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An Ecosystem Approach to Understanding Cities: Familiar Foundations and Uncharted Frontiers

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The ecosystem concept has been one of the most useful in ecology, and also has been embraced by managers and the public in general (Likens 1992; Golley 1993). Even though there are disparities between ecologists and nonspecialists on exactly what constitutes an ecosystem, the potential utility of the concept when applied to urban systems where people live and work argues for redoubled efforts to bring the ecological concept of ecosystems, which is based on "systems thinking," into usage in education. We take the stance here that cities can be understood as ecosystems and that the ecosystem concept is highly appropriate to understanding both ecological and social dynamics (and their interactions) in cities. Our charge was to outline the conceptual foundations and explore the intellectual frontiers of urban ecosystem understanding, and to do this by describing what ecosystem ecologists mean by "city as ecosystem" and identifying the appropriate conceptual frameworks and their importance. Thus, our view emphasizes urban ecosystem research, although we will attempt where possible to point out the value of the approach to education. In particular, we will argue that certain key concepts—ecosystem; nutrients; input, output, and retention of materials; energy use; and heterogeneity—can be taught and learned based on material educators have close at hand: the urban ecosystem that surrounds them.

Our objective is to compare traditional ways of understanding ecosystems with the new perspectives that will be required to understand and study cities as ecosystems. We maintain that the ecosystem approach can be used to understand how cities work, how they interact with surrounding local and global ecosystems, and how expected changes in landscapes and regions resulting from increased urbanization will affect the future of Earth's systems. Moreover, we will argue that ecosystem study as we know it is necessary but not sufficient to understand urban ecosystems. Modifications of existing theory and practice will be required.

Ecologists often identify with one of two general approaches to their subject matter: a population-community approach or a process-functional (sometimes referred to as an ecosystem) approach (O'Neill, et al. 1986), although there has been great interest in merging these perspectives (Jones and Lawton 1994). In application to urban environments, one might distinguish between ecology *in* and ecology *of* cities in the same vein, with the appropriate caution that the two contrasts are not strictly analogous (Grimm, et al. 2000). Ecology *of* cities has to do with how aggregated parts sum, that is, how cities or parts thereof process energy or matter relative to their surroundings; whereas ecology *in* cities focuses on how ecological patterns and processes (especially populations and organismal interactions) vary within cities, or differ in cities compared with other environments. Whether one or the other approach applies can change as the scale of interest changes; for example, a study of the ecology *of* a schoolyard (an ecosystem in its own right) may become part of an investigation of ecology *in* a city when the larger scale is of primary interest. In contrast to the preceding chapter, here we adopt a conceptual framework of ecosystem science and use the ecology *of* cities approach. Specifically, we will ask two questions: How is energy use or consumption of a city or parts of a city dependent upon other ecosystems outside the boundaries under consideration? Is the city a source or a sink for nitrogen in the context of its surroundings, and what are the dominant inputs and outputs of this element?

Familiar Foundations: The Ecosystem Approach in Brief

What is an ecosystem? An *ecosystem* is a piece of earth of any size that contains biotic and abiotic elements, and has both intrasystem interactions and interactions with its surroundings. Necessary components of an ecosystem include boundaries, biota, and abiotic elements; ecosystem ecologists concern themselves with fluxes, interactions, and transformations of energy and materials, and controls of these processes. The concept of ecosystem is not free from controversy. The term was first coined in 1935 by English plant ecologist A.G. Tansley who, rejecting earlier notions of the "superorganism" promoted by Clements and Phillips, preferred to consider animals and plants as associations together with the physical factors of their surroundings as "systems" (Ricklefs 1990). Tansley (1935) outlined his concept of the ecosystem as follows:

> The more fundamental conception is, as it seems to me, the whole system (in the sense of physics), including not only the organism-complex, but also the whole complex of physical factors forming what we call the environment of the biome—the habitat factors in the widest sense. Though the organisms may claim our primary interest, when we are trying to think fundamentally we cannot separate them from their special environment, with which they form one physical system.

By the 1950s, the ecosystem concept had widely pervaded ecological thinking. Francis C. Evans (1956) provided this definition of ecosystem:

In its fundamental aspects, an ecosystem involves the circulation, transformation, and accumulation of energy and matter through the medium of living things and their activities. . . . The ecologist . . . is primarily concerned with the quantities of matter and energy that pass through a given ecosystem and with the rates at which they do so.

This emphasis on the cycling of matter and the associated flux of energy is strongly associated today with the process-functional approach. Odum (1989) has further argued that an integral part of the ecosystem concept is a model of an open, thermodynamic nonequilibrium system, with the emphasis on the external environment.

Despite divergences and debates, Tansley's concept is still widely accepted, with the ecosystem having long been recognized as a fundamental organizational unit in ecology and a major structural unit of the biosphere (Krajina 1960). In modern ecology, we can distinguish between the *ecosystem concept* as defined in a widely used textbook (Begon, et al. 1990): "A holistic concept of the plants, the animals habitually associated with them, and all the physical and chemical components of the immediate environment or habitat which together form a recognizable self-contained entity," and an *ecosystem approach* (a particular branch of ecological research that emphasizes energy flow and material cycling and is characterized by systems thinking). Perhaps the fact that the ecosystem is an overarching and organizing concept that can encompass a variety of ideas within it, rather than being a single, coherent, tightly reasoned theory, makes it such a useful ecological paradigm (Kuhn 1962; Burns 1992).

