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METHANE AND NITROUS OXIDE FLUXES FROM URBAN SOILS TO THE ATMOSPHERE

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Abstract. Land-use change is an important driver of soil–atmosphere gas exchange, but current greenhouse-gas budgets lack data from urban lands. Field comparisons of urban and non-urban ecosystems are required to predict the consequences of global urban-land expansion for greenhouse-gas budgets. In a rapidly urbanizing region of the U.S. Great Plains, we measured soil–atmosphere exchange of methane (CH_4) and nitrous oxide (N_2O) for one year in replicated ($n = 3$) urban lawn, native shortgrass steppe, dryland wheat–fallow, and flood-irrigated corn ecosystems. All soils were net sinks for atmospheric CH_4, but uptake by urban, corn, and wheat–fallow soils was half that of native grasslands (-0.30 ± 0.04 g $C \cdot m^{-2} \cdot yr^{-1}$ [mean $\pm$ 1 SE]). Urban (0.24 ± 0.03 g $N \cdot m^{-2} \cdot yr^{-1}$) and corn ($0.20 \pm 0.02$ g $N \cdot m^{-2} \cdot yr^{-1}$) soils emitted 10 times more N_2O to the atmosphere than native grassland and wheat-fallow soils. Using remotely sensed land-cover data we calculated an upper bound for the contribution of lawns to regional soil–atmosphere gas fluxes. Urban lawns occupied 6.4% of a 1578-km^2 study region, but contribute up to 5% and 30% of the regional soil CH_4 consumption and N_2O emission, respectively, from land-use types that we sampled. Lawns that cover small portions of the landscape may contribute significantly to regional soil–atmosphere gas exchange.

Key words: *greenhouse gases; land use; lawn management; methane; nitrous oxide; urbanization; urban soils, contribution to greenhouse-gas fluxes.*

Introduction

Atmospheric greenhouse-gas concentrations are increasing rapidly, causing detectable changes in global climate (Houghton et al. 2001). Rising CH_4 and N_2O concentrations result not only from fossil-fuel use, but also from changes in land use that affect the production and consumption of gases by soil biota (Firestone and Davidson 1989). Most research linking land use to soil–atmosphere gas exchange has focused on agricultural ecosystems (Hutchinson and Mosier 1979, Mosier et al. 1991, Matson et al. 1998, Robertson et al. 2000, Dobbie and Smith 2003), but fertilization and irrigation that alter gas fluxes in agricultural ecosystems also occur in urban lawns (Horgan et al. 2002). Urban land occupies 2–3.5% of the conterminous United States, urban counties within or adjacent to cities cover 25% of the area, and in certain large regions >50% of the area is ''developed land'' (Dwyer et al. 2000, USDA 2001, Vogelmann et al. 2001). Urban land area in the US (Dwyer et al. 2000) has doubled since 1970 and global urban land area (UN 1987:130) may have doubled since 1980.

The rapid expansion of fertilized and irrigated urban ecosystems may have several social and ecological implications. Regional greenhouse-gas budgets that do not include urban land may be missing sources or sinks important to accurate biogeochemical modeling. These sources and sinks may represent a link between household lawn-management choices, landscape architecture (e.g., lawn area in a given development style), and regional biogeochemistry. Currently, such linkages are poorly quantified because we lack field comparisons of urban and non-urban ecosystems. One case study along an urban–rural gradient showed that soil N_2O emissions increased and soil CH_4 consumption decreased at the urban end of the gradient (Goldman et al. 1995, Groffman and Bohlen 1999). However, the sampling scheme focused on unmanaged forests, avoiding urban lawns and parks. At the landscape scale, the effects of urbanization must be addressed with field measurements in common urban ecosystem types (Collins et al. 2000, Grimm et al. 2000).

