Urban ecosystems and the North American carbon cycle

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Abstract

**Approximately 75–80% of the population of North America currently lives in urban areas
as defined by national census bureaus, and urbanization is continuing to increase. Future
trajectories of fossil fuel emissions are associated with a high degree of uncertainty;
however, if the activities of urban residents and the rate of urban land conversion can be
captured in urban systems models, plausible emissions scenarios from major cities may
be generated. Integrated land use and transportation models that simulate energy use
and traffic-related emissions are already in place in many North American cities. To these
can be added a growing dataset of carbon gains and losses in vegetation and soils
following urbanization, and a number of methods of validating urban carbon balance
modeling, including top down atmospheric monitoring and urban 'metabolic' studies of
whole ecosystem mass and energy flow. Here, we review the state of our understanding
of urban areas as whole ecosystems with regard to carbon balance, including both drivers
of fossil fuel emissions and carbon cycling in urban plants and soils. Interdisciplinary,
whole-ecosystem studies of the socioeconomic and biophysical factors that influence
urban carbon cycles in a range of cities may greatly contribute to improving scenarios of
future carbon balance at both continental and global scales.**

Keywords: CO_2 emissions , global carbon cycle, urban ecology

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Introduction

The carbon balance of North America has become the
subject of a US interagency research initiative aimed at
quantifying sources and sinks of carbon and the me-
chanisms underlying continental-scale carbon balance
(Wofsy & Harris, 2002). In North America, as at the
global scale, urbanization is a major component of
environmental change and alterations to the carbon
cycle. There has been increasing interest in the study
of urban systems as ecosystems in the sense that both
biophysical and socioeconomic components, as well as
their interactions, can be studied at various scales to

understand the system as a whole (Alberti & Waddel,
2000; Grimm et al., 2000; Pickett et al., 2001). Integrated
urban ecological studies can contribute greatly to un-
derstanding the carbon cycle, as they incorporate un-
derlying drivers of fossil fuel emissions, as well as
biological sources and sinks of carbon.

North America as a whole is a large contributor to
global-scale fossil fuel emissions and much of this
contribution is attributable to urban areas. Approxi-
mately 75–80% of the population of the continent is
urban as defined by the census bureaus of the United
States, Canada and Mexico; this proportion is projected
to continue to increase (Fig. 1). CO_2 emissions from
fossil fuel combustion for electricity generation, none-
lectrical residential energy, and the transportation

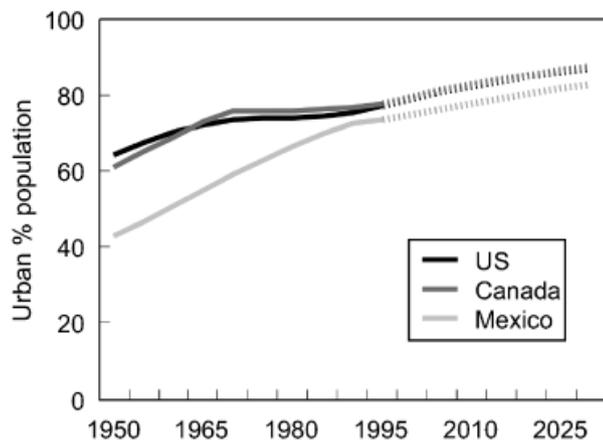Correspondence: Diane Pataki, e-mail: dpataki@uci.edu

Fig. 1 Actual and projected percentage of the population of the United States, Canada and Mexico that live in urban areas, estimated by the United Nations (2004b).

sector are highly dependent on the magnitude and distribution of urban areas, which vary greatly between the United States, Canada, and Mexico, as well as at regional scales. Urban areas also affect the carbon balance of terrestrial ecosystems directly through land conversion (e.g. Alig et al., 2002; Alig & Butler, 2004), and indirectly through effects of the urban 'footprint' on ecosystem processes.

Here, we review recent research on the linkages between urbanization and the current and future carbon cycle. We suggest that the trajectory of the North American carbon cycle will be highly influenced by the patterns and forms of urban development, and that research linking urban planning, engineering, decision-making, and natural ecosystem processes in and surrounding urban areas will greatly improve projections of continental scale carbon budgets in the 21st century.

