Chapter II.3

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

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Introduction

The ecosystem concept has been one of the most useful in ecology, and also has been embraced by non-ecologists and the public in general (Likens 1992, Golley 1993). While there are disparities between these two groups 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 ecosystem, which is based on "systems thinking", into usage in education. Here we take the stance 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.

Our objectives are to compare traditional ways of understanding ecosystems with the new perspectives that will be required to understand and study cities as ecosystems. We will explore two examples from ecosystem ecology: ecosystem metabolism and material balances. 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 (Grimm et al. in revision). 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) differ in cities compared with other environments. In contrast to the preceding chapter, here we adopt a conceptual framework of ecosystem science, using the ecology OF cities approach. Specifically, we will ask: How is energy consumption of a city or parts of a city dependent upon other ecosystems outside the boundaries under consideration? and 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 <u>concept</u> 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 <u>approach</u> (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 hold a variety of ideas within it, rather than being a single coherent tightly reasoned theory, makes it such a useful ecological paradigm (Kuhn 1962).

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 (for example, the shoreline of a lake) or complicated by movements of organisms or materials (for example, a stream). Alternatively, boundary definition may be accomplished with respect to purpose of the study (for example, a field or a forest patch of manageable size). One well known example of boundary delimitation is that employed by the watershed approach (Likens and Bormann 1995). The watershed ecosystem is the area drained by a particular stream. Boundaries are often defined by identifying a discontinuity in physicochemical 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 are often 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). A recent development that is relevant to this debate is hierarchy theory (e.g. Allen and Starr 1982; O'Neill et al 1986), whereby independent levels of causation are attributed to each level in a hierarchy of organization, each level having its own scale of space and time. Researchers at the two recent urban additions to LTER's network of sites, Phoenix and Baltimore, have espoused the importance of the hierarchical approach, since it is capable of integrating across subject boundaries, as well as across spatial and temporal scales. Both projects are using hierarchical patch dynamics as an important tool for integrative ecosystem research in urban settings (Zipperer et al. in press).

Once boundaries are established, the structure of the ecosystem is described, including the geophysical setting, plant and animal community structure, trophic relationships, soils and/or sediments, architecture, 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. Often, descriptions of ecosystem structure permit inferences about function or processes, although such inference 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 net primary production 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, 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 pattern of heterogeneous tracts of land, and asks questions about both the cause and origin of that pattern and its 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, 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 can be applied to different parts of the 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. As an example, consider the structure of an ecosystem: a forest's architecture is a function of growth form of the mix of tree species that make up the forest and how that is constrained by topography, climate, edaphic factors, and so forth. A city's structure is built and it is often designed. Even the "natural" components (trees in parks, 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 delineation of 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 not, what changes in theory will be necessary?

The expansion of ecological research into ever more human-dominated environments, and in particular to cities as 'end-member' ecosystems on this continuum, 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 pools of nitrogen and carbon relative to inputs; and 3) primary productivity, except in the mostly intensely urbanized parts of a city, is probably not appreciably different than it is in other ecosystems in the region. But the following attributes make cities unique end-member ecosystems: 1) they are heterotrophic 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 to other types of ecosystems; 3) they produce copious amounts of waste compared to 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, study of urban ecosystems is likely to provide insights that will lead to refinement of many aspects of ecosystem theory. As end-member ecosystems, can cities be adequately defined 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? Or do urban environments necessitate the development of an entirely new conception of the ecosystem, capable of integrating not only ecological but also socioeconomic, political and cultural factors (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. in revision), but a synthesis of this information is beyond the scope and intent of this chapter. Here we will identify 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, such as the sharp edges that delineate new housing developments from desert in the Phoenix metropolitan area (Fig.1). In other cases, where city ends 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 (US Census 1995). These population-based definitions may make little sense when considering, for example, elemental mass balance for an ecosystem, but of course the most appropriate definition will 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 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 U.S., population has doubled twice since 1960 accompanied by an amoeba-like spread of urban lands (Fig. 2). Hence boundaries of the CAP LTER have been drawn far outside the current urban fringe, to account for anticipated future expansion. To construct the N balance, for example (see below), the Salt River watershed was used, of which only 25% is urban or agricultural land (Baker et al. in review). This is akin to the treatment given Vancouver by Boyle & 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.

Urban Ecosystem Structure

All of the measures of structure for non-urban 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.

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. Whereas the intention of landscape architects may be to create an environment that bears some resemblance to non-urban environment, 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 (see, e.g., Pickett et al. 1997).

The urban landscape thus contains elements of natural ecosystem structure (species composition, trophic structure, architecture, soils, water), 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. Initially, these patches are most readily based on some combination of land use and land cover, but different patch structures may be definable for hierarchies of hydrologic, geologic, ecological, economic, political, or other units. Like boundaries, patch structure changes with time. 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 in revision)?