In this paper we link urban, agricultural, and native ecosystems with simultaneous field measurements to quantify effects of land-use change on regional bio-

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geochemistry. One of the fastest growing regions in the United States is the Rocky Mountain Front Range (U.S. Census 2000), where rapid urbanization occurs within a matrix of semiarid grasslands and dryland agriculture dissected by irrigated agriculture near water sources. At the northern end of this metropolis, we measured soil–atmosphere exchange of CH_4 and N_2O in replicated urban lawns, native grasslands, dryland wheat fields, and irrigated corn fields. We link these plot-scale measurements with remotely sensed land-cover data to address our primary research question: Do urban lawns contribute significantly to regional greenhouse gas fluxes from soils?

METHODS

Study sites and experimental design

The urban sites were in Fort Collins, Colorado, USA (latitude: 40.6° N, longitude: 105.1° W), grassland and agricultural sites were within 20 km, and all sites were similar in elevation (1493–1620 m). Our stratified (by land use) random-sampling design used land-use types covering >70% of the grassland region of Larimer County, Colorado. We sampled Aridic Argiustolls and Ustollic Haplargids with loamy to fine surface texture because these soils cover >50% of the region. All sites had been in their current land use since at least 1938, the year of the earliest aerial photograph. We gained permission to sample 4–10 sites that met these criteria and randomly selected three sites per land-use type. At these sites, we analyzed soil texture and there were no statistical differences among the sites in surface (0 to 15 cm) or subsurface (15 to 30 cm) concentrations of sand, silt, or clay (J. P. Kaye, *unpublished data*).

The urban ecosystems are lawns dominated by Kentucky bluegrass (*Poa pratensis* L.) and bordered by trees (mostly *Ulmus*, *Populus*, and *Fraxinus* spp.). Sprinkler irrigation monitored with rain gauges from May to October totalled 54 ± 4 cm/yr (mean ± 1 SE), and nitrogen (N) fertilization in June and October totalled 11 ± 1 g N/m^2 (two applications of 5.5 g N/m^2 each based on Colorado State University Extension Service Gardening Series Fact Sheet 7.202 [available online]).[6] All urban sites were mowed frequently (0.5 to 2 times per week) from April to October with a mulching mower. Clippings were left on site to mimic lawn-management practices in the region. The cornfields were continuous corn systems without legume or wheat rotations. They received 127 ± 13 cm irrigation from June to August, and were fertilized with 15 ± 2 g N/m^2 before being planted in May. One field was fertilized in November and two were fertilized in April. As is typical in this region, the corn was harvested for silage in September or October. The dryland wheat cropping system in this region is a winter wheat–summer fallow rotation (during any given year each field is half wheat and half fallow). We sampled wheat and fallow areas simultaneously and present the mean value here. Wheat fields were planted in October and harvested in June but received no irrigation or fertilization. Native vegetation is shortgrass steppe grassland with mean annual air temperature 8.9°C and precipitation 38.5 cm/yr (100-year record at Colorado State University, Campus Weather Station, Fort Collins, Colorado, USA), and wet atmospheric inorganic N deposition 2.5 g N·m^{-2}·yr^{-1} (National Atmospheric Deposition Program [data available online]).[7] To our knowledge (based on aerial photos, ethnography, range characteristics, and soil profiles), none of the native grasslands were ever ploughed, irrigated, or fertilized, but all were grazed in the past. All native-grassland sites are dominated by *Bouteloua gracilis* (HBK) Lag. ex Steud., *Agropyron smithii* Rybd., and *Buchloe dactyloides* (Nutt.) Engelm., but exotic species (e.g., *Bromus tectorum*) were also present.

We monitored soil temperature at each site with one HOBO datalogger (Onset Computer Corporation, Bourne, Massachusetts, USA) buried 5 cm below the mineral soil surface. Soil temperature was recorded hourly and daily averages are presented here. Soil moisture was measured gravimetrically (105°C for 48 h) on a composite of six soil cores (2-cm diameter × 15-cm depth) on every gas-sampling date in which soils were not frozen.

