The Changing Metabolism of Cities

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Summary

Data from urban metabolism studies from eight metropolitan regions across five continents, conducted in various years since 1965, are assembled in consistent units and compared. Together with studies of water, materials, energy, and nutrient flows from additional cities, the comparison provides insights into the changing metabolism of cities. Most cities studied exhibit increasing per capita metabolism with respect to water, wastewater, energy, and materials, although one city showed increasing efficiency for energy and water over the 1990s. Changes in solid waste streams and air pollutant emissions are mixed.

The review also identifies metabolic processes that threaten the sustainability of cities. These include altered ground water levels, exhaustion of local materials, accumulation of toxic materials, summer heat islands, and irregular accumulation of nutrients. Beyond concerns over the sheer magnitudes of resource flows into cities, an understanding of these *accumulation* or *storage* processes in the urban metabolism is critical. *Growth*, which is inherently part of metabolism, causes changes in water stored in urban aquifers, materials in the building stock, heat stored in the urban canopy layer, and potentially useful nutrients in urban waste dumps.

Practical reasons exist for understanding urban metabolism. The vitality of cities depends on spatial relationships with surrounding hinterlands and global resource webs. Increasing metabolism implies greater loss of farmland, forests, and species diversity; plus more traffic and more pollution. Urban policy makers should consider to what extent their nearest resources are close to exhaustion and, if necessary, appropriate strategies to slow exploitation. It is apparent from this review that metabolism data have been established for only a few cities worldwide, and interpretation issues exist due to lack of common conventions. Further urban metabolism studies are required.

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Introduction

Cities grow in complex ways due to their sheer size, social structures, economic systems, and geopolitical settings, and the evolution of technology (Hall 1998). Responding to waves of new technology, cities have grown outward from small dense cores, first as linear transit cities and then as sprawling automobile cities. Changes in industry have also been important; obsolete factories have often closed down, leaving behind contaminated soil and groundwater. Risks associated with rebuilding on such brownfield sites have encouraged developers to pursue greenfield sites on the edges of cities. Ever-growing urban populations also fuel the expansion of cities. Yet even cities showing no change or decreasing populations, such as some older industrial cities, are still growing outward.

Forty years ago, in the wake of rapid urban expansion, Abel Wolman published a pioneering article on the metabolism of cities. Wolman (1965) developed the urban metabolism concept in response to deteriorating air and water quality in American cities—issues still recognized today as threatening sustainable urban development. Wolman analyzed the metabolism of a hypothetical American city, quantifying the overall fluxes of energy, water, materials, and wastes into and out of an urban region of 1 million people.

The metabolism of an ecosystem has been defined by ecologists as the production (via photosynthesis) and consumption (by respiration) of organic matter; it is typically expressed in terms of energy (Odum 1971). Although a few studies have focused on quantifying the embodied energy in cities (Zucchetto 1975; Huang 1998), other urban metabolism studies have more broadly included fluxes of nutrients and materials and the urban hydrologic cycle (Baccini and Brunner 1991). In this broader context, urban metabolism might be defined as *the sum total of the technical and socioeconomic processes that occur in cities, resulting in growth, production of energy, and elimination of waste*.

Since Wolman's work, a handful of urban metabolism studies have been conducted in urban regions around the globe. By reviewing these studies, this article describes how the urban metabolism of cities is changing. It also demonstrates how understanding of accumulation processes in the urban metabolism is essential to the sustainable development of cities.

Sustainable development can be understood as development without increases in the throughput of materials and energy beyond the biosphere's capacity for regeneration and waste assimilation (Goodland and Daly 1996). Given this definition, a sustainable city implies an urban region for which the inflows of materials and energy and the disposal of wastes do not exceed the capacity of its hinterlands. As discussed in this article, this definition presents difficulties in the context of cities dependent on global markets; nevertheless, it provides a relative bar against which progress may be measured. In quantifying material and energy fluxes, urban metabolism studies are valuable for assessing the direction of a city's development.

The objectives of this article are twofold. The first objective is to review previously published metabolism studies to elucidate what we know about how urban metabolism is changing. A previous review article on energy and material flows to cities has been presented by Decker and colleagues (2000), but it made only minor reference to the metabolism concept; it did not include reference to many of the studies considered here; and it did not specifically ask how urban metabolism is changing. The number of metabolism studies is not sufficient to apply any form of statistical analysis, and therefore, some might argue, to identify any generalizable trends. Nevertheless, the balance of evidence generally points to increasing urban metabolism.

The second objective is to identify critical processes in the urban metabolism that threaten the sustainable development of cities. It has been argued that high levels of urban resource consumption and waste production are not issues about sustaining cities per se, but reflect concerns over the role of cities in global sustainable development (Satterthwaite 1997). Although partially agreeing, we aim to show that there are also processes within the urban metabolism that threaten urban sustainability itself. In particular we highlight storage processes—water in urban aquifers, heat stored in urban canopy layers, toxic materials in the building stock, and nutrients within urban waste dumps—all of which require careful long-term management. Understanding the changes to such storage terms in some cities may be as important as reducing the sheer magnitudes of inputs and outputs.

The cities studied are metropolitan regions, that is, similar to standard metropolitan statistical areas (SMSAs) in the United States, which often encompass several politically defined cities, that is, cities under the jurisdiction of a local municipal government. These metropolitan regions can generally be regarded as *commutersheds*, although the basis for definition may differ between countries.

The metabolism of cities will be analyzed in terms of four fundamental flows or cycles—those of water, materials, energy, and nutrients. Differences in the cycles may be expected between cities due to age, stage of development (i.e., available technologies), and cultural factors. Other differences, particularly in energy flows, may be associated with climate or with urban population density (table 1). Each of the two objectives will be addressed sequentially within a discussion of the four cycles.

