Chapter 5

Effects of Urban Land-Use Change on Biogeochemical Cycles

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5.1 Introduction

Urban land-use change, the conversion of agricultural and natural ecosystems to human settlements, has become an important component of global change. Virtually all of the projected increase in the world's population is expected to occur in cities so that by the year 2007 more than half of the global population is expected to live in urban areas (United Nations 2004). Yet, urban settlements and surrounding areas are complex ecological systems that have only recently been studied from a rigorous ecological perspective (Pickett et al. 2001). Clearly, urban areas represent ecosystems with modified biogeochemical cycles such that fluxes and pools of matter, energy, and organisms differ greatly from the previous ecosystem (Collins et al. 2000). Our knowledge of the magnitude and extent of these biogeochemical changes at local, regional, and global scales is a major area of uncertainty in our understanding of global change.

In the ecological sciences, the ecosystem approach has been used successfully in studying the effects of disturbance and exogenous inputs of nutrients and pollutants on biogeochemical cycles (Likens 1992). To study biogeochemical cycles in urban areas, Grimm et al. (2003) recommended the ecosystem approach where: (1) urban ecosystems are viewed as spatially homogenous, i.e., "well-mixed reactors," or (2) urban ecosystems are considered a heterogeneous assemblage of parts or "patches." Indeed, cities have been viewed as well-mixed reactors in answering questions about how they affect biogeochemical cycles at regional and global scales (Fig. 5.1). In so doing, mass balances have been constructed for entire cities to assess their "ecological footprint," that is, the amount of land resources acquired from other ecosystems needed to sustain the urban population (Rees 2003). Moreover, the mass balance approach allows investigations into the movement of nutrients and pollutants across ecosystem boundaries and the cycling of these materials within the system itself. For example, Baker et al. (2001) constructed a nitrogen balance for the Phoenix metropolitan area and found that inputs (human and

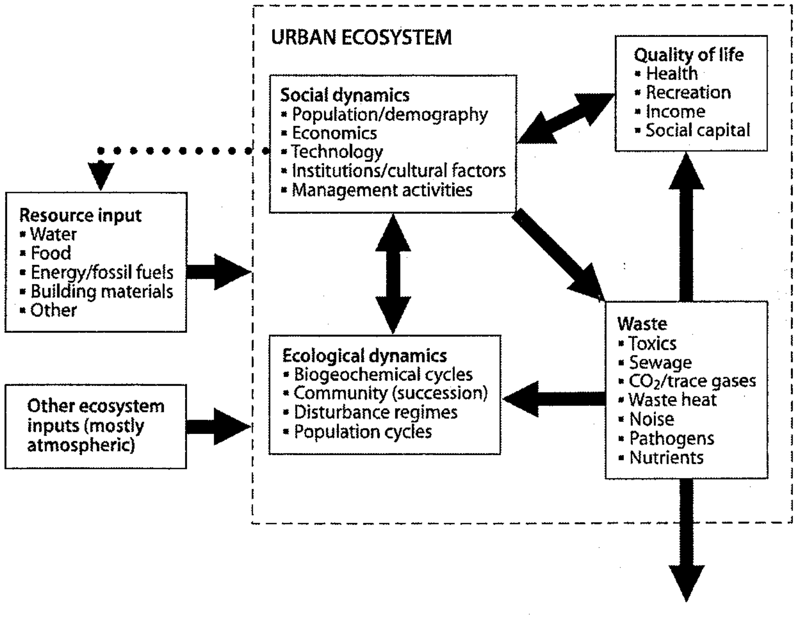

Fig. 5.1. Conceptual diagram of an urban ecosystem showing interrelationships between human and biophysical components. *Dashed line* indicates ecosystem boundary if city considered a "well-mixed reactor" such that inputs and outputs into and out of the ecosystem can be measured. Model includes feedback loop from the human social component of the system to resource acquisition, i.e., demand for resources dependent on population density, quality of life issues, and the ability of ecosystem to assimilate waste

natural) to the urban ecosystem were an order of magnitude higher than inputs to the surrounding desert areas. Furthermore, human hydrologic modifications within the urban area promoted the accumulation of N within the ecosystem.

Urban ecosystems maintain a high state of resiliency due to the socioeconomic influence of their inhabitants. This enables them to continually obtain resources from other areas. However, this dependence disconnects urban populations (and their social systems) from locally occurring ecological constraints (Luck et al. 2001). As a result, the ecological impact of cities can extend to regional and global scales without negative feedbacks directly affecting the system itself (Rees 2003). Moreover, urban areas can affect adjacent non-urban ecosystems through the production and transport of waste products and by altering regional weather patterns (e.g., Cerveny and Balling 1998; Shepherd and Burian 2003; Gregg et al. 2003).

An alternative to the well-mixed reactor approach is to delineate and recognize the parts or patches of urban ecosystems (Collins et al. 2000; Pickett et al. 2001). With this approach, ecosystems and the patches that comprise them are treated as well-mixed reactors at some scales and heterogeneous systems at others (Grimm et al. 2003). In this way input-output budgets can be developed not only for entire cities but also for individual functional parts of the urban ecosystem. This allows investigations of the effects of urban development on biogeochemical processes of a patch of lawn or remnant forest, which themselves may have cycling and fluxes of resources that interact with neighboring patches. Pouyat et al. (in press) suggested that researchers should exploit the heterogeneity of urban landscapes using the diverse array of patches as surrogates for field manipulations, or "natural experiments."

In this chapter we address the question: what is the net effect of urban land-use change on biogeochemical cycles at local and metropolitan area scales? Our use of the term *urban* is inclusive of landscapes having >2500 persons km^{-2} or densities at which human populations cannot be supported by local agricultural production (Ellis 2004). We use the phrase *urban land-use change* to describe coarse scale conversion of natural or agricultural ecosystems to urban land uses, as well as fine scale or local changes in land use that may occur in post-urban development. We will focus on the local scale because it is the most relevant one in making mechanistic assessments of urban biogeochemical processes. Our discussion relies on data available in the literature (mostly from temperate systems of North America) and from current research at the Long Term Ecological Research (LTER) site in Baltimore, MD USA (Baltimore Ecosystem Study). We devote most of our discussion to carbon (C) and nitrogen (N) pools and fluxes, both of which are important at multiple scales (Vitousek et al. 1997; Schlesinger and Andrews 2000) and may be indicators of urban stress and disturbance (Carreiro et al. in press). While heavy metals and organic compounds are important constituents in urban environments, their effects on plants, animals, and human health are beyond the scope of this chapter. Nor do we discuss the far-reaching effects of natural resource importation by cities, i.e., the aforementioned urban ecological footprints. The ability of urban inhabitants to exploit land resources at great distances has important anthropogenic impact on biogeochemical cycles at regional and global scales. But we believe that these impacts should be weighed against the effect of alternative settlement patterns on a per-capita basis – a comparison that requires extensive discussion. Excellent discussions of urban ecological footprints are included in Folke et al. (1997) and Rees (2003).