For materials balances, characterization of flow paths 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 in press). 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 is also 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) moves to groundwater, providing most of the groundwater recharge. To prevent flooding, natural precipitation is collected and diverted to detention basins, where it too recharges aquifers. Thus, the flowpaths of 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 as in any other ecosystem. However, a suite of social drivers also must 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. in revision). Perhaps the most obvious difference between urban and non-urban ecosystems is that urban ecosystems consume vastly more energy than they produce; that is, they are characterized by an extremely high energy expenditure. Odum (1989) reported that energy consumption in urban-industrial ecosystems exceeds by 1-2 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 (in review) points out in a consideration of the ecology of urban fire:

> "The fact is, modern cities remain fire-driven ecosystems. Fire's influence is everywhere, yet fire is almost everywhere invisible....Cars, trucks, buses, motorcycles, tractors, back-hoes, 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 along with 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 (CO_2, nitrogen oxides, sewage, solid wastes, water and air pollutants), 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, 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 flood control and 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 downflow 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 local to national scales 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 a 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, USA

Energy expenditure in urban ecosystems is such that all cities can be considered to be extremely heterotrophic ecosystems, 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 gained widespread appeal because of its apparent simplicity and comparability among cities, regions, or nations (but see van den Bergh et al. 1999). By comparison with a well known heterotrophic ecosystem type, a forest stream, 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 Bear Brook, NH is just 31% of the stream's area (Collins et al. in preparation).

An estimate of the ecological footprint of Phoenix based on per capita data for the USA as a whole (Table 1) provides an interesting perspective on the dependence of Phoenix on production that occurs elsewhere. Since the current Phoenix metropolis covers ca 2000 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. But this is probably an underestimate, since 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, recent 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 II.4).

The ecological footprint is a 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 recent 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 2). Simple calculation procedures that 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 settings (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 index, being transferrable 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 to rates of internal cycling, whereas in open ecosystems like streams transport fluxes dwarf those of nutrient transformations like uptake, mineralization, or denitrification (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 (and biologic inputs are often 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. Baker et al. (in review) constructed a nitrogen budget for the central Arizona-Phoenix ecosystem, which illustrates the dramatic quantitative and qualitative difference in fluxes for an urban ecosystem compared to a non-urban ecosystem. One clear qualitative distinction is deliberate compared to natural inputs and outputs (Table 3); for example, the import of nitrogenous fertilizer for agricultural production. 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. While the fate of this NO_X is currently unknown, it represents a huge flux in comparison with the natural inputs of surface water inflow and atmospheric deposition for Phoenix (Fig. 3), and probably for most cities.

One finding of the mass balance estimate for Phoenix (Fig. 3; Baker et al. in review) is that the city must be accumulating nitrogen at a very high rate (nearly 40 kg ha^{-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} compared to 39.3 kg N ha^{-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 watershed drained by the Salt River (12,000 km^2). Since 1950 the human population has increased from 50,000 to over 2.2 million inhabitants, largely due to immigration. Currently, 54% of water use in the CAP ecosystem is for irrigated agriculture, with 40% to municipal uses and the remainder to industrial uses (AZ Department of Environmental Quality 1994). Surface water from the Salt River supplies 48% of this, while 27% is from groundwater and 22% from the Colorado Rover via the Central Arizona Project Canal.

Since the early 1900's, 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 (Fig. 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 since 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 at a concentration of 20 mg nitrate-N/L, when supplied at a rate of 1.5 m/y (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. Over-fertilization also costs the cotton farmer, not just because of the expense of fertilizer and fuel, but also because high N levels inhibit bole formation and reduce the cotton crop.

The N mass balance also can be a powerful educational tool. Here we consider 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 isn’t looking is one with a long tradition indeed. And it is a national problem – about 32% 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 0, we could produce 32% less milk to satisfy our needs. This means that we would have 32% fewer cows and 32% 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%. Overall, the effect would be to reduce the amount of N pollution created by an individual by about 2 kg per year (Table 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 per year (Lauver and Baker in press). 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 wastewater is about 75%, which means that only 1.75 kg N cap^{-1} y^{-1} actually reaches the river (Lauver and Baker in press). Thus, if an individual stopped flushing the toilet, taking showers, and washing dishes, the reduction in N output to the environment would be 1.75 kg N/y.

This analysis shows that the apparently trivial action of eliminating household milk wastage would be more effective at reducing an individual's output of N to the environment (by 2 kg N/y) than would entirely eliminating his or her production of wastewater (1.75 kg N/y). 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, since the largest input terms are a consequence of human behaviors (e.g, driving patterns, fixation via combustion) and economics (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 sources, sinks, and transformations of nitrogen in the ecosystem? Again, we answer in the affirmative: understanding how humans have manipulated flowpaths may be key to unlocking the transport and transformation dynamics of materials in cities.

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 end-members of a spectrum of human-dominated ecosystems, the time is right to develop a more comprehensive ecosystem theory.