Previous Metabolism Studies

Insights into the changing metabolism of cities can be gained from a few studies from around the world, conducted over several decades. These studies are typically of greater metropolitan areas, including rural or agricultural fringes around urban centers. One of the earliest and most comprehensive studies was that of Brussels, Belgium by the ecologists Duvigneaud and Denaeyer-De Smet (1977), which included quantification of urban biomass and even organic discharges from cats and dogs (figure 1)! In studying the city of Hong Kong, Newcombe and colleagues (1978) were able to determine inflows and outflows of construction materials and finished goods. An update of the Hong Kong study by Warren-Rhodes and Koenig (2001) showed that per capita food, water, and materials consumption had increased by 20%, 40%, and 149%, respectively—from 1971 to 1997. Upward trends in per capita resource inputs and waste outputs were also reported in a study of Sydney, Australia (Newman 1999). Studying a North American city, Sahely and colleagues (2003) found that although most inputs to Toronto's metabolism were constant or increasing, some per capita outputs, notably residential solid waste, had decreased between 1987 and 1999. Metabolism studies of other cities include those for Tokyo (Hanya and Ambe 1976), Vienna (Hendriks et al. 2000), Greater London (Chartered Institute of Wastes Management 2002), Cape Town (Gasson 2002), and part of the Swiss Lowlands (Baccini 1997). Some studies have quantified urban metabolism less comprehensively than others. Bohle (1994) considers the urban metabolism perspective on urban food systems in developing countries. A few studies of nutrient flows in urban systems have been undertaken (Nilson 1995; Björklund et al. 1999, Baker et al. 2001), whereas others have specifically investigated metals or plastics in the urban metabolism. Collectively these metabolism studies provide a quantitative appraisal of the different ways that cities are changing worldwide.

Related to the urban metabolism concept is the application of the ecological footprint technique to cities (Wackernagel and Rees 1995). The ecological footprint of a city is the amount of biologically productive area required to provide its natural resources and to assimilate its wastes. Published studies include those for Vancouver (Wackernagel and Rees 1995), Santiago de Chile (Wackernagel 1998), Cardiff (Collins et al. 2006), and cities of the Baltic region of Europe (Folke et al. 1997). The equivalent areas of ecosystems for sustaining cities are typically one or two orders of magnitude greater than the areas of the cities themselves.

Water

In terms of sheer mass, water is by far the largest component of urban metabolism. Wolman's calculations for the 1960s for a one-million-person U.S. city estimated the input of water at 625,000 tonnes[1] per day compared to just 9,500 tonnes of fuel and 2,000 tonnes of food. Most of this inflow is discharged as wastewater, with the remainder being lost by activities such as the watering of lawns. Data from the cities in table 1 show that wastewater represents between 75% and 100% of the mass of water inflow (figure 2a). The six studies since 1990 typically have higher per capita water inputs than the four

Table 1 Characteristics of cities (metropolitan areas) where urban metabolism studies have been conducted

City	Reference	Year	Population (million)	Urbanized density (cap/km^2)	Location (geographic coordinates)	Altitude (m)	Av. temperature (°C)	
							Summer	Winter
U.S. typical	Wolman 1965	1965	1.0	—	—	—	—	—
Brussels	Duvigneaud and Denaeyer-De Smet 1977	1970s	1.075	6,640	50° N 4° E	76	16.4	4.2
Tokyo	Hanya and Ambe 1976	1970	11.513	6,632	35° N 139° E	17	24.3	6.9
Hong Kong	Newcombe and colleagues 1978	1971	3.940		22° N	88	28.0	17.0
	Warren-Rhodes and Koenig 2001	1997	7.0	58,000	114° E			
Sydney	Newman 1999	1970	2.79		33° S	0	22.3	13
		1990	3.657	2,000	151° E			
Toronto	Sahely and colleagues 2003	1987	4.038		43° N	105	19.9	−1.8
		1999	5.071	2,920	79° W			
Vienna	Hendriks and colleagues 2000	1990s	1.5	3,710	48° N 16° E	170	18.5	1.8
London	Chartered Institute of Wastes Management 2002	2000	7.4	6,730	51° N	14	16.7	4.9
	White 2003	2000			0° W			
Cape Town	Gasson 2002	2000	3.0	3,900	34° S 18° E	6	20.8	13.5

Note: cap = capita. One square kilometer (km^2, SI) = 100 hectares (ha) ≈ 0.386 square miles ≈ 247 acres. One meter (m, SI) ≈ 3.28 feet (ft). (Celsius temperature [°C] * 9/5) + 32 = Fahrenheit temperature [°F].

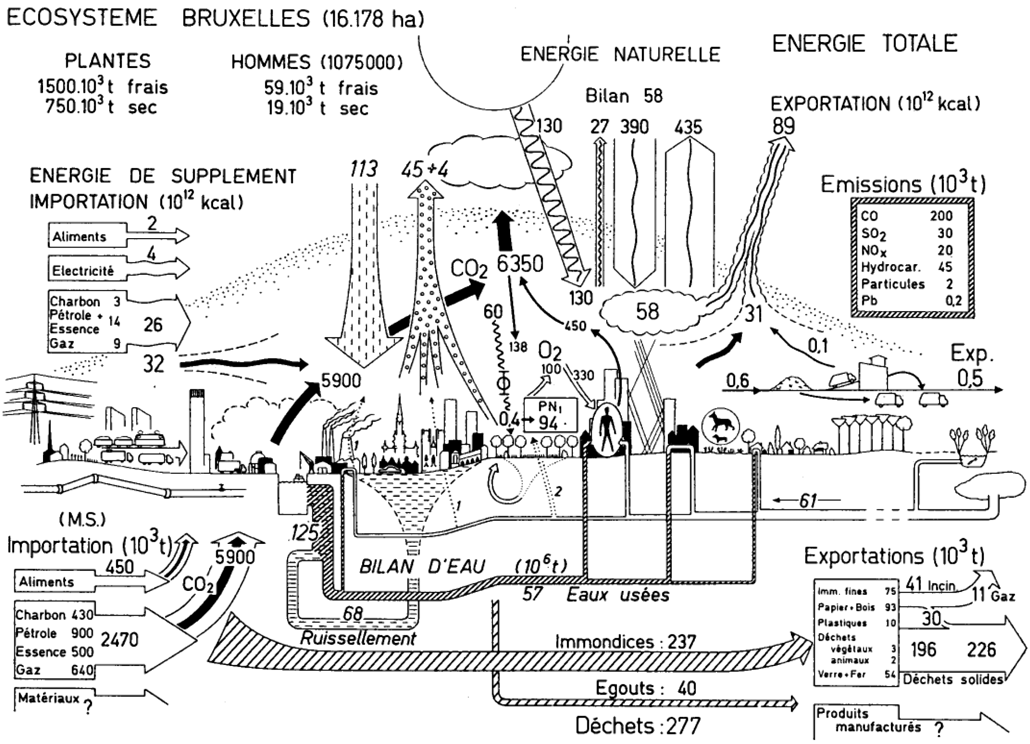
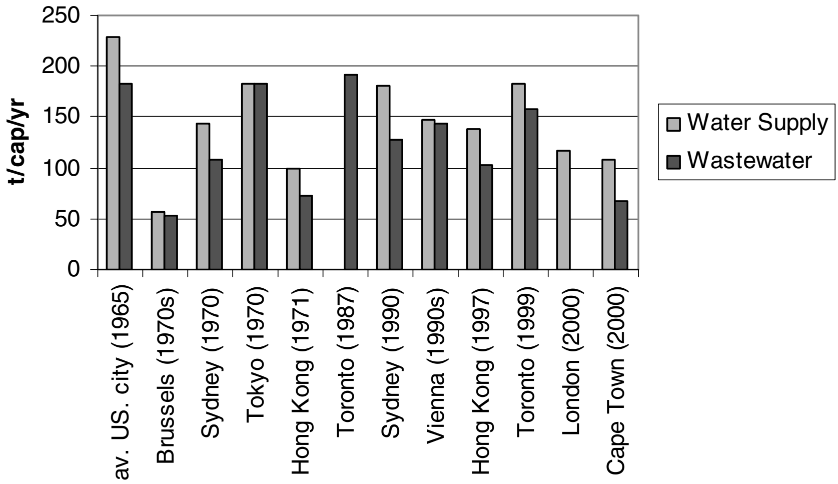