5.2 Urban Land-Use Change

In developed nations during the last half century, urban growth has occurred in a dispersed pattern relative to older and more densely populated cities that developed before and during the Industrial Revolution. Between 1980 and 2000 alone, land devoted to urban uses grew by more than 34% in the United States (USDA Natural Resource Conservation Service 2001). By contrast, the population grew by only 24% during the same period (USDC Census Bureau 2001). This dispersed growth pattern, or suburbanization, has occurred at the expense of agricultural and forested land (Imhoff et al. 1997). A recent analysis based on satellite imagery suggests that the land area covered by impervious surfaces (such as pavement and buildings) is larger than the surface area covered by herbaceous wetlands in the conterminous United States, or roughly the size of Ohio USA (Elvidge et al. 2004).

Although biogeochemical effects of agricultural land conversions and their recovery after abandonment have been relatively well studied, conversions to urban land uses have received little attention (Pouyat et al. 2002). We know that agricultural uses lead to drastic changes in biogeochemical cycling (Matson et al. 1997). However, once these lands are abandoned, many of the biogeochemical functions of the preagricultural ecosystem are recovered. For example, forest regrowth on agriculturally abandoned land has resulted in a gradual recovery of above- and below-ground C pools in the eastern and central United States (Houghton et al. 1999; Caspersen et al. 2000; Post and Kwon 2000). In fact, what are now considered "pristine" forest habitats at one time were some of the most densely populated regions of the world, for example, the Central American Mayan forests (Turner et al. 2003) and the Amazon River Basin (Heckenberger et al. 2003).

Converting agriculture, forest, and grasslands to urban and suburban land uses entails a complex array of land and ecosystem alterations. Dense human inhabi-

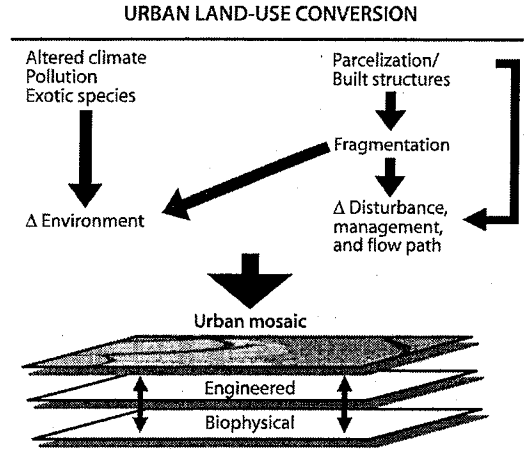

Fig. 5.2. Conceptual diagram of the effect of urban land-use conversions on native ecosystems. As landscapes are urbanized, natural habitats are increasingly fragmented as parcels of land ownership become smaller. During this process humans introduce novel disturbance and management regimes and flow paths to the landscape (*arrows on right*). Concurrently there is a change in the environmental conditions in which the ecosystem functions (*arrow on left*). See text for detailed explanation (modified from Pouyat et al. 2003)

tation along with urban land-use change necessitates the construction of various built structures, e.g., roads, buildings, civil infrastructure, as well as the introduction of human activities. The spatial pattern of these human features and activities are largely the result of "parcelization," or the subdivision of land by property boundaries, as landscapes are developed for human settlement. Parcelization typically leads to habitat loss and fragmentation as landscapes are continually subdivided into ever smaller patches, each with an individual land owner. Overlaid on the patch mosaic are various environmental factors that typically are associated with urban areas. The result is a patchwork or landscape mosaic of remnant ecosystems, managed parcels, and a built environment that overlay the preexisting ecosystem's biophysical features (Fig. 5.2).

5.3 Urban Environmental Factors

A complex of environmental factors altered by urbanization potentially affects biogeochemical cycling: atmospheric, soil, and water pollution; CO_2 emissions; micro- and meso-climates; and introductions of exotic plant and animal species (Pouyat et al. 1995). In some metropolitan areas, the net effect of such factors is analogous to predictions of global environmental change (Carreiro and Tripler 2005), e.g., increased temperatures and rising atmospheric concentrations of CO_2 (Ziska et al. 2004). In the following sections we discuss environmental factors that could affect biogeochemical cycles in urban and suburban landscapes.

5.3.1 Climate and Atmospheric Composition

Urban environments are characterized by localized increases in temperature known as the "heat island" effect (Oke 1990). Urban heat islands occur when vegetation cover is replaced by built structures. This reduces evapotransporation rates while the introduction of building materials increases the absorption and storage of solar energy that later is released as sensible heat (Oke 1990). The combined effect of these changes is to increase minimum (and to a lesser degree maximum) temperatures such that the difference between rural and urban temperatures typically is greatest several hours after sunset. This difference has been as much as 3 to 5 °C (Brazel et al. 2000). While cities embedded within desert ecosystems have warmer temperatures during the night, they typically have lower temperatures during the day, i.e., the "oasis effect" (Brazel et al. 2000).

How higher urban temperatures affect plant growth depends partly on the interactive effect of other environmental factors that may enhance plant growth such as N deposition and concentrations of atmospheric CO_2, the latter of which can reach 520 ppm in urban areas (Idso et al. 2002; Körner and Klopatek 2002; Pataki et al. 2003), or nearly double the preindustrial level of 280 ppm. Still other urban environmental factors may negatively affect plant growth such as tropospheric ozone (Chappelka and Samuelson 1998; Skärby and Ro-Poulsen 1998; Gregg et al. 2003).

The interactive effect of these multiple atmospheric and climate factors on plant productivity is a major uncertainty in global change research due to the difficulty of implementing controlled, factorial experiments in most ecosystems (Norby and Luo 2004). Hence, ecological studies in urban ecosystems have been proposed as a means of improving our understanding of multiple effects of global change (Pouyat et al. 1995; Grimm et al. 2000; Carreiro and Tripler 2005). Although this work is at an early stage, some results have been reported. Ziska et al. (2004) found plant productivity increases of 60 to 115% along an urban-rural gradient in Maryland, USA that corresponded to a 21% increase in CO_2 and a 3.3 °C temperature increase at the urban end of the gradient. Gregg et al. (2003) found that the growth of cottonwood seedlings was highest in the urban portion of an urban-rural gradient in the New York City area because of high ozone concentrations in rural areas downwind of urban pollution. Outside of the Los Angeles area, a large body of research in the San Bernardino Mountains Gradient Study has shown that foliar injury and chlorotic mottling is common at polluted sites due to ozone exposure, concurrent with decreased root biomass and increased aboveground biomass due to high N deposition (Arbaugh et al. 2003; Grulke et al. 1998; Grulke and Balduman 1999).