Trace-gas measurements

The flux of CH_4 and N_2O from the soil to the atmosphere was estimated using static soil covers (Hutchinson and Mosier 1981, Mosier et al. 1991, 1997). We used 35-mL syringes with nylon stopcocks to sample the headspace of 1.2-L soil covers at 0-, 15-, and 30-min intervals. Samples were analyzed within 24 hours by gas chromatography using a Poropak Q column and an electron capture detector for N_2O or a Poropak N column with a flame ionization detector for CH_4. Flux rates were calculated from the linear or logarithmic increase (N_2O) or decrease (CH_4) in gas concentrations in the soil cover. Samples were collected approximately twice per month during the growing season and monthly during the winter with additional samples preceding and following fertilization and irrigation events known to alter gas fluxes. Flux rates were calculated as the average of six chamber measurements per site taken between the hours of 9:00 and 13:00. Diel measurements suggested that this time was representative of the average flux value for the day in our research area. It was impractical to sample all 12 sites on a single morning. Values presented here as a mean flux rate on a specific day were actually collected on three different days within five days of each other. On each of the three days, one site from each land-use type was sampled. Net gas fluxes from soils to the atmosphere are reported as positive values (N_2O) while a

[6] ⟨http://www.ext.colostate.edu/pubs/garden/07202.pdf⟩

[7] ⟨http://nadp.sws.uiuc.edu/⟩

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net flux from the atmosphere to soil is negative (CH_4). To calculate annual fluxes, we used linear interpolation between measurement dates unless a measurement preceded or followed a climate or management event known to have a large effect on gas fluxes (e.g., snowmelt, fertilization, irrigation), in which case the interpolation was truncated based on the timing of the event. Annual fluxes were statistically analyzed using a one-way ANOVA with land use as the main effect (n = 3 sites per land-use type).

Scaling with remote sensing and GIS

We used the National Land Cover Characterization (Vogelmann et al. 2001) derived from early to mid-1990s Landsat Thematic Mapper satellite data to determine the regional coverage of the land-use types that we sampled. We considered only the portion of Larimer County east of 105.18°W (total area = 1578 km^2) to exclude montane forests. We assumed that data from our corn sites were representative of the mapped land-use class of ''row crops'' (in this region irrigated row crops other than corn are not common) and that data from our grassland sites were representative of the ''grasslands/herbaceous'' class. We combined the ''small grains,'' and ''fallow'' classes to calculate wheat–fallow coverage. We assumed that managed lawns are represented by the class ''urban/recreational grasses'' and by the proportion of the ''low-intensity residential'', ''high-intensity residential'' and ''commercial/industrial/transportation'' classes not occupied by impervious surfaces.

We estimated the proportion of impervious surfaces within the residential and commercial urban classes using a geographical information system (GIS) provided by the City of Fort Collins. This GIS was developed from a high-resolution aerial photograph taken in 1999, and includes polygons for residential and commercial buildings, parking lots, paved roads, water, and paved alleys. We assumed that the area not covered by these polygons were managed lawns. The calculation was made on a portion of the city that was residential in a 1938 aerial photograph because we drew our field sites from this photograph, and because in other parts of the city land not covered by the GIS may also contain grasslands or bare soil (which would not be classified as urban in the landcover map). We used the largest contiguous polygon of areas that represent ''low-intensity residential'' and ''commercial/industrial/transportation'' classes and derived the proportion of lawns from these polygons. The ''high-intensity residential'' category occupied just 0.5% of the urban land in this region and we assumed that the ''low-intensity residential'' and ''high-intensity residential'' had the same proportion of lawns. We assumed that trace-gas fluxes from impervious surfaces were negligible.