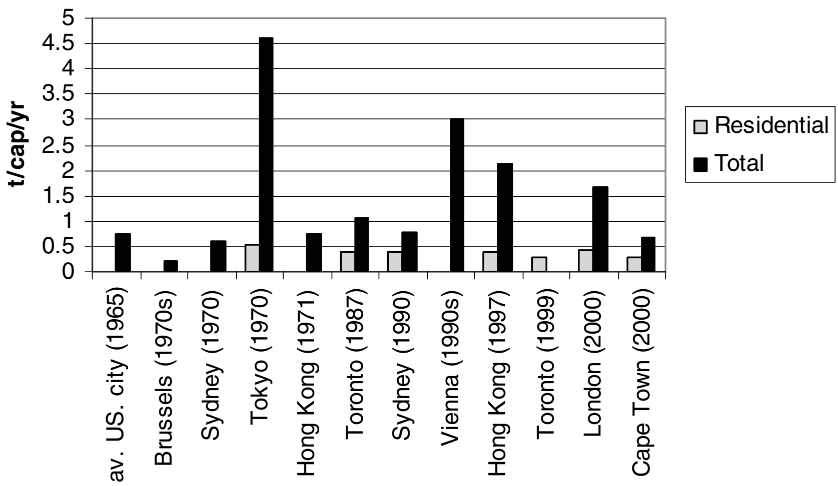
Figure 1 The urban metabolism of Brussels, Belgium in the early 1970s. *Source:* Duvigneaud and Denaeyer-De Smet 1977.

from the early 1970s, although Tokyo in 1970 and London and Cape Town in 2000 are significant exceptions. In the case of Toronto, per capita water use slightly declined over the 1990s, due largely to a reduction in industrial consumption. Nevertheless, none of the cities has quite reached the level of water use of the average U.S. city reported by Wolman.

The impacts of water on the sustainability of cities have further dimensions, beyond the crucial need for inhabitants to have a safe, reliable water supply. For those that rely on ground water, the long-term relationship between cities and their ground water environments is illustrative. As cities evolve from small settlements they typically go through several stages of development in which the relationship with an underlying aquifer changes (Foster et al. 1998). In the early settlement stage, water supply is obtained from shallow urban wells and boreholes, and wastewater and drainage waters are discharged to the ground or to a watercourse (figure 3a). As the settlement grows into a small city the underlying water table falls with increased extraction, so deeper wells are drilled. Overexploitation of ground water often occurs. Moreover, as a result of urban activities, often including continued discharge of wastewater to the ground, the shallow ground water in the city center becomes polluted. Subsidence may begin to occur, depending on soil characteristics. The paving of land surfaces and the growth of drainage systems also begin to have a discernible impact on the ground water system (figure 3b). As the city expands further and matures, a turnaround can sometimes occur. With widespread contamination of the aquifer below the city, or a movement of heavy, water-extracting industry away from city centers, pumping of the urban aquifer ceases. The city now relies on periurban well fields or expensive water imports from distant sources. With the cessation of pumping, the water table below the city begins to rise. Because of changes in the surface recharge, the water table may rise above that under virgin conditions, potentially causing flooding or damage to infrastructure (figure 3c). In this evolving



(a)

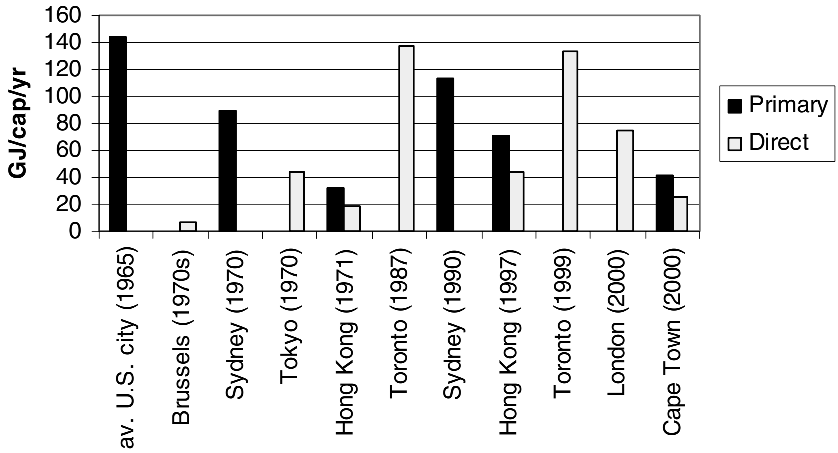


(b)