Defining Ecosystem Boundaries, Structure, and Function

Ecosystem ecologists begin their studies of ecosystems by delimiting the *boundaries* of the system of interest. This may be relatively simple (e.g., the shoreline of a lake) or complicated by movements of organisms or materials (e.g., a stream). Alternatively, boundary definition may be accomplished with respect to the purpose of the study (e.g., a field or a forest patch of manageable size). One well-known example of boundary delimitation is that employed in the watershed approach (Likens and Bormann 1995). The watershed ecosystem is the area drained by a particular stream. Boundaries often are defined by identifying a discontinuity in physical, chemical, or biological processes (O'Neill, et al. 1986), and the watershed is a clear example of this method.

Despite the widespread adoption and use of the ecosystem concept, some have argued that it remains diffuse and ambiguous (O'Neill, et al. 1986), in particular because boundaries often are abstract (Sjors 1955; Fredericks 1958). There also has been debate about the question of spatial scale when defining ecosystems. Colinvaux (1973) argued that one could choose any size area, provided it has defined boundaries. Indeed, in landscape ecology the term has been applied across a range of spatial scales: "The ecosystem concept, which includes structure, function, and development, may be applied at any level of spatial scale, from the size of a rabbit dropping, to the planet" (Forman and Godron 1986). One development that may help to resolve this debate is *hierarchy* theory (e.g., Allen and Starr 1982; O'Neill, et al. 1986). In using the term *hierarchy*, ecologists most often are referring to multiple levels or multiple scales of ecological phenomena. Scientists studying the urban ecosystems of Phoenix and Baltimore in the context of two recently initiated, long-term ecological research (LTER) projects have espoused the importance of a hierarchical approach because it is capable of integrating across subject boundaries, as well as across spatial and temporal scales (Grimm, et al. 2000; Zipperer, et al. 2000; Grove, et al. 2002, Chapter 11 in this volume). By examining ecological phenomena in the context of a hierarchy, simultaneous attention to several scales or hierarchical levels is possible.

Once boundaries are established, the *structure* of the ecosystem is described, including the geophysical setting, plant and animal community structure, trophic relationships (i.e., who eats whom), soils and/or sediments, architecture (e.g., the layering of a forest or the shape, height, and arrangement of vegetation clumps in a desert), and storage pools of major elements. Measurement of biomass in different trophic levels, or of carbon storage in soil, plant, and animal matter, are examples of how structure may be quantified. Descriptions of ecosystem structure permit inferences about function or processes, although such inferences must be made with caution, accompanied by appropriate process measurements.

Ecosystem *function* refers to the processes that occur within ecosystems and the net result of those processes for the system as a whole. Questions that address function include: What are the key players in ecosystem processes? What factors control their rates? What diversity of processes is represented in the ecosystem? The two main elements of ecosystem function on which ecologists have focused their efforts are energy flow and material cycling, which in any ecosystem are governed by the laws of thermodynamics. In the realm of energy flow, for example, ecosystem ecologists measure rates of primary production (i.e., photosynthesis) or respiration, or secondary production of consumer organisms. Specific nutrient transformations within ecosystems, fluxes of materials across ecosystem boundaries, or retention of materials (i.e., the difference between inputs and outputs) may be the focus of material cycling studies.

In most early work on ecosystems, the system was viewed as spatially homogeneous, that is, as a "well-mixed reactor". Emergence of the field of landscape ecology, and integration of some of the ideas of landscape ecology into ecosystem studies, have changed this view. Landscape ecology focuses on patterns in heterogeneous tracts of land, and asks questions about both the causes and origins of those patterns and their consequences for processes (Turner 1989). Forman and Godron (1986) chose to distinguish between ecosystems and landscapes on the basis of a homogeneity criterion: "Although one may apply the ecosystem concept to a heterogeneous region, landscape, or landscape fragment, in this volume we basically limit its use to relatively homogeneous areas within a landscape." In this chapter, we adopt the position that the ecosystem approach is applicable both to a well-mixed reactor model and one that views the ecosystem as a more heterogeneous assemblage of parts or patches. The parts and patches themselves, for example, upland forest, riparian zone, stream, or wetland, might be viewed as ecosystems within larger ecosystems (watersheds). Thus, an ecosystem and its component parts could be treated as well-mixed reactors at some scales and heterogeneous systems at others. *Input–output budgets*, which ask whether an ecosystem retains (inputs > outputs) or releases (inputs < outputs) materials, are built on the well-mixed reactor model but also can be applied to different parts of a complex ecosystem and hence can yield information about *spatial heterogeneity* in material retention.

The Uncharted Frontiers of Urban Ecosystems

From this familiar ground, there are challenges at every step in applying the ecosystem approach to cities. For example, consider the structure of an ecosystem: A forest's architecture is a function of the growth forms of the mix of tree species that make up the forest and how they are constrained by topography, climate, soil fertility, and so forth. A city's structure is largely built and often designed. Even the "natural" components (e.g., trees in parks and in front and backyards) are subject to modification, rearrangement, and conscious or accidental design by humans. How can we apply a simple and elegant concept like the watershed to delineate urban ecosystems when flowpaths may be altered to such an extent as to be unrecognizable by conventional ecological techniques? Are urban streams so modified that they can no longer be reasonably compared with their "natural" counterparts using conventional ecological theory? If so, what changes in theory will be necessary?

