment increases, lawns may approach equilibrium where inputs equal outputs, and the risk of N losses from residential landscapes increases with potential negative effects on water quality. Our model did not consider such dynamics.

In households where N fertilizer was not applied, the main inputs of N to the landscape were atmospheric N deposition and, to an almost equal extent, pet waste. Atmospheric deposition rates were on average about eight times lower than fertilizer rates, and we assumed that they were constant (per unit area) across all households. However, atmospheric N deposition at the local scale can be patchy, with higher rates near major roads with intense traffic due to N emissions from fossil fuel combustion (Baker et al. 2001). Because our model assumed a homogeneous distribution of atmospheric N across households, we likely underestimated actual N inputs as atmospheric deposition for some households and overestimated it for others.

Large uncertainty also surrounds pet waste N inputs to landscapes. In fact, very little information is available concerning the magnitude of pet waste entering the residential landscape and its fate in the urban environment. Notably, our assumption that 40% of dog owners do not pick up after their dogs in a residential landscape was based on a single survey conducted in the Chesapeake Bay area (Swann 1999). Also, our assumption that pet waste that is not picked up stays on the pet owner's property could have led us to overestimate actual N and P inputs to the landscape, but to date no data are available to examine this assumption. Our approach ignored the fact that pet waste likely affected small portions of the landscape that become hot spots of N and P rather than being homogeneously distributed through the entire landscape. Such areas of high element concentration may lead to higher losses than if elements were distributed evenly over the landscape. The literature on the fate of pet waste is limited, and for the most part it has focused on public land such as parks to understand its impacts on plant survival (EPA 2009; Sussman 2008) rather than on biogeochemical budgets.

Improved understanding of the fate of pet waste becomes particularly important when assessing P inputs to the landscape, as pet waste represented the most important P input flux to residential landscapes (no inputs from P fertilizer were included due to the state law). The large input flux associated with pet waste disposal shown by our work calls for more studies that quantify the return to the landscape of pet waste based on pet owners' habits of

pet walking and waste pick up (in city parks, within the neighborhood, or on the dog owner's property). Such studies would greatly improve understanding of residential landscape P cycling, especially for locations subject to P fertilization restrictions, where pets are the dominant P input to household landscapes.

After accounting for P accumulation, the difference between input and output fluxes of P through the landscape points to a large, negative inferred flux of P, indicative that the supply of P from external sources (e.g. P atmospheric deposition, pet waste) was not sufficient to satisfy ecosystem demand (i.e., accumulation of P in wood and soil). Our model did not account for previously accumulated P from fertilization that occurred before the fertilizer P restriction or for belowground P inputs to soil from weathering. It is likely that these sources supplied enough P to satisfy ecosystem demand, as field measurements suggest that over 70% of lawns in Minneapolis–Saint Paul, Minnesota metropolitan area were not P-limited before the fertilizer P restriction went into effect (Barten and Jahnke 1997) and that fertilizer P applied in excess of ecosystem demand either was retained in soil in insoluble forms or was exported to groundwater or surface water, contributing to water pollution. A similar correlation between P concentration in runoff and P fertilization was found across turfgrass sites in the New York area (Soldat et al. 2009). Inputs of P to soil through weathering have been estimated in the range of 0.05 to 1.0 kg P ha⁻¹ year⁻¹ across a number of ecosystems worldwide (Newman 1995), potentially satisfying a large fraction of the estimated 5 kg P ha⁻¹ year⁻¹ shortage of P across our sample. With the restriction on P fertilizer use in place, the primary pathway by which P is transported from yards to aquatic ecosystems is likely to be the flux of P in litterfall to the street from boulevard trees, a flux we did not measure.

Our work highlights the importance of residential landscapes in the context of urban biogeochemistry. Large variability in element fluxes resulted from management practice decisions, in particular regarding the abundance of trees, N fertilizer use, leaf litter and lawn clipping management, and pet waste disposal. Thus, an integrated approach that accounts for both biophysical and socio-cultural aspects of the household landscape will be needed to inform future guidelines for reducing pollution derived from residential landscapes. One major research need that follows from our work is the development of a process model of lawns that separates the inferred fluxes into components, particularly gaseous, runoff, and leaching losses. This research would further elucidate the role of household behavior and choice over various time scales, informing policies regarding urban nonpoint source pollution.

Conclusions

Improved understanding of the major inputs of elements to the household landscape, their cycling and fate in the environment, and which of these fluxes are under the direct influence of human management decisions is necessary to inform policies that aim to mitigate urban pollution. We reported high variability, hence high potential for change, associated with applying N fertilizer to lawns. Any change in this behavior will likely have a significant impact on water quality. An improved understanding of the magnitude and fate of P in pet waste and household behavior associated with pet waste management, especially in places where P application is restricted, will prove pivotal to developing policy strategies to reduce P runoff. Our work suggests that restricting P fertilizer likely results in P retention in residential landscapes, with potential positive effects on water quality, without compromising plant growth, because ecosystem demand is likely satisfied by current soil reservoirs and by inputs of P from weathering and from mineralization.