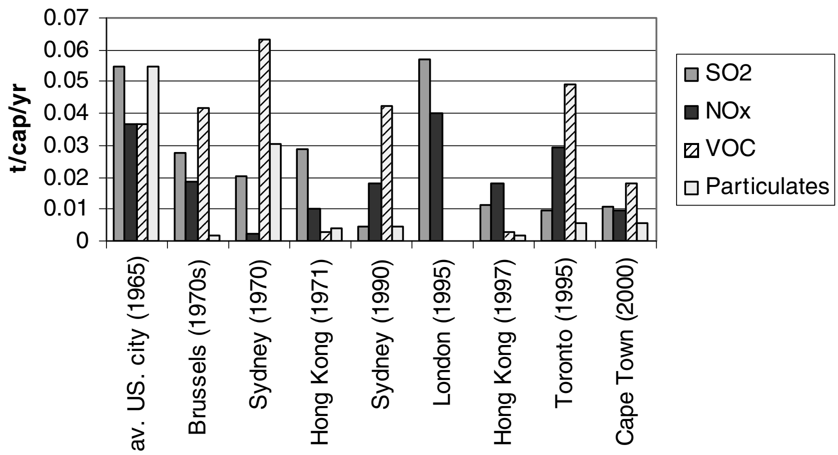
Figure 2 Metabolism parameters from selected cities. (a) Fresh water inputs and wastewater releases. (b) Solid waste disposal. (c) Energy inputs. (d) Contaminant emissions. Missing bars indicate data unavailable. Date refers to the date of measurement; t refers to tonnes. For sources see table 1. The data behind this figure are available in an electronic supplement (e-supplement) on the *JIE* Web site. One tonne (t) = 10^3 kilograms (kg, SI) ≈ 1.102 short tons. One gigajoule (GJ) = 10^9 joules (J, SI) ≈ 2.39 × 10^5 kilocalories (kcal) ≈ 9.48 × 10^5 British Thermal Units (BTU). SO_2 = sulfur dioxide; NO_x = nitrogen oxides; VOC = volatile organic compounds.

process cities can go from exploiting ground water resources to potentially being flooded by them.

Problems of overexploiting ground water have occurred in many cities. One example is Beijing, China, where the water table dropped by 45 m between 1950 and 1990 (Chang 1998). In coastal cities, overpumping has caused salt water intrusion, threatening ground water supplies; examples include Gothenburg, Perth, Manila, and Jakarta (Volker and Henry 1988). Decades of lowering of water tables in Mexico City have caused land subsidence of 7.5 m in the center of



(c)



(d)

Figure 2 Continued

the city, altering surface drainage and damaging infrastructure (Ortega-Guerrero et al. 1993).

Concerns over the potential impact of rising water tables applies to at least one of the study cities. Because industrial withdrawals decreased in London during the 1960s, the water table in an underlying chalk aquifer has been rising at a rate of 1 to 2.5 m per year under central London (Cox 1994; Castro and Swyngendow 2000). Moreover, leakage from London's water distribution systems, estimated to make up 28% of the total water supply, may be adding to this rise in the water table. Where the water table will reach equilibrium is an open question, especially given the change in surface features and local climate over the past decades.

Rising water tables have also been recorded for several cities in the Middle East: Riyadh, Jeddah, Damman, Kuwait, Al-Ajman, Beirut, Cairo, and Karachi (Abu-Rizaiza 1999). In many cases the cause is subterranean discharge of wastewater flows, where there is no surface water outlet. Such rising of ground water levels threatens urban infrastructure, including basements, foundations, tunnels, and other subsurface pipes and cables. Resulting cracks in columns, beams, and walls were reported in Riyadh, Saudi Arabia. In many respects the physical integrity of cities depends

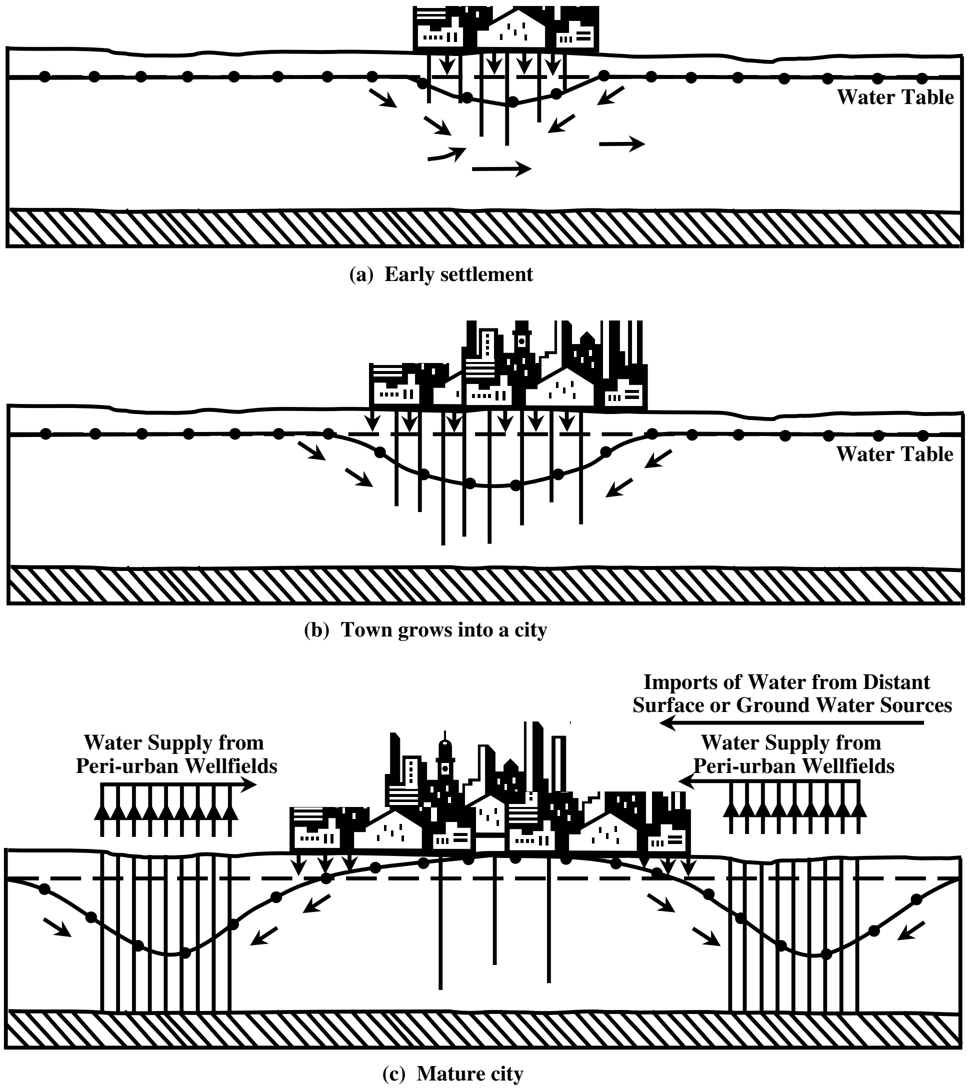


Figure 3 Evolution of water supply and wastewater disposal for a typical city underlain by a shallow aquifer. The arrows indicate water flows; the line with dots indicates the water table. *Source:* Adapted from Figure 1.2 of Foster et al. 1998.

on achieving equilibrium in the ground water component of the urban metabolism.