The expansion of ecosystem research into ever more human-dominated environments, and in particular to cities as one extreme on a continuum from "pristine" to human-managed or human-defined ecosystems, represents an important test for the generality of the ecosystem concept itself. In some ways cities are like any other ecosystem: (1) the number of species, species diversity, and the number and types of species guilds is probably comparable to, or perhaps even higher than, surrounding ecosystems; (2) soils represent major storage pools of nitrogen and carbon relative to inputs; and (3) primary productivity (rate of photosynthesis), except in the most intensely urbanized parts of a city, is probably not appreciably different than it is in other ecosystems in the region. The following attributes, however, make cities unique: (1) they are heterotrophic (primary production <<< respiration) and extremely energy intensive; (2) they therefore require large inputs of energy and materials—the relative importance of external inputs to internal production and recycling is very high compared with all other types of ecosystems; (3) they produce copious amounts of waste compared with most ecosystems and often lack effective assimilation mechanisms to handle these wastes (or strain existing ones); (4) urban ecosystem function is controlled not just by biophysical factors but also by social and political forces (although this type of control now affects most ecosystems to some extent, it affects cities in a profound manner); and (5) one keystone species—humans—exerts overwhelming control on ecosystem processes. Because of these features, the study of urban ecosystems is likely to provide insights that will lead to refinement of many aspects of ecosystem theory.

As extreme examples of human-dominated ecosystems, can cities be defined adequately by physical attributes (e.g., landscape pattern), population densities, functional attributes (e.g., energy inputs per unit area), or some combination of these traditional variables? Do urban environments necessitate the development of an entirely different conception of the ecosystem, capable of integrating not only ecological but also socioeconomic, political, and cultural factors (Boyden 1977; Redman 1999)? Development of such integrated conceptual models in application to urban and other human-dominated ecosystems is proceeding on numerous fronts (Costanza 1996; Pickett, et al. 1997; Carpenter, et al. 1999; Grimm, et al. 2000), building upon a few notable urban investigations of the 1970s and 1980s that used an ecosystem approach (Boyden 1977; Boyden, et al. 1981), but a synthesis of this information is beyond the scope and intent of this chapter. We will identify here some of the obvious modifications that are needed to understand urban ecosystem boundaries, structure, and function, providing examples from our early experience in the Central Arizona–Phoenix (CAP) ecosystem.

Urban Ecosystem Boundaries

What are the boundaries of a city? Some features of urban boundaries are distinct and easily defined (e.g., the sharp edges that delineate new housing developments from desert in the Phoenix metropolitan area) (Figure 7.1). In other cases, where cities end and suburbs or rural lands begin is more difficult to ascertain. The Census Bureau (1995) defined "urban" for the 1990 census as comprising all territory, population, and housing units in urbanized areas and in places of 2,500 or more persons outside urbanized areas. Urbanized areas comprise one or more places ("central place") and the adjacent densely settled surrounding territory ("urban fringe") that together have a minimum of 50,000 persons (U.S. Census 1995). These population-based definitions may make little sense when considering, for

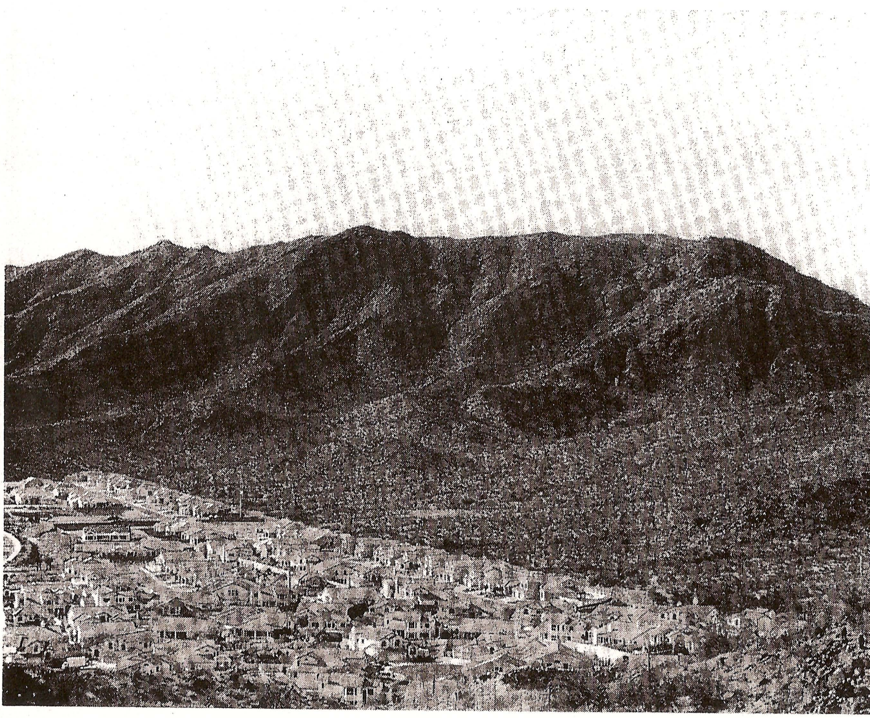

FIGURE 7.1. Phoenix's South Mountain, showing the distinct edge where urban development meets the desert. Photo by Ramón Arrowsmith, used with permission.