Fossil fuel emissions from urban areas

The direct effect of fossil fuel emissions is arguably the most significant effect of urbanization on the carbon cycle. In the 1990s, fossil fuel combustion resulted in emission of 6.3×10^3 Mt yr^{-1} of carbon globally, of which 3.2×10^3 Mt yr^{-1} was retained in the atmosphere (IPCC, 2001). North America contributed 1.6×10^3 Mt yr^{-1} of these emissions, of which more than 85% was contributed by the United States (Marland et al., 2005). Approximately 40% of total fossil fuel emissions in the United States, Canada, and Mexico originate from the transportation and residential sectors (World Resources Institute, 2005); of this, the vast majority is contributed by urban residents, given that they constitute most of the population of all three nations (Fig. 1).

Many aspects of urban form, structure, and growth may influence the magnitude of fossil fuel emissions and their rate of change. Of these, population and housing densities are key variables in addition to the rate of total population growth, affluence and technologies in use (Kates et al., 2003; Romero Lankao et al., 2004). North America is characterized by increases in the number of households that exceed the rate of population growth; according to United Nations estimates, between 1985 and 2000 the number of households in the United States increased by 25% while the population increased by only 15%. In Canada, households increased by 39% for a population increase of 19%, and in Mexico the increases for households and the total population were 60% and 33%, respectively (United Nations, 2002; United Nations Habitat, 2003). Reductions in household size result in increases in the number of dwellings and vehicles per capita (MacKellar et al., 1995; CEC, 2001), as well as increases in the consumption of land (Alig et al., 2003; Liu et al., 2003). MacKellar et al. (1995) estimated that if households rather than total population is analyzed in relation to global energy use, more than 40% of the increase in energy consumption of developed countries from 1970 to 1990 can be attributed to reductions in household size, due both to changes in age structure (i.e. aging populations), as well as other behavioral and societal changes. Hence, CO_2 emissions projections are very sensitive to demographic assumptions and units of scale (e.g. whether individuals or households are treated explicitly).

Building characteristics are also a key determinant of fossil fuel emissions from the residential sector. For example, the average size of newly constructed, single family residences has been increasing rapidly in the United States, from 140 m^2 in 1970 to more than 210 m^2 in 2004 (NAHB, 2005). At the same time, energy use per household has evolved with technological change; overall, there was a decrease in energy 'intensity' in the last three decades as measured by residential energy use per m^2. Per household, residential energy use declined between 1978 and 1987, and has remained constant due to opposing effects of decreased energy use for heating and cooling and increased appliance use (US Department of Energy, 2004). Multiple factors influencing CO_2 emissions from the residential sector can be modeled with residential energy end-use models that take into account characteristics of the housing stock (Aydinalp et al., 2003). One such model, the Canadian Residential End-use Energy Consumption and Emission Model (CREEEM), has been used to evaluate the potential for national CO_2 emissions reductions from energy efficiency measures in Canada, which is characterized by high emissions from residen-

tial heating due to its northerly location. Simulations with CREEEM indicated that upgrades to housing insulation and mechanical and appliance efficiency, individually, could result in emissions reductions of up to 9% of emissions from the existing Canadian housing stock. Multiple upgrades may additionally increase emissions reductions, although the effects are not additive (i.e. some multiple upgrades have cancel- ing effects). In addition, some of the major upgrades that constituted this reduction may be prohibitively expensive for many consumers (Guler, 2000). Hence, trends in household size and number, residence size, and construction of new housing in the coming decades may have a larger potential for impacts on residential CO_2 emissions than retrofits to existing housing.

Spatial patterns of housing and development also play a key role. 'Urban sprawl' has been linked qualita- tively to increasing fossil fuel emissions per capita in North America (CEC, 2001; Gonzalez, 2005). The term urban sprawl has been variously defined, but generally refers to one or more combinations of reductions in developmental density, segregation of residential and commercial districts, and expansion of the transporta- tion network (Ewing et al., 2003). Notably, the term 'urban' itself imparts some uncertainty to the study of land conversion as definitions of urban land vary; Theobald (2005) pointed out that the widely used US census definition excludes conversion to lower density, more rural development ('exurban'), which is also rapidly increasing in the United States (Stein et al., 2005). Such reductions in density and increases in the spatial extent of settlements and road networks result in increasing commuting distances, vehicle miles traveled, and related vehicular emissions (Ewing et al., 2003).