Additional work is necessary to determine whether these trends are robust in a large number of urban ecosystems.

Differences in air temperature also have been noted for soil temperatures. Mean annual temperatures in highly disturbed soils (0–10 cm depth) on a playground in New York's Central Park were more than 3 °C warmer than in soils in an adjacent wooded area (Mount et al. 1999). In comparing soil heat flux of several urban surfaces, Montague and Kjelgren (2004) found that heat fluxes were greatest beneath asphalt and concrete and least under pine bark mulch. For woodland soils in the New York City metropolitan area, surface temperatures (2 cm depth) differed by as much as 3 °C between urban and rural forest patches (Pouyat et al. 2003). Assuming that these soil temperatures are representative of those in urban areas, the difference between highly disturbed soils and natural forested soils in a metropolitan region can be as high as 6 °C. This difference in temperature can have significant effects on microbial activity and N availability in soil (Carreiro et al. in press).

Higher air temperatures and heat loading by impervious surfaces in urban areas also can affect the temperature of streams. First- and second-order streams in urban areas may have less shade than their rural counterparts and receive considerable runoff from impervious surfaces, which typically store a great amount of heat energy from solar radiation. In headwater streams, the major effect of urban-heat loading occurs early in the storm event as the first flush of runoff reaches the stream. The amount of temperature change is related to the percentage of impervious area in the watershed. For example, stream-temperature monitoring in the Baltimore Ecosystem Study showed that storm events in headwater streams resulted in a rapid increase in temperature. The increase was related directly to the housing density and proportion of impervious cover in the watershed (Fig. 5.3a). Similarly, in a study of headwater streams in Wisconsin and Minnesota USA, the percentage of impervious area in the watershed had a positive, linear relationship with water temperature during dry weather flows (Wang et al. 2003). Stream invertebrates and leaf breakdown processes are highly sensitive to elevated stream temperatures (Wang and Kanehl 2003; Webster and Benfield 1986).

Besides temperature, soil-moisture regimes can be modified by urban environments. Disturbed soils in urban areas typically have limited infiltration capacities due to hydrophobic surfaces, crust formation, and soil compaction (Craul 1992; Pouyat and Effland 1999). Compounding slow infiltration rate is the potential of impervious surfaces to drain concentrated flow into pervious areas. This can prevent water infiltration even during small storm events. Soil-moisture levels also can be reduced in urban areas due to higher air temperatures, which generate higher evaporative and transpirative demand on soil water resources. In contrast to factors that reduce soil moisture levels, soils in urban areas often are irrigated and have abrupt textural and structural interfaces that can restrict drainage resulting in higher soil-moisture content (Craul 1992). Moreover, below-ground infrastructures such as pressurized potable water distribution systems can leak water into adjacent soils by as much as 20 to 30% (Law et al. in press).

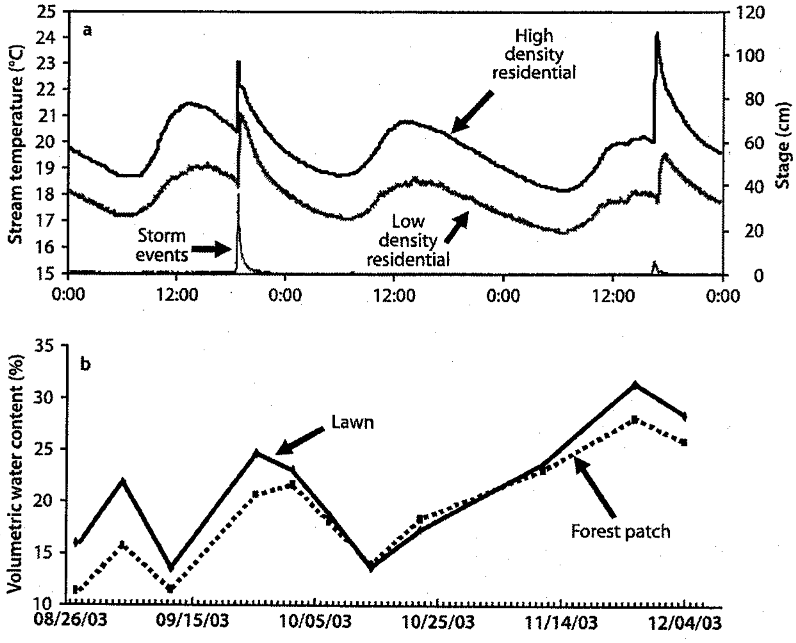

Fig. 5.3.
a Stream temperatures, May 25 to 27, 2004, for high density residential and lower density residential small headwater catchments. Stage at one of watersheds (*right axis legend*) shows two storm events (0.63 and 0.15 inches of rain). b Soil moisture (% by volume) of soils sampled at weekly intervals (depth of 0 to 10 cm) at Cub Hill eddy flux tower site of Baltimore Ecosystem Study. In forest patch, measurements taken along three transects perpendicular to slope ($n = 30$). Lawn measurements taken in two parcels along 30-m transects ($n = 24$)

Currently, we lack long-term continuous data with which to make comprehensive comparisons of soil moisture between urban and rural forest patches or between urban forest soils and disturbed soils. In the Baltimore Ecosystem Study, synoptic measurements of soil moisture (0–5 cm depth) in a medium-density residential area and adjacent deciduous woodland indicate that differences in soil moisture occur between land-use and cover types in urban landscapes. Residential lawns within a 150 m radius of the forest patch showed that un-irrigated turf soils had higher moisture levels than forest soils during the summer. There were no differences between the two patch types after leaf drop (end of October) (Fig. 5.3b). Presumably, differences in the summer were due to higher transpiration rates of the broad-leaved trees. In late fall and early winter, the moisture level in turf soil increased more rapidly than in forest soil, possibly reflecting concentrated runoff from roof gutters or snow melt (Fig. 5.3b). How these differences affect C and N cycling is being studied.