Materials

Material inputs to cities are generally less well quantified than water inputs, despite their significance to urban infrastructure. Detailed analyses, however, have been conducted for Hong Kong and Vienna. Material flow analyses have also been conducted for Hamburg and a few other European cities (Hammer et al. 2003). In 1997, Hong Kong's daily material inputs included 363 tonnes of glass; 3,390 tonnes of plastics; 9,822 tonnes of cement; 2,095 tonnes of wood; 7,240 tonnes of iron and steel; and 2,768 tonnes of paper. Relative to 1971, inputs of plastics had risen by 400%; iron and steel increased by close to 300%; cement and paper inputs were both up by 275%; and wood inputs had more than doubled. In comparison, the population only rose by 78%. The historical trend in the construction material input to Vienna is also upward. In most decades prior to the 1970s, the per capita use of construction materials was 0.1 m^3/yr (the 1930s being a significant exception at 1.4 m^3/cap/yr).[2] The volumes of input in the 1970s, 1980s, and 1990s, however, were 2.4, 3.3, and 4.3 m^3/cap/yr, respectively (Brunner and Rechberger 2004). Upward per capita trends, for the period 1992 and 2001, were also estimated for direct material input and domestic material consumption for the City of Hamburg (Hammer et al. 2003). This evidence suggests that cities are becoming increasingly material-intensive.

Measures of solid waste outputs are available for most of the cities in table 1, although these data need careful interpretation. Where cities have introduced significant recycling schemes, the disposal of residential household waste may be declining in per capita terms. For example, in Toronto, per capita residential waste declined by 27% between 1987 and 1999. If other waste streams such as the commercial and industrial are included, however, the overall trend may well be upward; three of the cities studied in the 1990s have total waste outputs greater than 1.5 t/cap/yr, whereas all cities studied in the 1970s, except Tokyo, had outputs below 1 t/cap/yr (figure 2b). Some caution has to be taken in interpreting material outputs. Construction waste is often the largest solid waste component (e.g., 57% for Tokyo), but because much of it is considered inert and can be recycled or used as fill, it may not appear in calculations of total waste outputs.

In Vienna, where some of the most extensive material flow studies have been conducted, the production of "holes" from mining of aggregates by far exceeds the city's output of waste materials. Construction materials, of course, go into the building stock and typically remain within the urban fabric for many decades. In Vienna the stock of materials in the buildings and infrastructure is estimated to be 350 t/cap. This stock is clearly growing: the input of construction materials and consumer products is on the order of 12 to 18 t/cap/yr, whereas solid waste is only 3 t/cap/yr. Taking a slightly different perspective, it is evident that the production of holes due to the excavation of construction materials by far exceeds the rate at which they are backfilled. The cumulative hole in the vicinity of Vienna due to excavation of materials since 1880 is estimated to be over 200 million m^3, and growing rapidly (Brunner and Rechberger 2004).

A dimension of material flows that impacts the sustainability of a city is the distance over which materials are transported. As cities grow and transportation infrastructure develops, raw materials seemingly travel increasingly long distances into cities. During the first half of the twentieth century, mineral aggregates used by the construction industry in Toronto came from quarries within a few kilometers of the city. By 1970, the aggregates were traveling an average of 160 km from various counties around the Toronto region (figure 4), the former sources having been exhausted or consumed within the urban area. Douglas and colleagues (2002) describe similar effects during the growth of Manchester, England, since the industrial revolution. In some respects the city is like a plant stretching its roots out further and further until its resource needs are met—a concept that parallels the ecological footprint. One aspect of this growth is that cities require greater expenditure of transportation energy as materials travel from increasing distances.

Energy

The quantification of urban energy fluxes as a component of urban metabolism has varied in depth and breadth between studies. One of the most comprehensive was that of Brussels, for which both natural and anthropogenic energy sources were quantified. Most other studies have focused directly on anthropogenic sources, neglecting net all-wave radiation and heat transfer due to evapotranspiration, local advection, soil conduction, and mass water transfer. The importance of urban heat islands, discussed below, however, indicates the significance of incorporating anthropogenic sources into the natural surface energy balance of cities.

Comparisons of the anthropogenic energy inputs for the cities of table 1 need to distinguish between the direct energy consumed and the primary energy consumption, which includes energy losses in the production of electricity (figure 2c). With its cold winters, and fairly warm summers, Toronto has the highest per capita