RESULTS AND DISCUSSION

Hourly and annual fluxes at the plot scale

Nitrous oxide is produced by soil bacteria that use inorganic nitrogen (N) as an energy source in aerobic soils or an electron acceptor in anaerobic soils, thus the availability of water and soil inorganic N exert strong control over soil N_2O fluxes (Hutchinson and Mosier 1979, Firestone and Davidson 1989, Bouwman et al. 2002, Dobbie and Smith 2003). Native grassland and wheat ecosystems had mean N_2O fluxes <4 μg $N \cdot m^{-2} \cdot h^{-1}$ at every measurement date except for two winter measurements of 15 and 19 μg $N \cdot m^{-2} \cdot h^{-1}$ (Fig. 1C). High winter N_2O fluxes are concurrent with snowmelt and freeze–thaw events (Fig. 1A) in these grasslands (Mosier et al. 1997). In corn ecosystems, peak fluxes of 190 and 350 μg $N \cdot m^{-2} \cdot h^{-1}$ coincided with spring fertilization (mid-May) and the onset of irrigation (late June), but outside of the corn-growing season mean fluxes were typically <10 μg $N \cdot m^{-2} \cdot h^{-1}$. In urban ecosystems, short-lived spikes in N_2O were observed immediately following spring soil thawing (Fig. 1A) and fertilization (Fig. 1C), but high (>10 μg $N \cdot m^{-2} \cdot h^{-1}$) flux rates occurred throughout the year (Fig. 1C, inset).

Aerobic soils are a globally important sink for atmospheric CH_4 (Houghton et al. 2001) because certain soil bacteria use CH_4 as an energy source, a process inhibited by high soil inorganic-N concentrations, cultivation, and excessively high or low soil moisture (Steudler et al. 1989, Mosier et al. 1991, 1997, Robertson et al. 2000, Smith et al. 2003). Native grasslands had the highest CH_4 uptake (Fig. 1D); means ranged from -25 to -45 μg $C \cdot m^{-2} \cdot h^{-1}$ for most of the year, but the most rapid uptake (-55 μg $C \cdot m^{-2} \cdot h^{-1}$) coincided with the onset of soil drying in June (Fig. 1B). Methane uptake by wheat soils was about half that of the native grasslands, though seasonal patterns in uptake were similar among ecosystems. Corn and urban ecosystems had the lowest mean CH_4 uptake rates, typically <-20 μg $C \cdot m^{-2} \cdot h^{-1}$, but uptake was distinctly lower in corn ecosystems after the onset of irrigation (Fig. 1D).

The mean annual flux of N_2O from urban soils to the atmosphere was 0.24 ± 0.03 g $N \cdot m^{-2} \cdot yr^{-1}$, which is comparable to irrigated corn, but more than an order of magnitude larger than wheat or native grasslands (Fig. 2A). For urban and corn ecosystems, annual N_2O fluxes were 2.2 ± 0.1% and 1.4 ± 0.2% of the added N fertilizer. The mean annual flux of CH_4 from urban soils to the atmosphere was -0.15 ± 0.02 g $C \cdot m^{-2} \cdot yr^{-1}$, which is similar to other managed ecosystems, but half of the flux in native grasslands (Fig. 2B). To account for differences in atmospheric life-span, we used Intergovernmental Panel on Climate Change conversion factors (296 for N_2O, and 23 for CH_4; Houghton et al. 2001) to calculate 100-year global warming potential in grams of CO_2 equivalents per square meter per year (Fig. 2, right y-axes). Values ranged from 10 to 110 for N_2O and from -3 to -9 for CH_4, suggesting that effects of urbanization on N_2O fluxes has a much greater impact on atmospheric radiative forcing than the effect of urbanization on CH_4 fluxes does. Robertson