example, the mass balance of elements (e.g., the inputs minus outputs of carbon) for an ecosystem, but the most appropriate definition will of course depend on the question being asked in the study. Once the question is identified, however, an important first step is to define boundaries and to be consistent in their application to the question at hand. This may be a challenge when urban boundaries are rapidly changing due to urban growth or human migration patterns. In Phoenix's Maricopa County, the fastest-growing county in the United States, population has doubled twice since 1960, accompanied by an amoebalike spread of urban lands (Figure 7.2). Hence, boundaries of the CAP LTER area have been drawn far outside the current urban fringe, to account for anticipated future expansion. To construct a mass balance for nitrogen (N), for example (see later), the Salt River watershed was used, of which only 25 percent is urban or agricultural land (Baker, et al. 2001). This is akin to the treatment given Vancouver by Boyle and Lavkulich (1997) in their investigation of carbon storage in the lower Fraser River Basin in British Columbia, Canada: The urban ecosystem was subsumed within the larger watershed they considered.

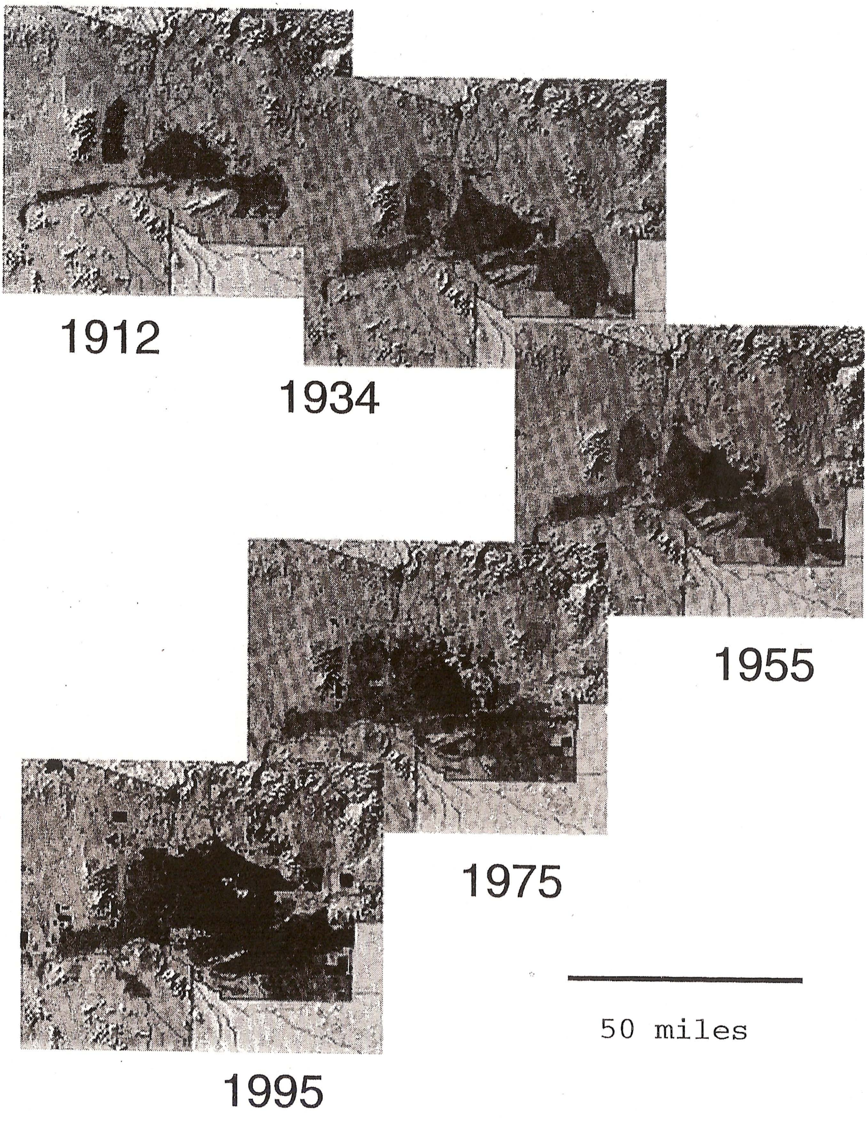

FIGURE 7.2. Changes in land use in the Central Arizona–Phoenix area from 1912 to 1995. Light gray—desert; medium gray—agricultural land; black—urban/suburban land. After Knowles-Yánez, et al. (1999).

Urban Ecosystem Structure

All of the measures of structure for nonurban ecosystems apply in cities, but there are additions; the most obvious of these is the built environment, including houses, buildings, roads, and service infrastructure (e.g., plumbing, wiring, and water delivery systems). These structural elements have characteristics that can influence heat budgets and material storage and transport in urban ecosystems (Newcombe, et al. 1978).

The so-called natural areas of cities, such as greenways, parks, preserves, and lakes and rivers, are often designed and intensively managed. Thus, they have characteristics that are unique in comparison to the rural hinterlands. Although the intention of landscape architects may be to create an environment that bears some resemblance to nonurban environments, often design features are selected because of their particular appeal to people, or because of some additional function that the design performs (such as flood control or retention of storm runoff). In Phoenix, many of the urban greenways and parks seem out of place in an arid environment, yet they share the label of urban "natural" areas with desert parks and preserves.