Highly quantitative linkages between urban develop- ment and fossil fuel emissions are a key next step. In many metropolitan and developing regions, powerful tools for analyzing patterns of land conversion and its impacts on energy use and traffic are already in place. A variety of land-use models that explicitly represent household choices and characteristics, geographic/en- vironmental factors, and transportation patterns asso- ciated with urban land expansion have been developed as land use and transportation planning tools (EPA, 2000; Hunt et al., 2005). These models may use a variety of methods including statistical, econometric, dynamic, cellular automata, or hybrid approaches to represent land-use change and its linkages to decision-making and biophysical processes (Agarwal et al., 2000). In- creasingly, urban land-use transportation modeling fra- meworks have been linked to pollutant emissions, environmental, and ecological models to evaluate im- pacts of land use alternatives on air quality, biodiversity, and ecological processes (Alberti & Waddel, 2000; EPA,
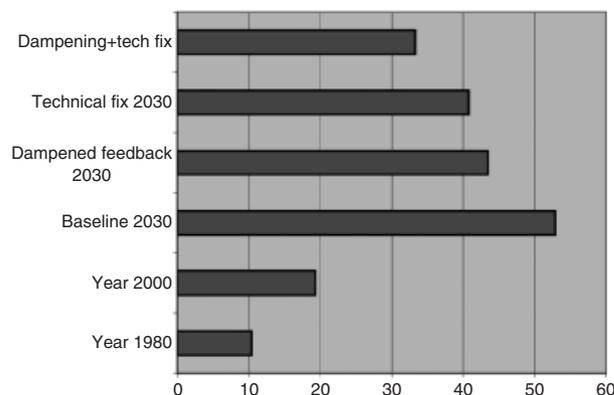

Fig. 2 Scenarios of CO_2 emissions (in Mt) for the Salt Lake– Ogden metropolitan area as estimated from a systems dynamics model of urban land conversion. The 'dampened feedback scenario' involves dampening of the causes of reduced develop- mental densities associated with urban sprawl. The 'technical fix scenario' represents improvements in fuel combustion efficien- cies and increasing use of alternative fuels. Further details are given in Emmi et al. (2005).

2000). There is great potential to utilize these frame- works to develop plausible scenarios of CO_2 emissions from electricity generation, traffic, and residential fuel combustion. For example, a systems dynamics model of urban growth in the Salt Lake City–Ogden metropolitan area in Utah, USA compared CO_2 emissions scenarios under various policy options involving technological innovation vs. dampening of urban sprawl (Fig. 2). An advantage to utilizing land-use-transportation models for emissions scenarios is their ability to incorporate multiple factors that influence decision-making about land use, residential energy use and transportation patterns, including environmental, economic, and demographic variables.

The role of vegetation and soils

Lacking in most, if not all, urban land-use models is explicit incorporation of the effects of urban land ex- pansion on CO_2 sources and sinks in vegetation and soil. Using satellite data and the CASA model, Imhoff et al. (2004) estimated that for the United States, the net effect of urbanization on net primary productivity (NPP) was negative in the mid-1990s due to losses of productive agricultural land and constituted a reduc- tion in C fixed by photosynthesis of $40 \, Mt \, C \, yr^{-1}$ – equivalent to annual fossil fuel emissions of a large North American city (e.g. Fig. 2). However, once land conversion has taken place, accretion of organic carbon may occur in constructed urban, suburban, and exur- ban ecosystems that contain managed vegetation and

soil. Of urban organic carbon pools, trees are probably the best studied, particularly in the United States. Estimates of carbon storage in urban trees in the coterminous United States are on the order of 700 Mt (335–980 Mt) with a gross sequestration rate of 22.8 Mt C yr^{-1} (13.7–25.9 Mt C yr^{-1}) (Nowak & Crane, 2002). Rates of net annual sequestration vary regionally, and have been estimated to range from 0.26×10^{-9} Mt C m^{-2} average forest cover in Atlanta to 0.12×10^{-9} Mt C m^{-2} cover in New York, with a median value of 0.2×10^{-9} Mt C m^{-2} cover (Nowak & Crane, 2002). Analogous calculations are not currently available for Canada and Mexico. In addition, uncertainties in these analyses include a lack of direct measurements of urban tree allometry and biomass, particularly belowground.