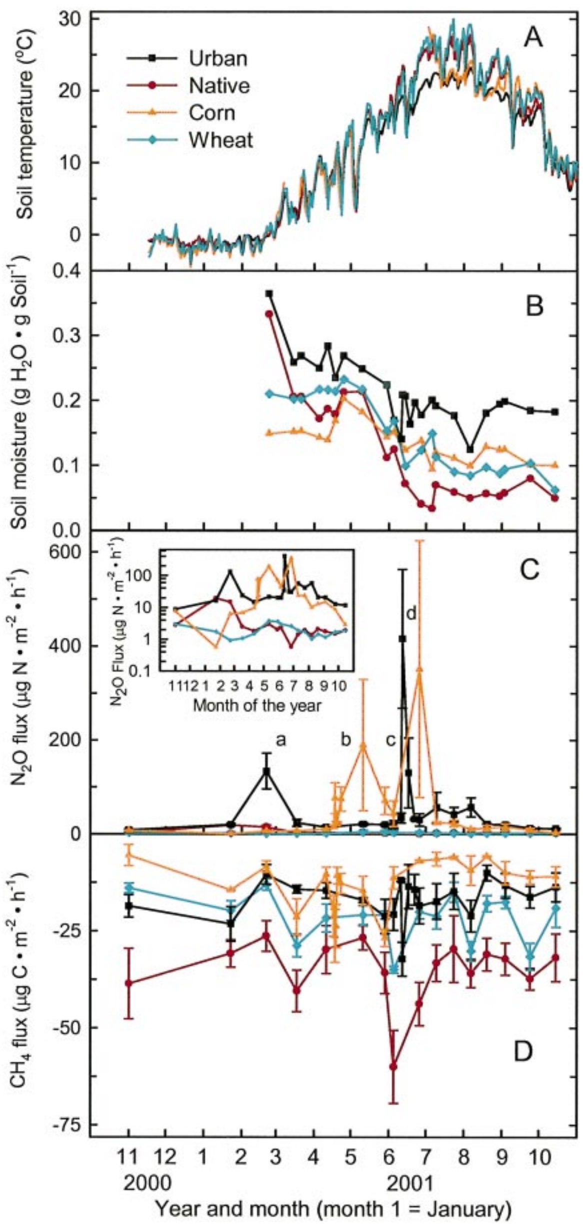

FIG. 1. Soil microclimate and greenhouse-gas fluxes in four land-use types. (A) Mean daily temperature from 1–3 sites per land-use type. Gaps in corn and wheat temperature records occurred when dataloggers were removed for agricultural management. (B) Mean gravimetric soil moisture in 2–3 sites per land-use type. (C and D) N_2O and CH_4 fluxes from soils to the atmosphere; data are means $\pm$ 1 SE, $n = 3$ sites per land-use type. Lowercase letters in (C) denote the time of various events: a, spring soil thaw; b, spring corn fertilization and initiation of urban irrigation; c, spring urban fertilization; and d, initiation of corn irrigation. Data from (C) are also plotted on a log scale (inset).

et al. (2000) came to a similar conclusion in agroecosystems; increased global warming potential from N_2O emission was larger than decreased global warming potential from CH_4 consumption in several fertilized cropping systems.

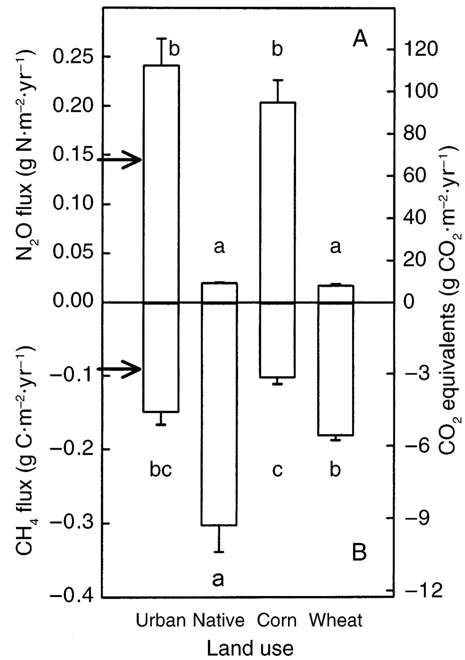

FIG. 2. Annual fluxes of (A) N_2O and (B) CH_4 from soils to the atmosphere in four land-use types. Data are means and 1 SE, $n = 3$ sites per land-use type. Left-hand axes show mass fluxes, and right-hand axes show 100-year global warming potential from each flux. Among land-use types, bars with the same lowercase letter are not significantly different at $P < 0.05$ (ANOVA). The main urban bar is flux per square meter of managed lawn, and horizontal arrows are urban flux per square meter of total urban land area (lawn plus water-impervious surfaces).