The demographic structure of the human population is another aspect of urban ecosystem structure that must be included. Such variables as age, sex, racial or ethnic group, and income are quantifiable from databases such as national censuses. Finally, there are unseen elements of structure with which ecologists have little experience: those associated with social institutions and culture. These might include, for instance, the economic system, the political system, cultural structures, and belief systems (Pickett, et al. 1997).

The urban landscape thus contains elements of natural ecosystem structure (species composition, trophic structure, vegetation architecture, soils, water), plus built structure, designed structure, and social structure. A promising way to deal with this complexity is through the landscape ecological approach of defining patches at a range of spatial scales (i.e., defining a patch hierarchy). *Hierarchical patch dynamics* models provide a relatively new way to look at complex systems that change through time (Wu and Loucks 1995). Processes are measured at a specific scale for fundamental units of the landscape at that specific scale. Those fundamental units are called patches, and their structure (sizes, arrangement, types) can be a major determinant of the processes. However, patch structure and arrangement also can change through time; hence, the models are dynamic. Furthermore, because patches can be identified at many scales, the models must also be hierarchical. Hierarchical patch dynamics models thus give us a way to consider patch dynamics simultaneously at multiple scales, rendering the complexity of urban systems more tractable. A fundamental question in both the Central Arizona–Phoenix and Baltimore urban LTER programs is: How does patch structure change with time and how does this in turn influence ecological patterns and the interaction between social and ecological spheres (Grimm, et al. 2000)? To answer this question, both projects are using hierarchical patch dynamics models as an important tool for integrative ecosystem research in urban settings (Zipperer, et al. 2000).

For element mass balances, characterization of flowpaths may serve as a useful measure of ecosystem structure. Much of the flow of water and materials is controlled within human-developed conduits that are often distinctly separated from natural flowpaths. In Phoenix, for example, all of the flow of the Salt River is diverted to canals for human utilization; the bed of the Salt River as it runs through Phoenix is now dry except during flood events (Graf 2000). Municipal water is distributed over the broad metropolitan area, and water that is not evaporated moves through sewer systems, where flows converge at a few large wastewater treatment plants and from there back to the natural river system. Agricultural water also is distributed widely, providing water to crops at a rate about 10 times higher than natural precipitation. Most of the agricultural water that is not evaporated (about 20% of input) enters the groundwater, providing most of the groundwater recharge. To prevent flooding, natural precipitation is collected and diverted to retention basins, where it too recharges aquifers. Thus, the paths of water flow in an urban ecosystem are very different from those of the natural system that preceded it.

Urban Ecosystem Function

Energy flow and nutrient cycling in urban ecosystems must conform to the same thermodynamic laws that apply to any other ecosystem (e.g., energy and matter cannot be created or destroyed, but they are transformed). A suite of social drivers, however, must also be considered and may prove to be as significant to ecosystem function as are biophysical variables. These include institutions and organizations, information flow, and cultural attitudes and perception (Grimm, et al. 2000). Perhaps the most obvious difference between urban and nonurban ecosystems is that urban ecosystems consume vastly more energy than they produce; that is, they are characterized by an extremely high energy expenditure (Newcombe, et al. 1978). Odum (1989) reported that energy consumption in urban–industrial ecosystems exceeds by one to two orders of magnitude that of even human-subsidized agricultural ecosystems. Why is energy expenditure so high in cities? It is high because in addition to plant, microbial, and animal (including human) respiration, cities have a hungry industrial metabolism. That metabolism is supported mainly by imported fossil fuels, as Stephen Pyne et al. (2001; p. 116) points out in a consideration of the ecology of urban fire:

> Modern cities remain fire-driven ecosystems. Fire's influence is everywhere, yet fire is almost everywhere invisible. . . . Cars, trucks, buses, motorcycles, tractors, backhoes, bulldozers, graders, generators, lawnmowers, the urban landscape overflows with a mechanical fauna that feeds on fossil fuels.

All of the human activities that lead to dependence on this imported energy, the economic, governmental, and social institutions that enable procurement of the needed energy, and finally, the connection between quality of life and the use of that energy, are rooted in social factors that fundamentally influence the function (metabolism) of urban ecosystems.

Accompanying this prodigious energy consumption in cities is production of wastes like CO_2, nitrogen oxides, sewage, solid wastes, and water and air pollutants (Boyden and Dovers 1992), in stoichiometric proportion to the materials imported and/or consumed. At the whole ecosystem scale, the magnitude of these flows of energy and matter is certainly a product of the activities of members of the dominant species (humans), but through collective behavior (i.e., the actions of social institutions), the species can modify material cycles beyond the summed total of those individual activities. In Phoenix, for example, the need to store and deliver water to meet human demand in this aridland city led to large-scale manipulation of the two large desert rivers that converge in the Phoenix basin. Manipulations included upstream impoundment for water storage, diversion into canals serving agricultural and municipal needs, and utilization of the river channel for gravel mining operations and as a recipient of treated wastewater. This tremendous alteration of hydrologic routing has no doubt altered the flows of materials in metropolitan Phoenix, and point additions of treated wastewater create a nutrient-enriched riverine system downstream from the city. A prevailing cultural attitude associated with these manipulations was that water development was essential for colonization of the American West (Reisner 1986; Gammage 1999), which led to implementation of policies at both local and national levels in strong support of large water projects such as those that made expansion of agriculture and later, the rapid population growth of Phoenix possible. An analysis of how social factors such as these might have influenced specific characteristics of material transport and cycling represents an important frontier for urban ecosystem understanding. Social factors also can ameliorate pollution effects. Changes in sewage treatment policy, for example, are clearly indicated in the long-term sediment record of Toolonlahti Bay, which receives inputs from the city of Helsinki, Finland (Tikkanen, et al. 1997).