5.3.2 Atmospheric and Soil Pollution

Urban environments usually have higher concentrations and depositional fluxes of atmospheric chemicals than rural environments (Gatz 1991). Most urban atmospheric pollutants originate from the combustion of fossil fuels and industrial emissions. These include nitrogen oxides, sulfur oxides, heavy metals, and various organic chemicals. The pollutants are emitted locally into a restricted geographic area, particularly relative to the area from which the resources were derived, resulting in high atmospheric concentrations (Wong et al. 2004). High pollutant concentrations in the atmosphere combined with numerous temperature inversions increase the depositional rates of chemicals into urban areas (Seinfeld 1989). Since the capacity of ecosystems to assimilate atmospheric chemicals is correlated with the amount of leaf area of plants and soil biological activity, highly altered urban ecosystems have greatly diminished capacities to assimilate chemicals, especially when inputs are high.

Not surprisingly, studies of forest ecosystems along urban-rural gradients suggest that urban forest remnants receive relatively high amounts of heavy metals, organic compounds, N, and sulfur (S) in wet and dry atmospheric deposition (Pouyat and McDonnell 1991; Lovett et al. 2000; Wong et al. 2004). Lovett et al. (2000) quantified atmospheric N inputs over two growing seasons in oak forest stands along an urban-rural gradient in the New York City metropolitan area. They found that N entering urban forests in throughfall was 50 to 100% greater than the N flux into rural and suburban forests. These inputs fell off in the suburban stands 45 km from the city (New York's Central Park), which Lovett et al. suggested was due to the reaction of acidic anions with larger alkaline dust particles (Ca^{2+} and Mg^{2+}) of 2 μm or less in diameter that precipitated closer to the city. They hypothesized these particles (urban dust) originated from construction and demolition activity within the city. Similar results were found for the city of Louisville, KY USA and the San Bernardino Mountains in the Los Angeles metropolitan area where both N and base cation deposition rates into urban forest patches have been found to be higher than in rural forest patches (Bytnerowicz et al. 1999; Carreiro et al. in press; Fenn and Bytnerowicz 1993).

Evidence for a similar depositional pattern was found for heavy metals along the New York City transect. Pouyat and McDonnell (1991) found two to threefold increase in contents of lead (Pb), copper (Cu), and nickel (Ni) in urban than in suburban and rural forest soils. A similar pattern but with greater differences was found by Inman and Parker (1978) in the Chicago, IL USA, metropolitan area, where levels of heavy metals were more than five times higher in urban than in rural forest patches. Other urban-rural gradient studies have shown the same pattern though for smaller cities differences between urban and rural stands were narrower (Carreiro et al. in press; Pavao-Zuckerman and Coleman in press). Finally, Wong et al. (2004) found a large gradient of Polycyclic Aromatic Hydrocarbons (PAH) concentrations in forest soils in the Toronto (Canada) metropolitan area, with concentrations decreasing with distance from the urban center to surrounding rural areas by a factor of at least 60.

How these pollutants affect biogeochemical fluxes is uncertain, but preliminary results suggest that the effects are variable and depend on various urban environmental factors (Carreiro et al. in press). For example, Inman and Parker (1978) found slower leaf litter decay rates in urban stands that were highly contaminated with Cu (75.7 mg kg^{-1}) and Pb (399.9 mg kg^{-1}) compared to unpolluted rural stands. Apparently, the heavy metal contamination negatively affected the activity of decomposers in these urban stands. By contrast, where heavy metal contamination of soil was moderate to low relative to other atmospherically deposited pollutants such as N, rates of decomposition and soil N-transformation increased in forest patches near or within major metropolitan areas in southern California (Fenn and Dunn 1989; Fenn 1991), Ohio (Kuperman 1999) and southeastern New York (Carreiro et al. in press).

5.3.3 Introductions of Exotic Species

The expansion of urban areas has resulted in some of the greatest local extinction rates observed in the world (McKinney 2002). Cities also have become epicenters of many of introductions of nonnative species (Steinberg et al. 1997; McKinney 2002), some of which have become important pathogens or insect pests, e.g., Dutch elm disease and the Asian long-horned beetle. Local extinctions of native species and the invasions by urban-adapted nonnative species have resulted in a pattern where the *species richness of nonnative species increases from outlying rural areas to urban centers while native species* decrease (Blair 2001; Hope et al. 2003).

While native plant and animal species richness may be depressed, plant species richness apparently is greater in urban than in rural environments. Urban plant communities tend to have higher species richness and diversity than natural forests because of planting choices of multiple land owners (Nowak 2000). For instance, Nowak et al. (2002) found 81 and 57 tree species with Shannon-Weiner diversity index values of 3.6 and 3.4 in Baltimore MD and Brooklyn NY, USA, respectively. These values are higher than those found for eastern deciduous forests (1.9 to 3.1) of the United States (Barbour et al. 1980). In Phoenix, USA, overall species richness was greater in the urban area than in the surrounding desert. Within the urban area, species diversity was positively correlated to median family income, such that the highest biodiversity was associated with the wealthiest neighborhoods, a phenomenon termed the "luxury effect" (Hope et al. 2003).

Invasive species can play a disproportionate role in controlling C and N cycles in terrestrial ecosystems (Bohlen et al. 2004; Ehrenfeld 2003). Therefore, the relationship between invasive species abundances and urban land-use change has important implications for biogeochemical cycling of C and N (Pouyat et al. in press). For example, in the northeastern and mid-Atlantic United States where native earthworm species are rare or absent, urban areas are important foci of nonnative earthworm introductions, especially Asian species from the genus *Amynthas*, which are expanding toward outlying forested areas (Steinberg et al. 1997; Groffman and Bohlen 1999; Szlavecz et al. 2006). These invasions into forests have resulted in highly altered C and N cycling processes (Bohlen et al. 2004; Carreiro et al. in press). Other examples of species invasions in urban areas that have altered C and N cycles include shrub (*Berberis thunbergii*), tree (*Rhamnus cathartica*), and grass (*Microstegium vimineum*) species (Ehrenfeld et al. 2001; Heneghan et al. 2002; Kourtev et al. 2002).

5.4 Disturbance and Management Effects

For most urban landscapes, human-caused disturbance is more pronounced during rather than after the land-development process. Urban development of land typically includes the clearing of existing vegetation, massive movements of soil, and the building of structures. The extent and magnitude of these initial disturbances is dependent on infrastructure requirements (e.g., storm water retention ponds), topography, and other site limiting factors. As an example, a topographic change analysis of 30 development projects in Baltimore County USA showed that the total volumetric change of soil per development was positively correlated with mean slope of the site ($r = 0.54$, $p = 0.002$) (McGuire 2004). In addition, these development projects resulted in massive soil disturbances with the potential to have large effects on soil organic C. Using data from this study, we estimated that the amount of soil organic C that was disturbed during a development project covering 2600 m^2 was roughly 2.7×10^4 kg. This assumes that the original soil had a soil organic C density of a deciduous forest (10.3 kg m^{-2}) to a depth of 3 m, i.e., the average depth of the soil disturbance. How much soil organic C actually is lost during the development process is unknown and depends partly on the ultimate fate of the surface soil layers (topsoil).