Our measured hourly (Fig. 1) and annual (Fig. 2) estimates of soil–atmosphere greenhouse-gas fluxes are comparable to previous measurements in native grassland, corn, and wheat ecosystems in North America (Hutchinson and Mosier 1979, Mosier et al. 1991, 1997, Robertson et al. 2000), suggesting that the duration and frequency of our sampling regime were adequate to quantify differences among land-use types. Our urban lawn data are novel, but long-term measurements in Scotland (Dobbie et al. 1996, Dobbie and Smith 2003) from fertilized (13 to 42 g $N \cdot m^{-2} \cdot yr^{-1}$), uncultivated grasslands with precipitation-water inputs comparable to precipitation plus irrigation in our urban sites provide an interesting comparison. These grasslands had N_2O emissions (Dobbie and Smith 2003) from <5 to 500 μg $N \cdot m^{-2} \cdot h^{-1}$ and methane uptake (Smith et al. 2003) from -3.5 to -20 μg $C \cdot m^{-2} \cdot h^{-1}$. Annual N_2O fluxes were 0.2–2.8 g $N \cdot m^{-2} \cdot yr^{-1}$ (0.4–6.5% of the added fertilizer N), and maximum N_2O losses occurred within weeks of fertilizer additions. Fertilized Scottish grasslands had CH_4 uptake rates that were one third that of adjacent unfertilized woodlands (Dobbie et al. 1996). Thus, the general magnitude of CH_4 and N_2O fluxes, the effect of land-use change on CH_4, and the timing of and fraction fertilizer lost as

N_2O in these intensively managed grasslands are all comparable to the urban lawns that we studied (Figs. 1 and 2). Dobbie and Smith (2003) attribute variability in the N_2O fluxes to three factors: water-filled pore space, temperature, and soil NO_3^--N concentrations. Our cross-ecosystem comparisons are consistent with these mechanistic controls on N_2O fluxes; irrigated and fertilized land-use types had the greatest N_2O losses. The only experimental tests of these factors within urban ecosystem types are short-term studies in experimental turf-grass facilities. Horgan et al. (2002) measured N_2O fluxes for 1–2 months following turf-grass fertilization in Illinois (USA) and 0.4–3.9% of the added fertilizer (4.9 g N/m^2) was emitted as N_2O. Maximum N_2O fluxes 100 times greater than ours were attributed to a large rainfall event that followed fertilization (Horgan et al. 2002). Mancino et al. (1988) reported denitrification fluxes (N_2O plus N_2) up to 10 times greater than ours, but very high fluxes only occurred when water-filled pore space exceeded 75%.

Our data demonstrate that the effect of urban lawns on soil gas fluxes can be as large as the effect of intensive agriculture. However, a more complete picture of land-use change–greenhouse-gas interactions could be gained by sampling multiple years and urban areas of multiple ages along gradients in fertilizer and irrigation management. In addition, for uncultivated ecosystems (urban and native) more frequent winter sampling and experimental studies are required to understand and quantify N_2O pulses during snowmelt and freeze–thaw events.

The importance of urban ecosystems at regional scales

Urban lawns are typically excluded from regional biogeochemical analyses based on the assumption that urban land area is too small to contribute important biological fluxes. We calculated an upper bound for the contribution of urban lawns to regional soil gas exchange using a satellite-derived land-cover map (Vogelmann et al. 2001) to estimate the aerial extent of land-use types in the grassland region of Larimer County, Colorado USA (Fig. 3), and a GIS of pavement and buildings to partition urban land-cover classes into lawns vs. water-impervious surfaces (Fig. 4). The land-use types that we sampled occupied 77% of the region (1578 km^2), and grasslands (50.2% of the region) were the most abundant land-cover class (Fig. 3, Table 1). Irrigated corn and wheat–fallow occupied 12.6% and 7.6% of the region, respectively. The major land-cover types for which we lack trace-gas data were water (5% of the region) and ''pasture/hay'' (12% of the region). Urban land-cover (lawns plus impervious surfaces) classes occupied 10.2% of the region. In urban residential areas, managed lawns occupied 63% of the area and the remaining 27% was covered by surfaces impervious to water (Fig. 4). In the urban commercial areas, the area covered with managed lawns and impervious surfaces was 28% and 72%, respectively.