The Ecosystem Approach and Potential Education Applications: Phoenix, Arizona

Energy expenditure in urban ecosystems is such that all cities can be considered to be extremely heterotrophic ecosystems (primary production < respiration), and therefore dependent upon import of energy. One well-known concept is that of the ecological footprint, an index that captures the essence of the dependence of a city on ecosystems outside it. The ecological footprint measures the productive land area required to continually produce all of the energy consumed in an ecosystem, without regard to where on Earth that production occurs (Wackernagel and Rees 1996). The ecological footprint has widespread appeal because of its apparent simplicity and comparability among cities, regions, or nations (see van den

TABLE 7.1. Approximate ecological footprint calculation for Phoenix.

Consumption of	Per cap. land consumption (ha)	Ecological footprint (km^2)
Food	1.55	34,000
Housing	1.06	23,000
Transportation	1.06	23,000
Consumer goods	1.06	23,000
Services	0.40	9,000
TOTAL	5.1	112,000

Source: Data on consumption are from Wackernagel and Rees (1996).
The area of the Central Arizona–Phoenix ecosystem (metropolitan area) is 2,000 km^2, thus, the ecological footprint for this ecosystem is 56 times its size.

Bergh, et al. 1999 for a critique). The ecological footprint of a city like Vancouver, BC, Canada, is 180 times the city area (Wackernagel and Rees 1996), whereas the ecological footprint of a well-known heterotrophic ecosystem, a forest stream (Bear Brook, NH), is just 31 percent of the stream's area (Collins, et al. 2000).

An estimate of the ecological footprint of Phoenix based on per capita data for the U.S. as a whole (Table 7.1) provides an interesting perspective on the dependence of Phoenix on production that occurs elsewhere. Because the current Phoenix metropolis covers approximately 2,000 km^2 of land area, a simple calculation based on published per capita energy use suggests that the ecological footprint may be some 56 times the size of the city itself. This is probably an underestimate, however, because in order to live in the arid southwest, Phoenicians expend vast amounts of energy to cool their homes and businesses and to bring water to the city. In addition, refinements of the ecological footprint concept that ask the question of how much land area is needed to absorb the wastes produced by a given population (e.g., Folke, et al. 1997) can extend the utility of the concept beyond energy consumption considerations (see also Chapter 8 in this volume; Luck, et al. 2001).

The ecological footprint is an heuristically useful tool, and for that reason it has seen widespread use by governments (Toronto, Canada; London, England; and all of the major cities of Australia are some examples, based on a cursory search of the World Wide Web) and has made its way into classroom curricula. One advantage of the approach from a teacher's point of view is its applicability at a range of scales, which can be defined based on social or ecological criteria (Table 7.2). Simple calculation procedures allow students to determine how changes in behavior at the individual, family, or neighborhood level can illustrate which are the critical variables in human energy use. For example, the effects of changes in eating habits might be compared with changes in driving patterns, revealing the much greater energy use (larger footprint) associated with the latter. Comparison of the ecological footprint among cities that differ in their climatic set-

TABLE 7.2. Scales of ecological footprints.

Geopolitical and cultural	Ecological
Individual	Individual
Household	Individual Property
Neighborhood	Land Use/Cover Patch[1]
School District	
City	Watershed
Metropolitan Area	
Region	Region or Large Watershed
Nation	Biome
Continent	Continent

[1] Patches defined on the basis of land use and cover might include residential areas (single family and multifamily), parks and preserves, industrial districts, urban core, agricultural areas, schools or other institutions.

tings (e.g., Phoenix and Baltimore), might show the greater summertime energy demand associated with air conditioning in Phoenix. More sophisticated exercises using the ecological footprint concept would examine the effects of different planning options (e.g., development of a public transportation system vs. freeway construction that encourages automobile use), or the effects of affluence (e.g., comparing industrialized to developing nations) on the dependence of regions or nations on external productivity. The point is that the ecological footprint, being transferable among scales and comparable among different situations, can illustrate the impact of individual human choices at a range of scales.

A Nitrogen Mass Balance for Phoenix

Mass balances are used by ecosystem ecologists to quantify inputs, outputs, and changes in storage pools of elements. In most terrestrial ecosystems, rates of input and output are small compared with rates of internal cycling, whereas in open ecosystems (e.g., streams), fluxes of nutrients across ecosystem boundaries are much larger than nutrient transformations within the ecosystem (e.g., Sprent 1987). A balance sheet of inputs and outputs for natural ecosystems includes atmospheric, hydrologic, and biologic vectors (Likens and Bormann 1995). The simplicity of the watershed approach is that hydrologic inputs are usually absent (i.e., there are no streams entering the watershed) and biologic inputs and outputs often are small; thus, retention can be measured as the difference between atmospheric deposition inputs and streamflow outputs.

Urban ecosystems introduce entirely new categories of inputs and outputs—those associated with human actions (Newcombe 1977). Baker, et al. (2001) constructed a nitrogen budget for the Central Arizona–Phoenix ecosystem that illustrates the dramatic quantitative and qualitative differ-

TABLE 7.3. Inputs and outputs of nitrogen for the CAP ecosystem.