The previous development scenario typically predates residential, institutional, and commercial land uses, for which turfgrass is the resultant plant-cover type. Nowak et al. (1996) found that residential, institutional, and commercial land uses made up on the average 39.0, 5.7, and 13.6% (total 58.3%) of the land cover in the major U.S. metropolitan areas, respectively. The total estimated lawn area for the conterminous USA is 16.38 ±3.58 million ha (Milesi et al. 2005). Management of this vast area of turfgrass typically includes adding pesticides and supplements of water and nutrients as well as being regularly clipped during the growing season. In the following sections we discuss how these lawn and horticultural management practices may affect biogeochemical cycles.

5.4.1 Lawn and Horticultural Management

To manage turfgrasses in lawns, homeowners and institutional land mangers in the USA apply about 16 million kg of pesticides each year (Aspelin 1997) as well as fertilizers at rates similar to or exceeding those of cropland systems (Talbot 1990). In addition, lawns typically are clipped on a regular basis during the growing season. This management scenario may or may not result in increases in plant productivity. In comparing results from studies of mowed lawns in Wisconsin, California, and Maryland, Falk (1980) estimated that the range for net primary productivity in temperate climates was about 1.0 to 1.7 kg ha yr^{-1}, most of which is below ground. Although above-ground productivity increased with N fertilization and irrigation, total productivity (above and below ground) did not differ. Other studies have shown somewhat lower productivity rates for lawns (0.6 to 0.7 kg ha yr^{-1}) (Blanco-Montero et al. 1995; Jo and McPherson 1995). Still others have reported direct relationships between management and productivity (Heckman 2000; Kopp and Guillard 2002; Golubiewski 2006). As a comparison, net primary production in temperate grasslands ranges from 0.1 to 1.5 kg ha yr^{-1} (Leith 1975).

While total turfgrass productivity may or may not increase with management inputs, soil organic C sequestration appears positively related to supplemental inputs of fertilizer and water. In comparing surface soils of 15 golf courses, Qian and Follett (2002) found that total C sequestration rates ranged from 0.9 to 1.0 t ha^{-1} yr^{-1}. This is similar to rates obtained for the recovery of perennial grasslands following cultivation (1.1 t ha^{-1} yr^{-1}) (Gebhart et al. 1994), but much higher than unmanaged grasslands (0.33 t ha^{-1} yr^{-1}) (Post and Kwon 2000). Similarly, Higby and Bell (1999) found that soil organic matter was higher in fertilized golf course fairways than in adjacent unfertilized areas. Using long-term simulations of the CENTURY model for lawn ecosystems, Qian et al. (2003) showed that N fertilization coupled with a management regime in which grass clippings were left on site increased soil organic C accumulations by up to 59% in comparison to sites that were not fertilized and clippings were removed. How much C is sequestered in soils under turfgrass depends on the differential effects of management practices on above- and below-ground productivity and the rate of decomposition in the surface soil (Qian and Follett 2002).

While lawn care management practices input chemicals at rates comparable to cropland systems, they are potentially less disruptive of biogeochemical cycles. Cropland systems have a greater magnitude and frequency of soil disturbances and generally remove a greater proportion of the standing crop (Asner et al. 1997; Pouyat et al. 2003). As a result, cropland systems lose substantial amounts of soil organic C and N (Matson et al. 1997). The net effect of an initial soil disturbance and subsequent lawn care management on C and N dynamics is less certain. In natural ecosystems, C sequestration increases with N additions until N is no longer limiting (Asner et al. 1997), i.e., a condition of N saturation (Aber et al. 1989). Nitrogen limitation is determined by biological demand and the ability of soil to accumulate N, which, in turn, is attributed partly to site history, climate, soil fertility, and vegetation type (Aber et al. 1998). In cultivated ecosystems, N additions over the long term can increase C storage, but due to annual disturbances of soil and the removal of plant biomass, these gains are less than those for grassland and forested ecosystems. On the other hand, turfgrass ecosystems can accumulate soil organic C at rates similar to those for grasslands and some forests due to the absence of annual soil disturbances (Qian and Follett 2002; Pouyat et al. 2003).

Carbon sequestration is an important regulator of the net effect of ecosystems on atmospheric carbon dioxide levels, which influence the atmospheric "greenhouse effect." This effect is also influenced by other soil-atmosphere gas exchanges, especially nitrous oxide and methane that are influenced by urban land-use change. Kaye et al. (2004) found that lawns in Colorado had reduced methane uptake and increased nitrous oxide fluxes relative to native shortgrass steppe, with fluxes similar to irrigated corn. Goldman et al. (1995) found reduced methane uptake in forests in urban areas. In both cases, these changes were linked to nitrogen enrichment associated with urban land-use.

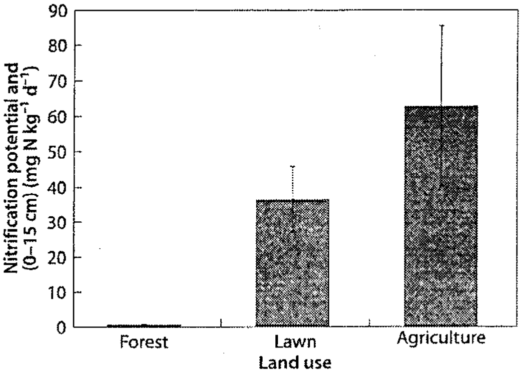

Fig. 5.4. Mean (±S.E.) potential net nitrification rates (mg NO$_3^-$ kg d^{-1}) of 0–15 cm mineral soil samples after 14-d incubation period of 0.04 ha plots in the Baltimore LTER study. *Bars* represent comparison of forest, lawn, and agriculture land-use types (*n* = 14, 10, and 10, respectively) (data from C. Williams, unpubl.)

The potential of turfgrass ecosystems to sequester soil organic C and thus maintain or increase microbial demand for N suggests a high capacity of these systems to retain N. Preliminary data from comparisons of lawn, forest, and cultivated patch types in the Baltimore Ecosystem Study suggest that lawns fall between temperate forests and cultivated land in C sequestration, N cycling, and N retention (Fig. 5.4). Currently, we are investigating the interactive effects of high N inputs, C pool dynamics, biomass removal, and altered hydrologic pathways in lawn ecosystems.