When considering total urban carbon balance, including fossil fuel emissions, urban forests can also affect carbon balance indirectly through their effects on urban energy balance and subsequently, CO_2 emissions related to energy use. Depending on their location relative to buildings, urban trees can reduce incident radiation through shading effects, altered wind patterns, and increased evaporative cooling (Oke, 1989; Taha, 1997; Akbari, 2002). Although some of these effects may increase energy use, for example, increased heating due to winter shading or 'windbreak' effects of trees near buildings that may increase air conditioning use, and increased net radiation due to lower albedo, most studies have estimated a net decrease in energy use and subsequent CO_2 emissions from electricity generation resulting from placing trees near buildings (Huang *et al.*, 1987; Akbari & Taha, 1992; Akbari *et al.*, 1997; Akbari, 2002; Akbari & Konopacki, 2005). However, many of these calculations are based on model simulations that include untested assumptions about urban vegetation and surface processes. Some field measurements support reduced summertime temperatures near urban forest canopies (Taha *et al.*, 1991; Spronken-Smith *et al.*, 2000; Mueller & Day, 2005), but additional direct measurements are needed to quantify more complex

effects, such as increases in both latent and sensible heat fluxes due to low forest albedo (Grimmond *et al.*, 1996).

Estimates by Nowak *et al.* (2002) suggested that if fossil fuels are used to maintain urban vegetation, net effects will eventually become negative (net emitters of carbon) at varying rates depending upon species, disposal techniques and maintenance intensity, unless trees are planted in energy conserving locations to offset maintenance emissions. Using currently available estimates of forest effects on energy use, McPherson *et al.* (2005c) estimated the carbon balance of several municipal urban forests (street and park trees) including direct C sequestration, indirect effects of energy savings, and fossil fuel emissions from forest maintenance and management. Carbon costs and benefits varied regionally: in Cheyenne, WY and Glendale, AZ avoided CO_2 emissions due to energy savings slightly exceeded direct carbon sequestration from growth, while in the remaining cities, energy savings constituted 37–48% of net CO_2 emissions reductions (Table 1). In all cases, the presence of urban trees resulted in a net reduction in CO_2 emissions (Table 1). Notably, these calculations pertain to growth and indirect emissions associated with the vegetative component of urban forests only; the impact of soil C pools in urban forests is more uncertain.

In general, soil organic matter (SOM) and soil organic carbon (SOC) pools appear to increase following an initial loss of carbon from tillage and disturbance in conversion to various types of urban land. The greatest increases have been observed in the most highly managed soils – for example SOM in golf course fairways was 1.76% 1 year after turfgrass planting, 3.8% after 20 years, and 4.2% after 31 years (Qian & Follet, 2002). In the Denver–Boulder metropolitan area, the SOC pool to a depth of 30 cm was 2.1×10^{-9} Mt m^{-2} in the first decade after construction – an initial decrease below native grassland pools; but within two decades of development, the urban SOC pool rose to 3.2×10^{-9} Mt m^{-2} and surpassed that stored in grasslands (Golubiewski, 2006, Fig. 3). Likewise, in a comparison of

Table 1 Average annual carbon dioxide reductions (avoided and sequestered) and releases (maintenance and decomposition) in kg per tree for street and park tree populations in seven US cities

	Ft Collins	Cheyenne	Bismarck	Berkeley	Glendale	Minneapolis	Boulder
Tree numbers	30 943	17 010	17 821	36 485	21 481	198 633	35 802
Avoided	33.05	59.73	44.07	36.23	19.52	126.10	51.12
Sequestered	56.67	55.05	60.85	54.26	16.06	134.85	64.23
Maintenance	3.43	2.52	4.27	0.35	0.18	0.37	0.93
Decomposition	7.23	8.66	8.10	7.94	1.51	8.82	7.30
Net reduction	79.07	103.59	92.54	82.22	33.90	251.76	107.12

Avoided emissions refer to emissions reductions from energy savings associated with tree cover. Maintenance refers to emissions associated with urban forest management activities. Data from McPherson *et al.* (2005a–c).
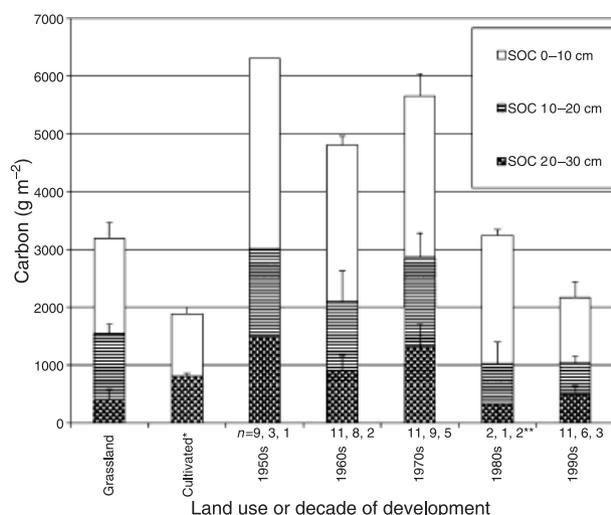
Fig. 3 Soil organic carbon (SOC) in different land cover types. Grassland and cultivated categories show literature values, while SOC from a study of urban green spaces in the Denver metropolitan region are shown by decade of development. Sample sizes for woody vegetation C, lawn stubble C, and soil measurements are shown for each category. From Golubiewski (2006).