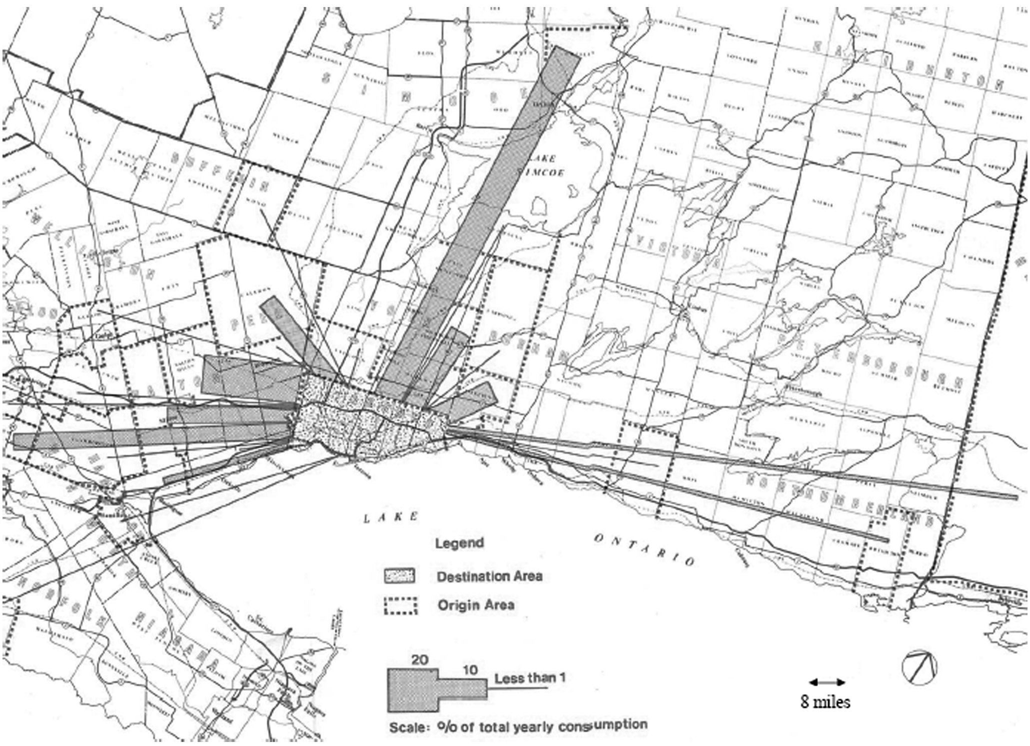


Figure 4 The distribution of mineral aggregate sources for Metropolitan Toronto in 1972. Metropolitan Toronto, in the center of the figure, is the destination of all flow shown. *Source:* Ontario Ministry of Natural Resources.

energy use of the cities. London and Sydney use significantly more energy per capita than Hong Kong, potentially due to its warm winters and higher urban density. A further interesting contrast is that between London in 2000 and Brussels in the early 1970s. Both cities lie at approximately the same latitude and have similar climates and almost identical population densities, yet per capita consumption in modern-day London is an order of magnitude higher. Upward trends in energy consumption since the 1970s are also evident for Sydney and Hong Kong. A minor reduction has been experienced in Toronto since 1987, perhaps due to changing industrial demands and increasing efficiency. With the exception of Toronto, none of the other cities have reached the energy consumption levels of Wolman's average U.S. city.

The relationship between transportation energy demand and urban form has been widely studied, but with various conclusions. Transportation accounts for a significant proportion of urban anthropogenic energy. Among the most significant and highly debated findings are those of Newman and Kenworthy (1991), which illustrate per capita transportation energy consumption decreasing as population density increases (figure 5). All of the cities from table 1 except Cape Town were included in the study, which used data from the 1980s. Hong Kong is the prime example of a dense city with low transportation energy; the European trio Brussels, London, and Vienna are less energy-intensive than the newer cities of Sydney and Toronto. Although the premise that urban form influences transportation energy has been criticized, most notably by Gordon and Richardson (1989), studies have demonstrated that although population density may not in itself contribute to explaining transportation demand, distance from the central business district and other employment centers does (Miller and Ibrahim 1998). As such,

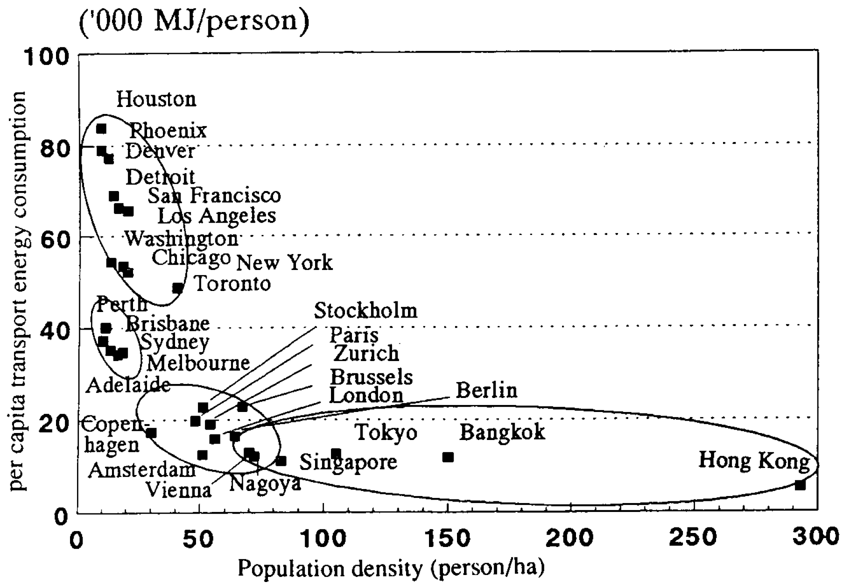


Figure 5 Variation in annual transportation energy consumption and population density among several global cities during the 1980s. '000 MJ = one thousand megajoules = one gigajoule (GJ) = 10^9 joules (J, SI) ≈ 2.39 × 10^5 kilocalories (kcal) ≈ 9.48 × 10^5 British Thermal Units (BTU). One hectare (ha) = 0.01 square kilometers (km^2, SI) ≈ 0.00386 square miles ≈ 2.47 acres. *Source:* Newman and Kenworthy 1991. Reprinted with permission.

the hypothesis linking transportation demand, energy consumption, and urban spatial structure is valid. The higher per capita energy consumption reflected in the urban metabolism of North American cities may thus be explained at least in part by their expansive urban form.

A further aspect of the urban energy balance influencing sustainability is the urban heat island. Most mid- and high-latitude cities exhibit high urban air temperatures relative to their rural surroundings, particularly during the evening (Taha 1997). On a calm clear night after a sunny windless day, elevated temperatures at the canopy layer can be as much as 10°C different in large cities. This urban heat island has been identified as a consistent phenomenon by urban climatologists (Landsberg 1981; Oke et al. 1991; Oke 1995). Much of the heat island effect is due to the built form distorting the natural energy balance—rooftops and paved surfaces absorb heat and reduce evaporation, whereas street canyons and other microscale effects can act as heat traps. An interesting issue, however, is the relative importance of anthropogenic energy inputs in elevating temperatures. All of the energy that is pumped into cities will eventually turn into heat. Estimated anthropogenic contributions range from 16 W/m^2 for St. Louis to 159 W/m^2 for Manhattan (Taha 1997).[3] Santamouris (2001) summarizes a collection of studies, some of which present contradictory findings, or at least highlight the importance of city-specific features.