5.4.2 Management Effort

Perhaps the most distinctive characteristic of urban landscapes is the number and diversity of potential land managers on a per area basis. Parcels of land generally tend to become smaller from rural to suburban and urban residential areas, with a slight increase at the urban core (Pouyat et al. in press). This pattern potentially results in greater variation in management practices in high-density residential areas than in larger rural parcels because of the greater number of potential land managers (Collins et al. 2000). Parcel size also may affect the amount of effort of individual land owners. For example, in comparing fertilizer application rates in two small residential watersheds in Baltimore County, MD USA, Law et al. (2004) found that the application rate per unit of lawn area was higher for the watershed with smaller lot sizes, suggesting that homeowner input is inversely related to lot size. They also found that soil bulk density, an indicator of soil compaction, was positively related to the average annual application rate of fertilizer. The relationship between human impact and lot size also was evident at a coarser scale. In Baltimore, mean bulk densities of surface soils (0–5 cm) in high- and medium-density residential land-uses were 1.30 and 1.17 Mg m^{-3}, respectively. Bulk density was inversely related ($r = 0.52$, $P < 0.001$) to soil organic matter (Pouyat et al. 2002).

In addition to parcel size, management effort varies with socioeconomic factors in residential neighborhoods. In a survey conducted in the Columbus, OH USA, metropolitan area, Robbins et al. (2001) found that residents with houses valued at more than $250 000 were much more likely to apply chemicals on their lawns than owners of houses valued at less than $80 000. However, Law et al. (2004) and Osmond and Platt (2000) found that households with intermediate value applied the most lawn fertilizer. Whatever the relationships between socioeconomic factors and management effort, decisions involving horticultural management in much of the urban landscape are largely independent of government regulations (Robbins et al. 2001).

Management effort also should be related to the magnitude of the natural constraints on the system. For example, irrigation rates for lawns in dry land areas should be higher than for lawns in more temperate climates. The net result of these relationships is that while natural constraints on biological systems in urban landscapes are important, human desires and efforts to overcome these constraints result in vegetation structure and soil characteristics that are remarkably similar across urban settlements on a global scale (McDonnell et al. in press). Therefore, urban landscapes might be similar in vegetation structure and soil characteristics at coarse scales (regional and global) but highly variable at finer scales (Pouyat et al. 2003).

5.5 Effects of Built Environment

To fully grasp the effects of urban land-use change on biogeochemical cycles, we need to understand the ways the built environment affects energy and material flows in urban ecosystems. Our conceptual framework of urban land-use change (Fig. 5.5) incorporates the importance of built structures and the effect of impervious surfaces on ecosystem processing by differentiating between natural and engineered templates, either of which may constrain ecosystem processes. For example, both soil drainage and irrigation infrastructure can partially constrain distributions of plant species and trace gas fluxes, as well as the movement of nutrients and contaminants in urban ecosystems. Perhaps more interestingly, the natural and engineered templates may intersect; the proportion and "connectivity" of each determining the importance of each template in constraining biogeochemical processes. For example, hydrologists use the concept of "effective imperviousness" to more accurately model runoff volumes, that is, the connectivity between the natural and engineered templates. Effective impervious areas (EIA) drain immediately to storm-drain pipes and are equal to the total impervious area (TIA) minus the impervious surfaces draining to pervious areas where infiltration is possible (Sutherland 1995).

If measurements of EIA and TIA were made at the scale of a small watershed (<100 ha) for a given metropolitan area we suggest they would form a continuum of watersheds ranging from totally pervious cover to those with an almost entirely engineered template. Thus, the continuum would represent an array of small watersheds varying in area and connectivity of the engineered and natural templates, at least for hydrologic flow paths. Measurements of connectivity between engineered and natural templates also can be made for other flow paths (e.g., organic-matter transport or atmospheric deposition onto built and biological surfaces) and for other ecosystem delineations at various scales.

These relationships are important in determining the degree to which the engineered system mediates flows of energy and material. A totally engineered system has pathways that entirely separate or disconnect material flows from natural elements of the ecosystem, though over time, all human built structures fail to function to some degree, resulting in exchanges between engineered and natural templates. This disconnect "short circuits" the biological system, which diminishes the system's overall ability to buffer changes in water, nutrient, and contaminant inputs. As a result, the system's capacity to retain or process these materials is altered. For totally engineered systems, atmospherically derived contaminant and nutrient inputs can accumulate on impervious surfaces and be washed off repeatedly by small rainfall events into the engineered system. Thus, there is little chance of being accumulated in soil or biologically processed in the terrestrial system. In addition, gaseous exchanges between the atmosphere and the soil-plant continuum will be diminished, again short circuiting the ability of the biological system to assimilate C or gas-phase contaminants. We propose that the point in the continuum at which the constraints of the engineered and natural templates shift in importance from one to the other may serve as a useful delineation between urban and non-urban-dominated ecosystems.

At some point, all engineered pathways flow to natural systems, for example, sewage treatment outflows, power plant emissions, storm water runoff, and effluent from septic systems. All of these systems have important effects on ecosystems receiving their flows. In most cases, flows from engineered pathways concentrate materials collected from a larger area. The functional size of the area is dependent on the extent and connectivity of the infrastructure. For example, septic systems collect waste from a single detached house and release these flows into a relatively small volume of soil, while sewage treatment