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References

- Akbari H (2002) Shade trees reduce building energy use and CO₂ emissions from power plants. *Environ Pollut* 116:S119–S126
- Anderson LM, Cordell HK (1988) Influence of trees on residential property values in Athens, Georgia (U.S. A.): a survey based on actual sales prices. *Landsc Urban Plann* 15:153–164
- Baker LA, Xu Y, Hope D, Lauver L, Edmonds J (2001) Nitrogen mass balance for the central Arizona-Phoenix (CAP) ecosystem. *Ecosystems* 4:582–602
- Baker LA, Wilson BD, Fulton D (2008) Disproportionality as a framework to target nutrient reduction from urban landscapes. *Cities and the Environment* 1, Article 7, 15 pp. <http://escholarship.bc.edu/cate/vol1/iss2/7>
- Barr Engineering Inc. (2004) Minnesota phosphorus study. Conducted for the Minnesota Pollution Control Agency, St. Paul, MN
- Barten JM, Jahnke E (1997) suburban lawn runoff water quality in the Twin Cities metropolitan area, 1996 and 1997. Report to the Suburban Hennepin Regional Park District, Maple Plain, MN
- Bijoor NS, Czimczik CI, Pataki DE, Billings SA (2008) Effects of temperature and fertilization on nitrogen cycling and community composition of an urban lawn. *Glob Chang Biol* 14:2119–2131
- Byrne LB (2005) Of looks, laws and lawns: how human aesthetic preferences influence landscape management, public policies and urban ecosystems. In: Laband D (ed) *Emerging issues along urban-rural interfaces: linking science and society conference proceedings*. Auburn University, Auburn, pp 42–46
- Byrne LB, Grewal P (2008) Introduction to ecological landscaping: a holistic description and framework to guide the study and management of urban landscape parcels. *Cities Environ* 1:1–20
- Cheng Z, Richmond DS, Salminen SO, Grewal PS (2008) Ecology of urban lawns under three common management programs. *Urban Ecosyst* 11:177–195
- Christensen A, Weterholm R, Almen J (2001) Measurements of regulated and unregulated exhaust emissions from a lawn mower with and without an oxidizing catalyst: a comparison of two different fuels. *Environ Sci Technol* 35:2166–2170
- Elliott ET (1986) Aggregate structure and carbon, nitrogen, and phosphorus in native and cultivated soils. *Soil Sci Soc Am J* 50:627–633
- Environmental Protection Agency, EPA (2009) National pollutant discharge elimination system (NPDES) factsheet http://cfpub.epa.gov/npdes/stormwater/menuofbmps/index.cfm?action=factsheet_results&view=specific&bmp=4
- Fissore C, Baker LA, Hobbie SE, King JY, McFadden JP, Nelson KC, Jakobsdottir I (2011) Carbon, nitrogen, and phosphorus fluxes in household ecosystems in the Minneapolis-Saint Paul, Minnesota, urban region. *Ecol Appl* 21:619–639
- Frank KW, O'Reilly KM, Crum JR, Calhoun RN (2005) The fate of nitrogen applied to a mature Kentucky bluegrass turf. *Crop Sci* 46:209–215
- Groffman PM, Williams CO, Pouyat RV, Band LE, Yesilonis ID (2009) Nitrate leaching and nitrous oxide flux in urban forests and grasslands. *J Environ Qual* 38:1848–1860
- Grove JM, Troy AR, O'Neil-Dunne JPM, Burch WR Jr, Cadenasso ML, Pickett STA (2006) Characterization of households and its implications for the vegetation of urban ecosystems. *Ecosystems* 9:578–597
- Guillard K, Kopp KL (2004) Nitrogen fertilizer form and associated nitrate leaching from cool-season lawn turf. *J Environ Qual* 33:1822–1827
- Hammer RB, Stewart SI, Winkler RL, Radeloff VC, Voss PR (2004) Characterizing dynamic spatial and temporal residential density patterns from 1940–1990 across the Northern Central United States. *Landsc Urban Plann* 69:183–199
- Heckman JR, Liu H, Hill W, Demilia M, Anastasia WL (2000) Kentucky bluegrass responses to mowing practice and nitrogen fertility management. *J Sustain Agric* 15:25–33
- Horgan BP, Branham BE, Mulvaney RL (2002) Mass balance of N-15 applied to Kentucky blue grass including direct measurement of denitrification. *Crop Sci* 42:1595–1601