Increases in temperature directly impact summer cooling loads, thus introducing a potentially cyclic effect on energy demand. For U.S. cities with populations greater than 100,000, peak electricity loads increase by about 1% for every degree Celsius increase in temperature (Santamouris 2001). For high ambient temperatures, utility loads in Los Angeles have demonstrated a net rate of increase of 167 megawatts (MW)[4] per °C. In Toronto, a 1°C increase on summer days corresponds to roughly a 1.6% increase in peak electricity demand. Hot summer days are critical in Toronto in that they cause the highest electricity demand throughout the year. Such demands are often met through increased coal-generated power, elevating emissions of greenhouse gases and other air pollutants. Thus to the extent that anthropogenic energy contributes to the summertime heat island, there is a small positive feedback loop raising both temperature and contaminant emissions.

As air contaminants are largely associated with energy production or utilization, it is perhaps appropriate to analyze them with the energy cycle. Unlike air quality, which can be directly measured, contaminant emissions can be difficult to quantify due to their wide range of sources within an urban region. With the exception of some unpublished values for London in 1995, per capita sulfur dioxide (SO_2) and nitrogen oxides (NO_x) emissions in the group of cities (appearing in figure 2d) are all lower than for Wolman's U.S. city. The data from Hong Kong and Sydney show that although SO_2 emissions are decreasing, NO_x has increased since the 1970s. Emissions of volatile organic compounds (VOCs) are fairly similar for Toronto in 1997, Sydney in 1990, Brussels in the early 1970s, and the average U.S. city in 1965; the highest values seen were those for Sydney in 1970, whereas Hong Kong has had consistently low emissions of VOCs. Particulate emissions reported in the past two decades are significantly lower than the 0.05 t/cap for the average 1965 U.S. city.

As urban air quality is a concern, it may be more appropriate to express emission intensities in kg per unit area, rather than kg per capita, as in figure 2d. (The data behind figure 2 are included in an e-supplement for interested readers.[5]) Note that cities can also be subject to external sources of air pollution.

Greenhouse gases are a further form of air pollutant of concern for global sustainability. Carbon dioxide (CO_2) emissions from the group of cities correspond quite closely with energy inputs. Emissions from Toronto are estimated to be around 14 t/cap/yr, followed by Sydney at close to 9 t/cap/yr. Yet there is still quite a high level of uncertainty in urban greenhouse gas emissions; for example, CO_2 emissions reported for London in the late 1990s range from 5.5 to 8.5 t/cap/yr. Overall, the emissions of contaminants remain quite closely tied to anthropogenic energy emissions; but this link could weaken as cities learn to exploit renewable energy sources, for example, using photovoltaics, solar water heating, or energy recovery from wastewater.

Nutrients

Understanding the flow of nutrients through the urban system is vital to successful nutrient management strategies and urban sustainability. Consequences of improper management include eutrophication of water bodies, release of heavy metals onto agricultural lands, acid rain, and groundwater pollution (Nilson 1995; Baker et al. 2001). The Hong Kong metabolism study (Warren-Rhodes and Koenig 2001) included an analysis of key nutrients: nitrogen and phosphorus. A nitrogen balance for the Central Arizona–Phoenix (CAP) ecosystem (Baker et al. 2001), a phosphorus budget for the Swedish municipality of Gävle (Nilson 1995), and a study of Bangkok (Faerge et al. 2001) also provide insight into the flow of nutrients through urban systems.

The studies reveal the extent to which natural nutrient flows are altered in human-dominated ecosystems. In the CAP and Gävle regions, approximately 90% of nitrogen and phosphorus inputs are human-mediated. Whereas the majority of phosphorus fluxes are related to human food production, import, and consumption (agricultural production and food import), nitrogen fluxes in urban systems are becoming increasingly linked to combustion processes (Björklund et al. 1999, Baker et al. 2001). In the CAP region (which includes agricultural land and desert in addition to urban Phoenix), fixation from NO_x emissions from combustion is the single largest nitrogen input, greater than nitrogen inputs from commercial fertilizers applied to both crops and landscapes! In Hong Kong, 42% of the nitrogen output is NO_x from combustion, which is equivalent to nitrogen outputs from wastewater. Thus, reductions in nitrogen inputs and outputs might be most readily achieved by reducing NO_x emissions (Baker et al. 2001).

In addition to managing nutrient inputs and outputs of the urban system, nutrient storage must also be considered. Nutrients may remain in the urban system through accumulation in soil or groundwater by inadvertent losses or direct disposal or through nutrient recycling, such as the use of food waste for agricultural fertilizer.

Accumulation often results in negative environmental consequences, such as groundwater pollution. All study areas exhibit high levels of nutrient storage. Approximately 60% of all nitrogen and phosphorus inputs to Hong Kong, 60% of phosphorus inputs to the Gävle municipality, 20% of nitrogen inputs to the CAP region, and 51% of phosphorus (but only 3% of nitrogen) in Bangkok do not leave the system. Although a small amount of these nutrients are recycled, the majority are accumulated in municipal and industrial waste sites, agricultural soils, and groundwater pools (Nilson 1995; Baker et al. 2001; Warren-Rhodes and Koenig 2001).

The relatively low levels of nutrient recycling practiced in these study areas highlight the lack of synergy that exists between urban centers and their hinterlands. Girardet (1992) suggests that for cities to be sustainable from a nutrient perspective, they must practice fertility exchange, in which the nutrients in urban sewage are returned to local farmland. This relationship between the city and its hinterlands requires adequate sewage treatment and an appropriate means of sewage transportation, as well as a good supply of local agriculture. The process of urban–rural nutrient recycling was prevalent in the United States during the nineteenth century, until the development of the modern fertilizer industry (Wines 1985). Similar processes existed more recently in China, where 14 of the country's 15 largest cities were largely self-sufficient in food, supplying a majority of their food requirements from agricultural suburbs, which were kept fertile using treated human waste (Girardet 1992); such practices have since changed. In contrast, Hong Kong produces only 5% of its food needs locally, and human food wastes, which used to be recycled as bone meal fertilizers, are now disposed of in landfills (Warren-Rhodes and Koenig 2001). Given that the Hong Kong model is more representative of most modern cities than the former Chinese case, what can be done to improve urban sustainability from a nutrient perspective?