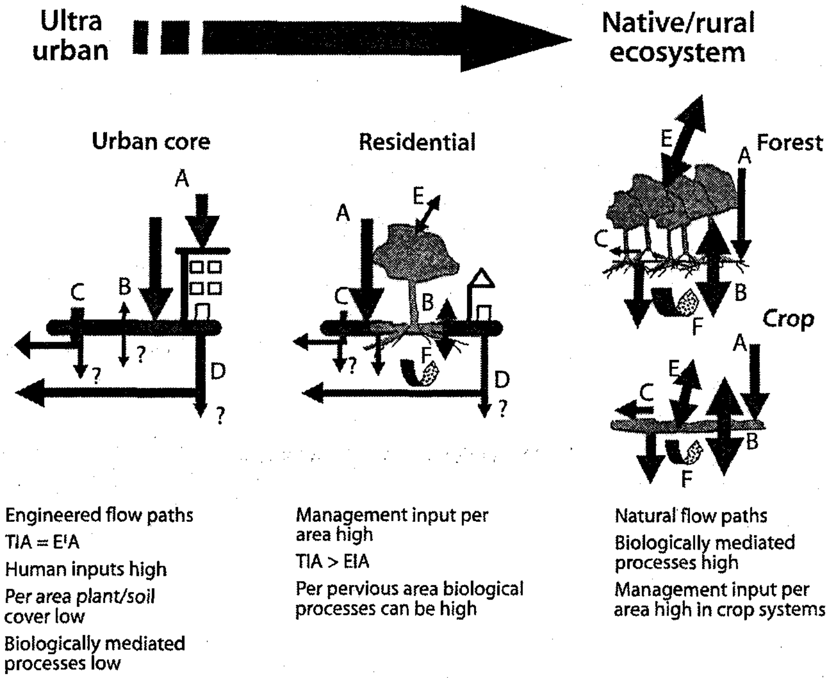

Fig. 5.5. Conceptual framework where anthropogenic and biophysical ecosystem characteristics form a continuum from highly urban (almost entirely human-made) to native or rural ecosystem types (those with the least human modification). Agricultural ecosystems also are depicted. The framework incorporates the importance of built structures and the effect of impervious surfaces on ecosystem processing. At the urban core, engineered flow paths (*C* and *D*) disconnect material inputs and flows (*A*) from natural processing that occurs in native/rural ecosystems. In these areas, the connectivity between the engineered and biophysical templates is low (*B*). In residential areas, the connectivity between engineered and biophysical templates can be relatively high depending on the spatial relationship of impervious and pervious surfaces. Management and environmental inputs in residential areas can be high (*A* and *E*) on a per-unit pervious area. However, depending on site history, soil type, and the concentration of flows these areas can have surprisingly high cycling rates (*F*) for processing or storing these inputs (adopted from Kay et al. 2006)

plants receive waste flows from tens and hundreds of thousands of residences and release flows to surface waters with varying abilities to assimilate the waste. As mentioned previously, the waste flows in both cases are byproducts of resources originating outside the urban ecosystems – often from great distances. Compounding the concentration of flow is the ability of built surfaces to accumulate various atmospherically derived contaminants through the development of organic films that increase the capture efficiency of impervious surfaces (Law and Diamond 1998). These films ultimately increase the movement of contaminants from the atmosphere to surface waters following wash-off (Diamond et al. 2000).

A clear example of the effects of concentrated flows is the impact of impervious cover on stormwater flows and the resulting impact on urban-stream ecosystems. Schueler (1994) and Morse et al. (2003) concluded from reviewing data in the literature that as little as 6% impervious cover in a watershed can result in measurable responses in stream-channel morphology and invertebrate populations. Moreover, changes in soil-drainage patterns associated with urban development and altered stream morphology can have marked effects on soil organic C and N processing, particularly in riparian soils (e.g., Groffman et al. 2002). An important question in urban hydrology is whether responses in stream ecology and riparian function differ between catchments with the same total impervious area but with different measures of effective impervious areas (Walsh 2004).

The tendency of the built environment and human activity to concentrate flow paths and chemical inputs can result in the development of "hot spots" in the landscape. Hotspots are areas or patches that show disproportionately high reaction rates relative to the surrounding area or matrix (McClain et al. 2003). The concept of hotspots developed from studies of N processing in soil cores (Parkin 1987) and riparian zones that showed that anoxic microsites with high C content were zones of elevated denitrification rates. Generally, hotspots are sites where reactants for specific biogeochemical reactions coincide in an environment conducive for the reaction to take place (McClain et al. 2003). Human activities and the introduction of built structures provide such conditions in urban landscapes at various scales. Examples include septic systems, golf greens, stormwater retention basins, and compost piles.

The importance of hotspots in the biogeochemistry of urban ecosystems is largely unknown. In urban fringe areas, there is concern about nitrate contamination of ground water from septic systems, which when functioning properly can be thought of as purposely engineered hot spots for processing human waste (Band et al. 2004). Hotspots in urban landscapes also can be sinks for contaminants, nutrients, or C. For example, the conversion of what was predominately prairie grassland into an urban landscape resulted in the development of C sequestration hotspots in the Boulder, CO USA, metropolitan area (Golubiewski 2003). These hotspots were composed of relatively small parcels with highly managed turfgrass and woody vegetation whose productivity rates and C storage were much higher than in the native grasslands they replaced. Similarly, detention basins designed to capture urban stormwater and protect streams may be hotspots of denitrification, replacing functions that occurred in riparian areas before urbanization (Groffman and Crawford 2003).

5.6 Assessing Biogeochemical Effects – *the Importance of Scale*

A critical task in using the ecosystem approach is the ability to make practical and meaningful delineations of ecosystem boundaries (Likens 1992). In nonhuman-dominated ecosystems, this task is challenging since many environmental factors vary as a continuum on the landscape. Ecologists and biogeochemists have addressed this challenge by using discontinuities of biophysical processes in the landscape as ecosystem boundaries, e.g., a watershed. Determining boundaries for urban ecosystems is an even greater challenge as there are no generally accepted ecological definitions of "urban" (Mcintyre et al. 2000). Nor do we know how urban environmental factors vary spatially (Pouyat et al. in press). This is especially problematic in viewing cities as well-mixed reactors because urban areas often are expanding in area, forming roughly concentric circles of development (McDonnell and Pickett 1990). For landscapes where development patterns are less dispersed, e.g., desert ecosystems, the boundary of urban expansion is more apparent (e.g., Baker et al. 2001).

To address the difficulty in delineating urban ecosystem boundaries for investigations of biogeochemical cycles, we propose a two-pronged approach: (1) conduct investigations for parts or patches of the urban landscape that have more readily recognizable boundaries, and (2) assess the interrelationship of these patches at multiple scales: one at the scale of individual parcels (<km) each with a distinctive landscape context, site history, land manager, and natural and engineered template, and the other at coarser scales, e.g., watershed or well-defined area of the landscape (e.g., Ellis 2004), to assess the interrelationships among parcels and investigate the net change in biogeochemical cycles due to urban-land transformations. At coarse scales, we suggest comparisons on both a per-unit pervious cover and per-unit total area basis. The latter includes built surfaces and considers urban landscapes as a whole. To distinguish between the two approaches, in the first case, productivity on a pervious-area basis can be greater than the native ecosystem replaced (or nonurbanized areas at the rural end of the continuum); in the second case, a city generally would have lower overall productivity than the native ecosystem it replaced. When human use of fossil fuels are included in these calculations, cities would be considered heterotrophic (primary productivity < total ecosystem respiration) rather than natural ecosystems, which are autotrophic (Grimm et al. 2003).