carbon storage in urban and native soils in the region of six US cities, Pouyat *et al.* (2006) found that urban areas have the potential to either sequester or lose SOC. For those cities in the Northeast with high concentrations of C in native soils (Boston and Syracuse), there was a 1.6-fold decrease in SOC pools following urbanization. By contrast, cities located in warmer and/or drier climates, such as Chicago and Oakland, showed slight increases (6% and 4%, respectively) in SOC pools. In another study, they found that variation in SOC density was higher within a city than among cities for a given soil type (Pouyat *et al.*, 2002). In New York City, for example, the highest SOC density (1-m depth) occurred on a golf course ($28.5 \times 10^{-9} \, \mathrm{Mt \, m^{-2}}$), while the lowest density occurred in an old dredge site ($2.9 \times 10^{-9} \, \mathrm{Mt \, m^{-2}}$), nearly a 10-fold difference. For soils derived from clean fill, however, SOC density varied by only $3 \times 10^{-9} \, \mathrm{Mt \, C \, m^{-2}}$ across several cities. Similarly, residential yards in Chicago, Moscow, and Baltimore exhibited relatively little variation in soil C density ($14.4 \pm 1.2 \, 10^{-9} \, \mathrm{Mt \, m^{-2}}$).

These measurements have recently been used to estimate total carbon storage of urban soils in the coterminous United States: $1.9 \times 10^3 \, \mathrm{Mt}$ with an average SOC density of $7.7 \pm 0.2 \times 10^{-9} \, \mathrm{Mt \, m^{-2}}$ (Pouyat *et al.*, 2006). Notably, estimates of urban soil C storage are highly sensitive to whether total urban land area includes impervious surfaces, or whether soil C is expressed on a pervious surface basis. When considering pervious surfaces only (i.e. excluding soil carbon stored beneath paved surfaces), soil C densities may be higher than the native ecosystem they replaced. Although managed urban soils may contain more SOM and SOC than preurban, native soils, total soil greenhouse gas emissions including other trace gas fluxes and energy inputs from soil management (e.g. irrigation and fertilizer production and transport) may result in a net loss of carbon. Relative to native soils, urban soils have been shown to have higher fluxes of both CO_2 and N_2O (Koerner & Klopatek, 2002; Kaye *et al.*, 2004, 2005) – the latter being a far more potent greenhouse gas (IPCC, 2001). Full carbon accounting of urban soil C cycling including fossil fuel emissions from management remains to be conducted, although these analyses have been conducted for agricultural ecosystems. Mosier *et al.* (2005) and Robertson *et al.* (2000) assessed total net greenhouse gas emissions in several cropping systems managed with rainfed, irrigated, till, or no-till practices in Colorado and Michigan. Their calculations included net CO_2, N_2O, and CH_4 emissions associated with fertilizer production, irrigation, tillage, plowing, and other farm operations. Most conventional till cropping systems were associated with positive net radiative forcing potential (net release of greenhouse gases to the atmosphere), while negative forcing potentials (net removal of greenhouse gases) were achieved in irrigated, no-till, continuous cropping systems. These systems have similarities to management practices in some highly managed urban soils such as irrigated turfgrass. Key determinants of net emissions in the cropland studies were rates of SOM formation and the magnitude of emissions of N_2O, which suggests that additional measurements of these variables in a variety of urban systems would provide key constraints for estimating net urban radiative forcing potential.

Results to date for urban ecosystems suggest that the net effect of urban land-use conversion will depend partly on the characteristics of the native or rural ecosystem replaced. For arid regions, the net overall effect of urbanization may be higher productivity rates, with the potential to actually increase net C storage; in more humid environments, the net effect may be a reduction in soil C storage (Pouyat *et al.*, 2002; Golubiewski, 2003). It is clear that a great deal of additional data and cross-system comparisons on regional and global scales will be necessary to improve estimates of the net effect of urbanization on biogeochemical cycles.