- Jiang Z, Sullivan WM, Hull RJ (2000) Nitrate uptake and nitrogen use efficiency by Kentucky bluegrass cultivars. *Hortscience* 35:1350–1354
- Jo H-K, McPherson EG (1995) Carbon storage and flux in urban residential greenspace. *J Environ Manag* 45:109–133
- Kaye JP, McCulley RL, Burke IC (2005) Carbon flux, nitrogen cycling, and soil microbial communities in adjacent urban, native, and agricultural ecosystems. *Glob Chang Biol* 11:575–587
- Kobe RK, Lepczyk CA, Iyer M (2005) Resorption efficiency decreases with increasing green leaf nutrients in a global data set. *Ecology* 86:2780–2792
- Kopp KL, Guillard K (2002) Clipping management and nitrogen fertilization of turfgrass growth, nitrogen utilization, and quality. *Crop Sci* 42:1225–1231
- Kussow WR (2004) Phosphorus runoff losses from lawns. *Better Crops* 88:12–13
- Law NL, Band LE, Grove JM (2004) Nitrogen input from residential lawn care practices in suburban watershed in Baltimore County, MD. *J Environ Plan Manag* 47:737–755
- Lohr VI, Pearson-Mims CH, Tarnai J, Dillman DA (2004) How urban residents rate and rank the benefits and problems associated with trees in cities. *J Arboriculture* 30:28–35
- Milesi C, Elvidge CD, Dietz JB, Tuttle BT, Ramkrishna RN, Running SW (2005) Mapping and modeling the biogeochemical cycling of turf grasses in the United States. *J Environ Manag* 36:426–438
- Nelson KC, Grayzeck S, King J, Hobbie S, Baker L, McFadden JP (2008) Our household choices in urban living survey, University of Minnesota, St. Paul. <http://www.forestry.umn.edu/People/Nelson/index.htm#choices>
- Newman EI (1995) Phosphorus inputs to terrestrial ecosystems. *J Ecol* 83:713–726
- Nowak DJ, Crane DE, Stevens JC (2006) Air pollution removal by urban trees and shrubs in the United States. *Urban For Urban Green* 4:115–123
- Nowak DJ, Crane DE, Stevens JC, Hoehn RE, Walton JT, Bond J (2008) A ground-based method of assessing urban forest structure and ecosystem services. *Arboriculture Urban For* 34:347–358
- Osmond DL, Hardy DH (2004) Characterization of turf practices in five North Carolina communities. *J Environ Qual* 33:565–575
- Parker JH (1983) Landscaping to reduce the energy used in cooling buildings. *J For* 81:82–85
- Petrovic AM (1990) The fate of nitrogenous fertilizers applied to turfgrass. *J Environ Qual* 19:1–14
- Petrovic AM, Soldat DJ, Gruttadaurio J, Barlow J (2005) Turfgrass growth and quality related to soil and tissue nutrient content. *Int Turfgrass Soc Res J* 10:989–997
- Qian YL, Bandaranayake W, Parton WJ, Mecham B, Harivandi MA, Mosier AR (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
- Raciti SM, Groffman PM, Fahey TJ (2008) Nitrogen retention in urban lawns and forests. *Ecol Appl* 18:1615–1626
- Rodin LE, Bazilevich NI (1967) Production and mineral cycling in terrestrial vegetation. Oliver and Boyd, Edinburgh and London
- Rosen CJ, Horgan BP (2005) Regulation of phosphorus fertilizer application to turf in Minnesota: historical perspective and opportunities for research and education. *Int Turfgrass Sci Res J* 10:130–135
- Sander H, Polasky S, Haight RG (2010) The value of urban tree cover: a hedonic property price model in Ramsey and Dakota Counties, Minnesota, USA. *Ecol Econ* 69:1646–1656
- Sharma ML, Herne DE, Byrne JD, Kin PG (1996) Nutrient discharge beneath urban lawns to a sandy coastal aquifer, Perth, Western Australia. *Hydrogeol J* 4:103–117
- Simpson JR, McPherson EG (1998) Simulation of tree shade impacts on residential energy use for space conditioning in Sacramento. *Atmos Environ* 32:69–74
- Soldat DJ, Petrovic MA, van Es HM (2009) The effect of soil phosphorus and nitrogen and phosphorus fertilization on phosphorus runoff losses from turfgrass. In: Nett M et al. (eds) *The fate of nutrients and pesticides in the urban environment*. ACS symposium series, American Chemical Society, Washington, D.C. pp. 97–106
- Sussman D (2008) Since dogs do what they do, what can we do to improve the doo-doo situation at dog parks? Supplement to *context* alumni magazine. Conway School of Landscape Design
- Swann C (1999) A survey of residential nutrient behaviors in the Chesapeake Bay. Widener Burrows, Inc. Chesapeake Research Consortium. Center for Watershed Protection, Ellicott City, p 112
- Theriault, M, Kestens Y, Des Rosiers, F (2002) The impact of mature trees on house values and on residential location choices in Quebec City. <http://www.iemss.org/iemss2002/proceedings/pdf/volume%20due/191.pdf>
- Wollheim WM, Pellerin BA, Vörösmarty CJ, Hopkinson CS (2005) N retention in urbanizing headwater catchments. *Ecosystems* 8:871–884
- Wright IJ et al (2004) The worldwide leaf economics spectrum. *Nature* 428:821–827