These relationships suggest a paradox of urban ecosystems: the engineered template reduces the ability of an ecosystem to assimilate or process energy and materials on a per-unit ecosystem basis such that when compared to natural systems urban ecosystems exhibit less biological activity. At the same time, the engineered template and human activity concentrate energy and matter into smaller areas (volumes) such that fluxes and biological activity will be higher on a per-unit pervious area basis than in the native ecosystem replaced. The latter characteristic is significant since the ability of biological systems to assimilate energy and material is determined at the scale at which organisms are using the resource. This can occur at very fine scales, e.g., hotspots. Thus, assessments at coarse scales that fail to distinguish between the built and pervious component of urban ecosystems will miss the scale at which biological processes are being constrained.

A comparison of N budgets of residential watersheds in the Baltimore Ecosystem Study illustrates the importance of scale. Law et al. (2004) calculated N input from fertilizer into a low- and medium-density residential watershed at three spatial aggregations: by the area of the watershed, subdivision, and residential lawns. For the more sparsely populated watershed, application rates were 9.5, 27.8, and 37.1 kg N $ha^{-1} yr^{-1}$ on a watershed, subdivision, and residential lawn area, respectively. For the more densely populated watershed, rates were 12.5, 26.7, and 83.5 kg N $ha^{-1} yr^{-1}$. Therefore, the greatest distinction between the two watersheds is on a lawn-area basis, or the scale at which added N will be processed by the soil community and management decisions will be made by individual land owners. In addition, the watershed with lower housing densities had an aggregated septic load that was significantly lower than N fertilizer inputs and roughly equal to atmospheric deposition inputs (Band et al. 2004). However, this septic load was concentrated in 1 to 2% of the watershed area (leach fields), resulting in a loading rate that was an order of magnitude higher than N loading per unit of watershed area. Indeed, stream sampling in the Baltimore Ecosystem Study revealed that nitrate concentrations were higher in watersheds at the suburban fringe that are septic-serviced than in central city streams or sanitary sewer-serviced suburbs with similar lawn fertilization rates (Band et al. 2004; Groffman et al. 2004).

As another example, Nowak (2000) estimated the amount of C sequestered by trees in Baltimore to be 0.71 t C $ha^{-1}yr^{-1}$. His estimate was for the entire land area of the city. Estimates for pervious areas in Baltimore are not available, though Jo and McPherson (1995) measured carbon budgets on a per-pervious-area basis for two residential neighborhoods in Chicago, IL USA. They found that C sequestered by woody vegetation was as high as 3.29 t C $ha^{-1}yr^{-1}$. Assuming that C budgets of residential areas are similar across urban ecosystems, comparison of these rates to gross sequestration rates for a 25 yr old loblolly plantation (2.6 t C $ha^{-1}yr^{-1}$) and a naturally regenerating spruce-fir forest (1.0 t C $ha^{-1}yr^{-1}$) (Birdsey 1996) suggests the urban values are somewhat low when the entire landscape is considered (thus the net change from converting from a natural ecosystem to an urban ecosystem), but high when only pervious surfaces are considered. This suggests that on average, urban trees have higher growth rates than trees growing in nonirrigated and nonfertilized stands under rural environmental conditions. This finding is consistent with measurements of the productivity of urban trees (McPherson 2000), urban-rural comparisons of containerized plants (Gregg et al. 2003), and eddy flux tower measurements above a residential area (Hom et al. 2003).

On the basis of these comparisons, we propose that the net effect of urban land-use change on biogeochemical cycles depends on the overall constraints of the native ecosystem replaced. For example, converting a temperate deciduous forest ecosystem to an urban landscape results in a decrease (per land area) in plant productivity, but potentially higher rates on a pervious area basis. By contrast, converting a desert ecosystem to an urban landscape should result in both a land-area and per-pervious-area increase in primary productivity. Differences in the direction of the ecosystem response are the result of the relatively severe limitations in soil-water availability in desert vs. temperate environments. Thus, the technical ability and desire of humans to manage for specific types of plant communities irrespective of natural limiting factors have resulted in a convergence of urban vegetation structure and ecosystem function on a global basis (Pouyat et al. 2003).

5.7 Summary and Conclusions

Urban areas represent ecosystems with modified biogeochemical cycles such that fluxes and pools of matter, energy, and organisms differ greatly from the previous unaltered ecosystem. The conversion of agriculture, forest, and grasslands to urban and suburban land uses entails a complex array of land and ecosystem alterations. These include the construction of various built structures, e.g., roads, buildings, and civil infrastructure, as well as the introduction of horticultural management practices. In addition to land altering activities, a complex of urban environmental factors can potentially affect biogeochemical cycling including atmospheric and soil pollution, CO_2 emissions, micro- and meso-climates, and introductions of exotic plant and animal species. In some metropolitan areas, the net effect of such factors is analogous to predictions of global environmental change, which we feel represents an opportunity to study ecosystem responses to such factors at a scale not practical in controlled field experiments.

To fully understand the effects of urban land-use change on biogeochemical cycles, the effect of the built environment must be considered. A conceptual framework of urban land-use change was presented that incorporates the importance of built structures and the effect of impervious surfaces on ecosystem processing. This framework differentiates between natural and engineered systems, either of which may constrain biogeochemical processes. A totally engineered system has pathways that entirely separate or disconnect material flows from natural elements of the ecosystem, though all human built structures fail to function to some degree, resulting in exchanges between engineered and natural templates. This disconnect "short circuits" the biological system, which diminishes the system's overall ability to buffer changes in water, nutrient, and contaminant inputs.

To assess the overall impact of urban land-use change on biogeochemical cycles, we propose comparisons at: (1) individual parcels each with a distinctive set of characteristics, and (2) coarser scales to assess the interrelationships among parcels and investigate the net change in biogeochemical cycles due to urban land transformations. At coarse scales, we suggest comparisons on both per-unit pervious cover and per-unit land area (ecosystem) basis. The latter includes built surfaces and considers urban landscapes as a whole. Preliminary analyses of the available data suggest that process rates on a pervious-area basis can be greater than those of the native ecosystem replaced, while on an ecosystem basis urban areas tend to have lower overall process rates than to the native ecosystem replaced. The former comparison suggests the continued importance of biological processes in urban areas, which we believe results from human efforts to overcome environmental constraints on biological processes and the tendency for horticultural management activities and urban infrastructure to concentrate flows of energy and matter into the remaining biologically active areas in the urban landscape.

We conclude that global and regional comparisons of urban ecosystems on an ecosystem and pervious-area basis will provide an analysis of the net effect of urban land conversion on biogeochemical cycles, e.g., soil C pools and fluxes, as well as insight into the mechanisms causing those changes. We hypothesize 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 and thus the potential to actually increase C sequestration; in more humid environments, the net effect will be a reduction in C sequestration rates. Therefore, we suggest that cross-system comparisons on regional and global scales will be necessary to determine the net effect of urbanization on biogeochemical cycles.

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