The current and future carbon balance of North American settlements

Is it possible that biological C sinks in urban vegetation and soils could substantially offset CO_2 emissions from

fossil fuel burning associated with urban activities? The answer is likely no, barring very large changes in the carbon intensity of North American urban regions. However, a number of municipalities in Canada, the United States, and Mexico have committed to voluntary programs of greenhouse gas emissions reductions, many under the Cities for Climate Protection program (CCP) of International Governments for Local Sustainability (ICLEI, formerly the International Council of Local Environmental Initiatives), which currently lists 269 towns, cities, and counties in North America committed to conducting emissions inventories, establishing a target for reductions, and monitoring the results of reductions initiatives (http://www.iclei.org). Because emissions reductions targets tend to be relatively modest, biological sinks may be important from a policy perspective for municipalities that choose to utilize tree planting and related activities to meet reductions targets. But how effective are biological sinks in offsetting CO_2 emissions in the context of total system carbon balance? How do these sinks compare with the impacts of energy efficiency measures or alternative land use and transportation designs in reducing direct emissions from fossil fuel combustion? A number of 'accounting' methods are available for evaluating total urban ecosystem carbon balance, though a comprehensive analysis and comparison of multiple regions remains to be conducted.

Local and regional carbon balance may be estimated from atmospheric measurements using eddy covariance (Grimmond *et al.*, 2002, 2004; Nemitz *et al.*, 2002; Soegaard & Moller-Jensen, 2003) or with measurements of CO_2 concentration combined with atmospheric transport modeling (Lin *et al.*, 2004). Tracers such as carbon monoxide and the isotopic composition of CO_2 may be used to distinguish emissions from fossil fuel combustion vs. natural sources; in addition, it is possible to further distinguish different fuel types such as gasoline vs. natural gas, which have distinct carbon isotope ratios (see Box 1). Atmospheric, 'top-down' approaches to estimating carbon balance have been successfully applied in a number of natural ecosystems, and have been under-utilized in urbanizing environments. Atmospheric monitoring programs in cities for greenhouse gases, as well as more commonly measured pollutants may provide very useful information about patterns and processes of emissions, as these datasets may be applied at various temporal and spatial scales depending on the location and frequency of measurements and the type of transport model. However, there is a limit to the spatial disaggregation that can be applied to top–down estimates; in addition, atmospheric measurements capture only local sources and sinks of greenhouse gases. 'Upstream' emissions such as CO_2 emitted from remote power plants that supply energy to urban areas are not detected.

An additional, complementary approach has been termed the urban 'metabolism' approach, which offers a conceptual framework for understanding urban carbon flows. Named for analogies with metabolic processes of organisms in that 'cities transform raw materials, fuel, and water into the built environment, human biomass and waste' (Decker *et al.*, 2000), these studies quantify the inputs and outputs of energy, water, nutrients, materials, and wastes, including components of the carbon cycle. Applied to evaluating greenhouse gas emissions, these analyses would ideally consider emissions from fuel combustion within a given city, as well as upstream power generation; emissions from industrial processes, waste, and periurban agriculture within the city region; and the impacts of organic carbon pools in vegetation and soils. However, consistent standards for analyzing carbon balances in urban metabolism have not been established. One of the greatest challenges is that no international organization collects data on city level energy use and published data on fuel use at this scale are scarce (Decker *et al.*, 2000); in addition, datasets quantifying the direct and indirect impacts of land conversion on organic C pools have also been lacking. Given these limitations, in North America only the Greater Toronto Area (GTA) has been evaluated 'metabolically' for total greenhouse gas emissions, yielding estimates of 51.5–73.0 Mt CO_2 equivalent in 1990, with a 14% increase by 1999 (Sahely *et al.*, 2003). On a per capita basis (14 Mt CO_2 cap^{-1} yr^{-1}), this is the highest estimate yet reported in comparison with other cities internationally (Table 2), although these results

Table 2 CO_2 emissions per capita in various metropolitan areas as estimated with urban ''metabolic'' approaches (see text for details)

City	Emissions (t CO_2 cap^{-1} yr^{-1})	Source
1970s		
Brussels	5.9	Duvigneaud & Denayeyer-De Smet (1977)
Hong Kong	2.3	Newcombe *et al.* (1978)
Sydney	7.1	Newman (1999)
Tokyo	4.3	Hanya & Ambe (1976)
1990s		
Hong Kong	4.8	Warren-Rhodes & Koenig (2001)
London	5.5	Chartered Institute of Wastes Management (2002)
Sydney	9.1	Newman (1999)
Toronto	14.0	Sahely *et al.* (2003)