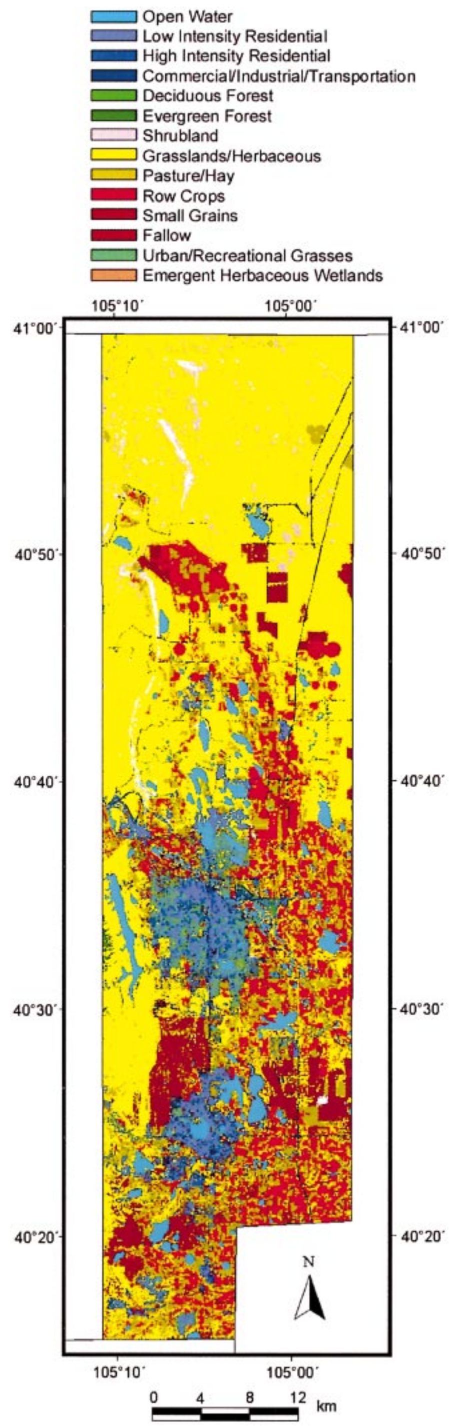

FIG. 3. Land-cover map of eastern Larimer County, Colorado, USA, derived from the National Land Cover Characterization (Vogelmann et al. 2001).

Combining the remotely sensed land-cover data and GIS estimates of the fraction of urban area occupied by lawns, we calculate that urban lawns occupying