It is not clear that the fertility exchange between a city and its hinterlands described by Girardet (1992) is an appropriate goal for the modern city. Traditionally, there was a symbiotic relationship between cities and their surrounding rural areas, but this relationship has been significantly weakened by modern changes in transportation technology and access to global markets. As cities grow and sewage treatment becomes more centralized, the costs of transporting sewage sludge to local agricultural lands become more prohibitive. Moreover, pharmaceuticals and other toxics present in wastewater may make sludge recycling hazardous to health without expensive treatment. These challenges may make other sewage resource recovery alternatives, such as energy and water reclamation, more attractive for implementation.

The extent to which a city should be dependent on its hinterlands to maintain a sustainable food supply is a difficult question. Many cities obtain their food from continental or global networks of suppliers. For example, 81% of London's 6.9 million tonnes of annual food is imported from outside the United Kingdom (Chartered Institute of Wastes Management 2002). Clearly, cities have grown away from dependence on the surrounding landscape. The relationship of mutual dependence between the city and its hinterlands described by von Thünen's (1966 [originally published 1826]) "isolated state" no longer holds. Peet (1969) shows how the average distance of British agricultural imports increased from 1,820 miles (from London) in 1831–1835 to 5,880 miles by 1909–1913; he argues that a continental-scale von Thünen-type agricultural system has operated since the nineteenth century. Being part of a continental food network may have economic advantages—and the network as a whole may be more resilient than a single city. But the sustainability of entire continental food webs is itself in question. In the United States for example, agriculture is heavily reliant on fossil fuels; 7.3 units of energy are consumed to produce one unit of food energy; depletion of topsoil exceeds regeneration; and groundwater withdrawals exceed recharge in major agricultural regions (Heller and Keolian 2003). Further research is needed to determine whether more locally grown food production, either within urban areas or periurban surroundings, offers a more sustainable agricultural system.

Even though cities have grown away from dependence on the surrounding landscape, they cannot function in isolation from their natural environment. The most effective nutrient management strategies are those that are specifically tailored to individual ecosystems (Baker et al. 2001). The creation of such strategies requires an understanding of nutrient input, accumulation, and output, which can be acquired from detailed nutrient balances within an urban metabolism study.

Conclusions

With data from metabolism studies in eight cities, spread over five continents and several decades, observation of strong trends would not be expected from this review. Moreover, there are concerns over the commensurability of data from different cities, especially for waste streams. Nevertheless, with three different cities having been analyzed at different times, and other supporting evidence, some sense of how the metabolism of cities is changing can be gleaned. Many of the data suggest that the metabolism of cities is increasing: water and wastewater flows were typically greater for studies in the 1990s than those in the early 1970s; Hamburg, Hong Kong, and Vienna have become more materials-intensive; and energy inputs to Hong Kong and Sydney have increased. However, some signs point to increasing efficiency, for example, per capita energy and water inputs leveling off in Toronto over the 1990s. Other changes are mixed; for example, cities that have implemented large-scale recycling have seen reductions in residential waste disposal in absolute terms, but other waste streams—such as commercial and industrial—may well be on the increase. Similarly, emissions of SO_2 and particulates may have decreased in several cities, whereas other air pollutants such as NO_x have increased. Changes in urban metabolism are quite varied between cities.

Future studies might attempt to identify different classes of urban metabolism. These are perhaps apparent from the transportation energy data, where old European, dense Asian, and New World cities are distinctive. Climate likely has an impact on the type of metabolism; cities with interior continental climates would be expected to expend more energy on winter heating and summer cooling. The cost of energy may also influence consumption. The age of a city, or its stage of development, could be a further factor in the type of urban metabolism.

Beyond concerns over the sheer magnitudes of resource flows into cities, there are more subtle imbalances and feedbacks that threaten sustainable urban development. Access to large quantities of fresh water is clearly vital to sustaining a city, but the difference between the input from and output to surface waters may be as important as the sheer volume of supply. Where a city imports large volumes of water, but releases water into underlying aquifers, whether through leaking water pipes, septic tanks, or other means, changing water tables may threaten the integrity of urban infrastructure. Whether it is water in an urban aquifer, construction materials used as fill, heat stored in rooftops and pavements, or nutrients accumulated in soils or waste sites, these accumulation processes should be understood so that resources can be used appropriately. Further examination of how resources are used and stored within a city may yield some surprises. For example, Brunner and Rechberger (2001) suggest that “Modern cities are material hot spots containing more hazardous materials than most hazardous landfills. . . .” Moreover, *growth*, which is inherently part of a city’s metabolism, implies a change in storage. Thus, in addition to quantifying inputs and outputs, future studies should aim to characterize the storage processes in the urban metabolism.

Beyond scientific interest in the nature of city growth, there are practical reasons for studying urban metabolism. The implications of increasing metabolism, at least with current predominant technologies, are greater loss of farmland, forests, and species diversity, plus more traffic and more pollution. In short, cities will have even larger ecological footprints.

The vitality of cities depends on spatial relationships with surrounding hinterlands and global resource webs. As cities have grown and transportation technologies have changed, resources have traveled greater distances to reach cities. For heavier materials, which are more expensive to transport, the exhaustion of the nearest, most accessible resources may at some point become a constraint on the growth of cities. For many goods, including food, modern cities no longer rely on their hinterlands; rather, they participate in continental and global trading networks. Thus, full evaluation of urban sustainability requires a broad scope of analysis.

Urban policy makers should be encouraged to understand the urban metabolism of their cities. It is practical for them to know if they are using water, energy, materials, and nutrients efficiently, and how this efficiency compares to that of other cities. They must consider to what extent their nearest resources are close to exhaustion and, if necessary, appropriate strategies to slow exploitation. It is apparent from this review that metabolism data have been established for only a few cities worldwide and there are interpretation issues due to lack of common conventions; there is much more work to be done. Resource accounting and management are typically undertaken at national levels, but such practices may arguably be too broad and miss understanding of the urban driving processes.

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Notes

1. One tonne (t) = 10^3 kilograms (kg, SI) $\approx$ 1.102 short tons.
2. One cubic meter (m^3, SI) $\approx$ 1.31 cubic yards (yd^3).
3. One watt (W, SI) $\approx$ 3.412 British Thermal Units (BTU)/hour $\approx 1.341 \times 10^{-3}$ horsepower (HP). One square meter (m^2, SI) $\approx$ 1.20 square yards (yd^2).
4. One megawatt (MW) = 10^6 watts (W, SI) = 1 megajoule/second (MJ/s) $\approx 56.91 \times 10^3$ British Thermal Units (BTU)/minute.
5. Available at the *JIE* Web site.

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