Box 1 Tracing CO_2 sources from atmospheric measurements

In many urban ecosystems, CO_2 sources contain unique isotopic signatures that can be used to trace their origin. Fossil fuels contain no radiocarbon by virtue of their age in contrast to respiration of modern carbon from plants and soils (Zondervan & Meijer, 1996; Meijer *et al.*, 1997; Takahashi *et al.*, 2001, 2002). The oxygen isotope composition of CO_2 is also generally distinct for respiration vs. combustion-derived CO_2, depending on the isotopic composition of local water plant available water (Florkowski *et al.*, 1998; Pataki *et al.*, 2003, 2005a). For anthropogenic sources, CO_2 derived from natural gas combustion is more isotopically depleted in stable ^{13}C than CO_2 derived from gasoline combustion (Tans, 1981; Andres *et al.*, 2000; Pataki *et al.*, 2005a). As a result, natural gas combustion, gasoline combustion, and biogenic respiration have distinct combinations of isotope tracers (Fig. 4) that can be used to solve for the proportional contributions of each source to total CO_2 in the atmosphere. For example, for the dual stable isotope tracers of ^{13}C and ^{18}O in CO_2, the mass balance equations can be written as

$$C_B + C_R + C_G + C_N = C_T \qquad (1)$$

$$\delta^{13}C_B C_B + \delta^{13}C_R C_R + \delta^{13}C_G C_G + \delta^{13}C_N C_N = \delta^{13}C_T C_T \qquad (2)$$

$$\delta^{18}O_B C_B + \delta^{18}O_R C_R + \delta^{18}O_G C_G + \delta^{18}O_N C_N = \delta^{13}O_T C_T \qquad (3)$$

where C is the CO_2 concentration, $\delta^{13}C$ is the carbon isotope composition of CO_2, $\delta^{18}O$ is the oxygen isotope composition of CO_2, and the subscripts B, R, G, N, and T refer to the background atmosphere, the biogenic respiration source, the gasoline combustion source, the natural gas combustion source, and the total atmospheric CO_2 concentration, respectively. Pataki *et al.* (2003) used this approach to quantify the contribution of C_R, C_B, and C_N to the total, non-background CO_2 concentrations ($C_T - C_B$) in Salt Lake City, Utah, and found that biogenic respiration contributed up to 60% of local, non-background, CO_2 during the growing season. In contrast, natural gas combustion constituted 30–70% of local CO_2 in the wintertime depending on ambient temperatures and time of day, with colder temperatures resulting in increased natural gas consumption from residential furnaces, and evening rush-hour periods showing a greater contribution from vehicular traffic (Pataki *et al.*, 2003, 2005b, 2006).

cannot be directly compared with other North American cities without additional analyses.

Future pathways of urban C balance depend on the diverse processes that underlie current and future carbon emissions by coupled human-environment systems. These include economic growth, demographic dynamics, technologies, and institutional settings, as well as climate and biophysical processes. The rate of population growth has declined over the last 20 years in North America (from 1.7% in 1950–1975 to 0.9 in the last two decades in the United States and Canada, and from 3.8% in the 1970s to 1.8% in the last decade in Mexico) but in absolute terms the region is still within an era of demographic increase (United Nations, 2004a). Even as the rate of population growth continues to decline, the region's total population will rise substantially, while if current trends continue, developmental densities and household sizes will decline. In addition, rates of growth vary between countries and regions. Categorically, regions and cities in the United States and Canada belong to a different cluster of development than Mexico: the United States and Canadian economies have become less carbon intensive; however, they still concentrate by far the highest share of international trade, production, energy-consumption, and carbon emissions (Romero Lankao, 2004). Their urban regions and cities enjoy relatively higher standards of life and are proto-
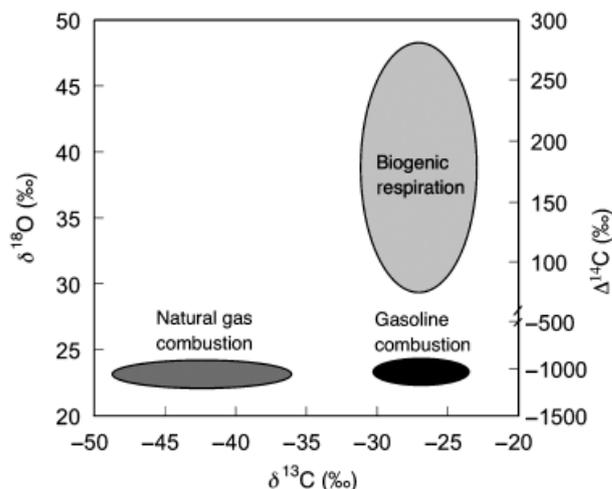