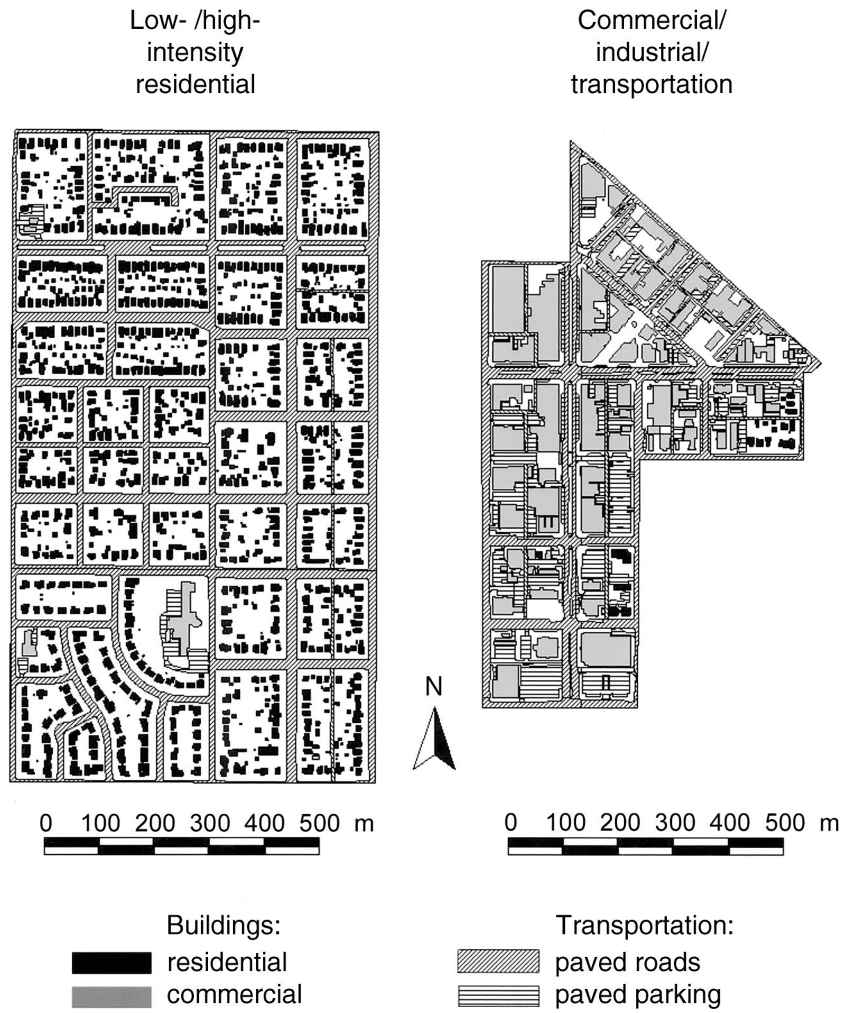

FIG. 4. Land-cover map of two areas within Fort Collins, Colorado, USA, derived from a geographical information system developed by the City of Fort Collins. We assumed that the area not covered by impervious surfaces was managed lawn (white in both maps).

TABLE 1. The contribution of different land-use types to regional soil–atmosphere N_2O emission and CH_4 consumption.

Land use	Land area (% of region)	Flux from sampled area (%)	
		N_2O emission	CH_4 consumption
Urban impervious	3.8	0	0
Urban lawns	6.4	29.6	5.1
Corn	12.6	49.0	6.9
Wheat–fallow	7.6	2.5	7.3
Shortgrass steppe	50.2	18.9	80.7
Total sampled area	76.8	100	100

Notes: The ''region'' refers to the entire eastern portion of Larimer County, Colorado, USA, and ''sampled area'' refers to the area within that region that is occupied by the land-use types that we sampled for greenhouse-gas fluxes. Fluxes from impervious urban surfaces were not measured; they were assumed to be zero.

6.4% of the region accounted for up to 30% of the soil N_2O emission and 5% of the CH_4 consumption from land-use types that we sampled (Table 1). We interpret these values as an upper bound for the contribution of urban lawns to regional gas fluxes for three reasons. First, we assumed that land not occupied by pavement and buildings is lawns, causing an overestimation of lawn area in some cases (e.g., there are also gravel driveways, vacant lots). Second, poorly managed lawns would have smaller irrigation and fertilizer inputs than our plots. Third, newer housing developments may have less lawn per unit area than the older neighborhood that we sampled. In contrast, our calculated total urban land area is an underestimate because of the rapid urban expansion between the time of the land-cover classification (early 1990s) and our analysis. Despite these uncertainties, our data clearly show that regional gas fluxes were correlated with management regimes rather than the total land area covered. Fertilized and irrigated corn and urban systems dominated regional N_2O emissions despite covering a small portion of the land area (Table 1). Native grasslands occupied about half of the region but accounted for 81% of regional CH_4 uptake from land-use types that we sampled.

One of the most important challenges in urban ecology is identifying links between social and ecological processes (Collins et al. 2000, Grimm et al. 2000) and our results support three hypothetical links between human decisions and biogeochemistry at multiple scales. At the regional scale, our data suggest that biogeochemical fluxes depend on which land-cover types are converted to urban use. The conversion of grasslands to urban use (increasing N_2O, decreasing CH_4) will have a much different effect than conversion of corn to urban use (moderate decrease in N_2O, no change in CH_4). At the landscape or neighborhood scale, we hypothesize that effects of urbanization depend on landscape architecture, especially the proportion of the landscape that is lawns vs. impervious surfaces. At the household scale, we hypothesize that individual management decisions (irrigation and fertilization levels) are important determinants of biogeochemical fluxes. Long-term research across gradients in landscape architecture and lawn-management intensity in multiple cities would enable tests of these hypotheses.

Acknowledgments

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