Inputs
Natural Inputs
Atmospheric deposition
Surface water inflows
Biological N_2 fixation (desert vegetation)
Deliberate Human Inputs
Fertilizer
Human food
Animal feed
Fuels
Other imports
Biological N_2 fixation (alfalfa)
Human immigration
Inadvertent Human Inputs
NO_X production by fossil fuel combustion
Outputs
Deliberate Human Outputs
Crop exports
Meat and milk exports
Human emmigration
Inadvertent Outputs
Volatilization and denitrification
Surface water outflows
NO_X export in air
Change in Storage
Deliberate
Landfills
Vegetation (in part)
Built structure
Human population
Inadvertent
Groundwater and vadose zone
Vegetation and soils

ence in fluxes for an urban ecosystem compared with a nonurban ecosystem. One category of inputs is the *deliberate* import of materials, such as the import of nitrogenous fertilizer for agricultural production, which contrasts with natural inputs and outputs (Table 7.3). Furthermore, *inadvertent* inputs and outputs may make up a significant portion of the mass balance. For example, fixation of N_2 by fossil fuel combustion produces NO_X compounds. Even though the fate of this NO_X currently is unknown, it represents a huge input in comparison with the natural inputs of surface water inflow and atmospheric deposition for Phoenix (Figure 7.3), and probably for most cities.

One finding of the mass balance estimate for Phoenix (Figure 7.3; Baker, et al. 2001) is that the city must be accumulating nitrogen at a very high

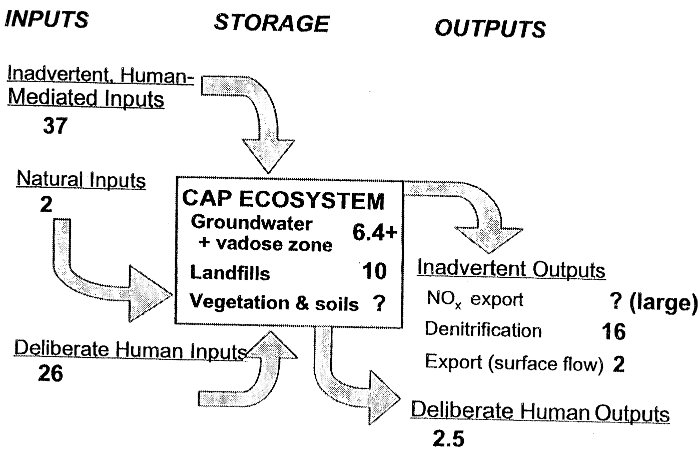

FIGURE 7.3. Nitrogen budget for the CAP ecosystem, showing major categories of inputs, outputs, and change in storage (see Table 7.3 for subcategories). All values in $kg\,N\,ha^{-1}\,y^{-1}$ (redrawn from data in Baker, et al. 2001).

rate (nearly $40\,kg\,ha^{-1}\,y^{-1}$, or $44\,lb\,acre^{-1}\,y^{-1}$). Some of this retention can be accounted for as increases in N stored in groundwater and landfills. The amount of N accumulation in other storage pools (especially vegetation and soils) is not yet known. Even if we do not consider the large, combustion-derived NO_X input, inputs still exceed outputs ($16.6\,kg\,N\,ha^{-1}\,y^{-1}$ or $18.6\,lb\,acre^{-1}\,y^{-1}$, compared with $39.3\,kg\,N\,ha^{-1}\,y^{-1}$ or $44.1\,lb\,acre^{-1}\,y^{-1}$ when NO_X inputs are included).

The history of development of the Phoenix metropolis and associated changes in land use in the region help to explain the causes and implications of the CAP ecosystem's present-day nitrogen accumulation. The Salt and Gila Rivers converge at the site of modern-day Phoenix, and, indeed, these rivers are the environmental feature that allowed establishment of a large ancient civilization (the Hohokam) in this hot, arid region (annual precipitation ~100 mm). The nitrogen budget was constructed for the lower portion of the watershed drained by the Salt River ($12{,}000\,km^2$). Since 1950 the human population has increased from 330,000 to more than 3 million inhabitants, largely due to immigration. Currently, 54 percent of water use in the CAP ecosystem is for irrigated agriculture, with 40 percent to municipal uses and the remainder to industrial uses (AZ Department of Environmental Quality 1994). Surface water from the Salt River supplies 48 percent of this water, whereas 27 percent is from groundwater and 22 percent from the Colorado River via the Central Arizona Project Canal.

Since the early 1900s, agriculture has played a key role in the development of Phoenix and surrounding municipalities (Gammage 1999). Urban expansion occurred largely at the expense of desert until the most recent 20 years, when agricultural land use has begun to decline as residential and commercial uses have expanded (Figure 7.2). As in most areas of the developed world, fertilizer use has increased dramatically since World War II, and, not surprisingly, increases in nitrate concentration have been observed in many groundwater wells in the region: concentration exceeds 10 mg/L nitrate-N throughout most of the area, and is as high as 50 mg/L in some wells. This increase in groundwater N contamination is cause for concern because levels exceeding 10 mg/L are considered a threat to human health. One potential use of the nitrogen mass balance, therefore, is to identify the major sources and accumulation zones for this element, so that changes in policy or behavior can be guided by scientifically based understanding. For example, an understanding of groundwater N accumulation could aid cotton production while at the same time reducing deleterious effects of fertilizer use. Irrigating crops with groundwater (approximately 27 percent of the water use in Phoenix is from groundwater sources, and most of that is used by agriculture) at a concentration of 20 mg nitrate-N/L, when supplied at a rate of 1.5 m/year (typical irrigation rate for cotton), would provide 300 kg N ha^{-1} y^{-1}. This is about 150% of the fertilization requirement (~200 kg N ha^{-1} y^{-1}) for cotton. Nitrate supplied in excess of crop requirements will leach back into aquifers, adding to net accumulation. Overfertilization also costs the cotton farmer, not just because of the expense of fertilizer and fuel, but also because high N levels inhibit boll formation and reduce the cotton crop.