Fig. 4 Common ranges of the stable carbon isotope composition ($\delta^{13}C$ relative to the V-PDB Standard), oxygen isotope composition ($\delta^{18}O$ relative to the V-SMOW Standard) and radiocarbon composition ($\Delta^{14}C$ relative to the oxalic acid standard) of urban CO_2 sources.

types of economic and institutional success (United Nations Habitat, 2003), although their ecological footprints continue to be much larger than in less developed regions (Fischer-Kowalski & Amann, 2001). Romero Lankao *et al.* (2004) analyzed greenhouse gas emissions

of Mexico City and placed the results in a national context. In contrast to the United States and Canada, Mexico still depends heavily on primary-commodities export as source of international trade. Romero Lankao *et al.* (2004) suggested that during the last two decades, the Mexican states promoted economic liberalization – influenced by international organizations, their ability to support domestic industries and invest in economic growth and infrastructure has been weakened. The result has been difficulty in financing social and urban infrastructural expenditure. Therefore, Mexico has a relatively smaller share of carbon emissions than its North American counterparts. Pressing issues remain including social segregation, financial constraints, and local environmental problems such as air quality (Romero Lankao *et al.*, 2004; Wade, 2005); responses to these social and environmental issues may affect the future trajectory of national greenhouse gas emissions. In short, disparate underlying patterns of development in the United States, Canada, and Mexico must be taken into account in future projections of urban development and its impacts.

Outlook for the future

Future projections of urban growth incorporate multiple factors, such as regional changes in population and income, which influence development and its regional variability. For example, Alig & Healy (1987) provided projections of changes in the US developed area from 1982 to 2000 using a cross-section of nationwide land-use data available at the time. Although the results indicated a significant future increase in developed area, external demographic and macroeconomic projections did not anticipate the above-average growth that occurred during in the 1990s in the United States. This growth was accompanied by an accelerated rate of development – in that decade alone, US census-defined urban areas increased from 2.6% to 3.1% of the land base (Nowak *et al.*, 2005). However, despite the inherent uncertainty always present in generating future projections, there are a growing number of tools available for mechanistically linking population projections to emissions and total carbon balance in population centers. Current projections for urban land development in North America highlight the importance of utilizing these tools and gathering additional data to improve them. For example, Alig *et al.* (2004) projected an increase in the coterminous US developed area over the next 25 years of 79%, raising the proportion of the total land base that is developed from 5.2% to 9.2%.

North America is an urbanized continent and the proportion of urban, suburban, and exurban land is still rapidly increasing. Comparative ecological studies of urban carbon cycles that combine analyses of energy use, fossil fuel emissions, ecological footprints, and plant and soil carbon pools have only begun. A focus on cities as a major influence on the future trajectory of atmospheric CO_2 necessarily begins with a local and regionalized approach using new quantitative tools to analyze mass and energy flow and their underlying causes at the scale of individual metropolitan areas. In order to determine major similarities and differences among urban areas and their emergent properties at increasing temporal and spatial scales, more of these localized studies must be conducted and compared across regions, nations, and the developed and developing world. An excellent model for regional, interdisciplinary urban studies is provided by the NSF Long Term Ecological Research (LTER) research program, which includes two urban sites (Grimm *et al.*, 2000; Pickett *et al.*, 2001). To expand the integration of causes and consequences of urbanization into carbon cycle science, urban, suburban and exurban sites should be incorporated into other national and international research networks such as the CO_2 flux networks (Baldocchi *et al.*, 2001) and the manipulative global change experiments (Mooney *et al.*, 1999; Canadell *et al.*, 2000). Key questions for the next several years include:

- What is the spatial and temporal variability in fossil fuel emissions in urbanizing regions in North America?

- What is the role of direct and indirect effects of biological processes in net carbon emissions and climate in North American cities?

- What are the major factors that will determine the trajectory of anthropogenic and biogenic C sources and sinks in urbanizing regions in the 21st century?

Linked to socioeconomic and coupled human–environment studies, interdisciplinary, ecological studies of urbanizing regions will increasingly contribute to quantifying the causes and effects of urban land conversion for the continent and for the global climate system.

Acknowledgments

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