The N mass balance also can be a powerful educational tool. We consider here the impact of a simple change in individual behavior on the nitrogen budget. Everyone who has children can appreciate the difficulty of convincing them not to waste their milk: It seems that the behavior of pouring one's milk down the kitchen sink when Mom or Dad is not looking is indeed one with a long tradition. It is a national problem—about 32 percent of the milk produced in this country is not consumed by humans and is therefore considered to be wasted somewhere between the dairy and a consumer's mouth (Kantor, et al. 1997). What impact would reducing this wastage have on one's personal nitrogen budget?

If milk wastage were reduced to zero, we could produce 32 percent less milk to satisfy our needs. This means that we would have 32 percent fewer cows and 32 percent less cow manure leached to aquifers. It would also mean that we could produce less high-protein grain concentrates to feed the cows, which in turn would mean that we would reduce fertilizer consumption, and therefore fertilizer leaching, by 32 percent. Overall, the effect would be to reduce the amount of N pollution created by a milk-drinking individual by about 2 kg/year (Table 7.4).

For comparison, per capita output of N to sewers in the CAP ecosystem, which includes human waste, detergents, and garbage grinder wastes, is about 7 kg N/year (Lauver and Baker 2000); however, this waste is treated to remove nitrogen before it is discharged to the Salt River channel. In the CAP ecosystem, the overall treatment efficiency for N removal in waste-

TABLE 7.4. Calculating the effect of avoiding milk wastage on an individual's N balance.

Budget term and reference	Current[1]	No waste[1]
Milk consumed (USDA 1999)	3.3	3.3
Wastage, as fraction (Kantor et al. 1997)	0.32	0.00
Milk produced	4.9	3.3
Feed required	13.3	9.0
Manure produced (N in feed that does not become milk)	8.4	5.7
Manure recycled as fertilizer[2]	4.2	2.8
Manure N leached to groundwater	2.1	1.4
N from alfalfa (33% of feed; Ensminger 1993)	5.0	3.4
N from concentrates (rest of feed)	8.2	5.6
Fertilizer (manure + chemical)	16.4	11.2
Chemical fertilizer (total fertilizer minus manure N)	12.3	8.3
Fertilizer N leached to groundwater	4.1	2.8
Total N leached to groundwater	6.2	4.2

[1] Units in kg N cap^{-1} y^{-1} unless otherwise indicated.
[2] Assuming 50% loss by volatilization and leaching.

water is about 75 percent, which means that only 1.75 kg N cap^{-1} y^{-1} actually reaches the river (Lauver and Baker 2000). Thus, if one individual stopped flushing the toilet, taking showers, and washing dishes, the reduction in N output to the environment would be 1.75 kg N/year.

This analysis shows that the apparently trivial action of eliminating retailer and household milk wastage would be more effective at reducing an individual's output of N to the environment (by 2 kg N/year) than would entirely eliminating his or her production of wastewater (1.75 kg N/year). Further analysis of the N budget is likely to reveal other simple and inexpensive methods of reducing N contamination of the environment that would not have been recognized by intuition or targeted by government pollution reduction programs. Moreover, the power of this approach is that it can be developed in the classroom and related directly to pupils' and their families' everyday lives.

Conclusions

We have presented examples from the Central Arizona–Phoenix urban ecosystem, promoting the view that we can apply familiar techniques of ecosystem ecology to cities, just as we would to any ecosystem. Given our charge to explore the intellectual frontiers of urban ecosystem understanding, it may be useful to consider whether our initial approach to mass balance should be modified. In particular, do we need to incorporate models of human behavior or economic drivers? The answer here is probably *yes* because the largest inputs are a consequence of human behaviors (e.g., driving patterns, fixation via combustion) and economics (e.g., imports of food, animal feed, and fuels). Would a different view of ecosystem structure improve our ability to put the mass balance to use in informing policy that will promote environmental protection? More fundamentally, would it improve our ability to predict the major points of production, accumulation, and transformation of nitrogen in the ecosystem? Again, we answer in the affirmative: Understanding how humans have manipulated paths of water flow within cities may be key to unlocking the dynamics of transport and transformation of materials.

We suggest that the next step in understanding urban ecosystems is to begin to incorporate social scientific explanations, controls, and mechanisms into our existing ecosystem models. Just as ecologists learned to speak the language of physical scientists when an understanding of climatic controls and changes was required for ecological explanations, we must now engage in a dialogue and sharing of conceptual models with the social sciences. With our new emphasis on the urban extreme along a spectrum of human-dominated ecosystems, the time is right to develop a more comprehensive ecosystem theory.

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