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Table Captions

Table 1. Approximate ecological footprint calculation for Phoenix. Data on consumption are from Wackernagel and Rees (1996). The area of the Central Arizona-Phoenix ecosystem (metropolitan area) is 2000 km^2; thus the ecological footprint for this ecosystem is 56 times its size.

Table 2. Scales of ecological footprints.

Table 3. Inputs and outputs of nitrogen for the CAP ecosystem.

Table 4. Calculating the effect of avoiding milk wastage on an individual's N balance. Units in kg N cap^{-1} y^{-1} unless otherwise indicated.

Figure Captions

Figure 1. Phoenix’s South Mountain, showing the distinct edge where urban development meets the desert. Photo by Ramón Arrowsmith, used with permission.

Figure 2. Changes in land use in the central Arizona-Phoenix area from 1912-1995. Light gray - desert; medium gray - agricultural land; black - urban/suburban land. After Knowles-Yánez et al. (1999).

Figure 3. Nitrogen budget for the CAP ecosystem, showing major categories of inputs, outputs, and change in storage (see Table 3 for subcategories).All values in kg N ha^{-1} y^{-1} (redrawn from data in Baker et al. in review).

Table 1. Approximate ecological footprint calculation for Phoenix. Data on consumption are from Wackernagel and Rees (1996). The area of the Central Arizona-Phoenix ecosystem (metropolitan area) is 2000 km^2; thus the ecological footprint for this ecosystem is 56 times its size. ***Note: this copy for review only.***

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**

Table 2. Scales of ecological footprints. ***Note: this copy for review only.***

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 multi-family), parks and preserves, industrial districts, urban core, agricultural areas, schools or other institutions, etc.

Table 3. Inputs and outputs of nitrogen for the CAP ecosystem. ***Note: this copy for review only.***

INPUTS

Natural Inputs

- Atmospheric deposition
- Surface water inflows
- Biological N_2 fixation (desert)

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

<u>CHANGE IN STORAGE</u>

Deliberate

Landfills

Vegetation (in part)

Built structure

Human population

Inadvertent

Groundwater and vadose zone

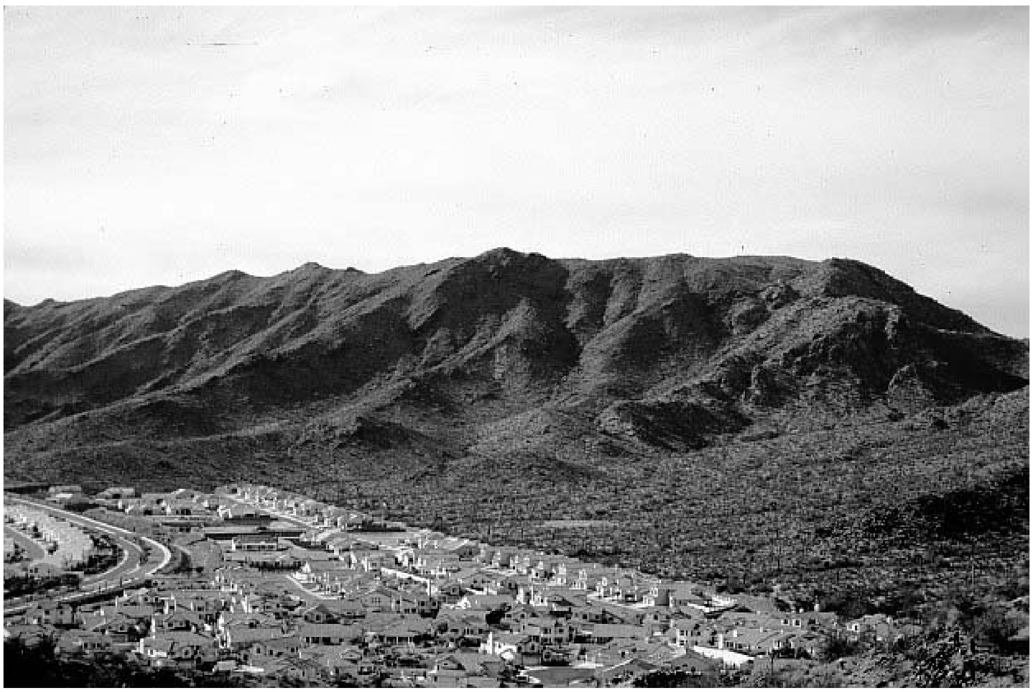
Vegetation and soils

Table 4. Calculating the effect of avoiding milk wastage on an individual's N balance. Units in kg N cap^{-1} y^{-1} unless otherwise indicated. ***Note: this copy for review only.***

Budget Term and Reference	Current	No Waste
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[1]	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] Assuming 50% loss by volatilization and leaching

Figure 1. Phoenix's South Mountain, showing the distinct edge where urban development meets the desert. Photo by Ramón Arrowsmith, used with permission. ***Note: this copy for review only.***



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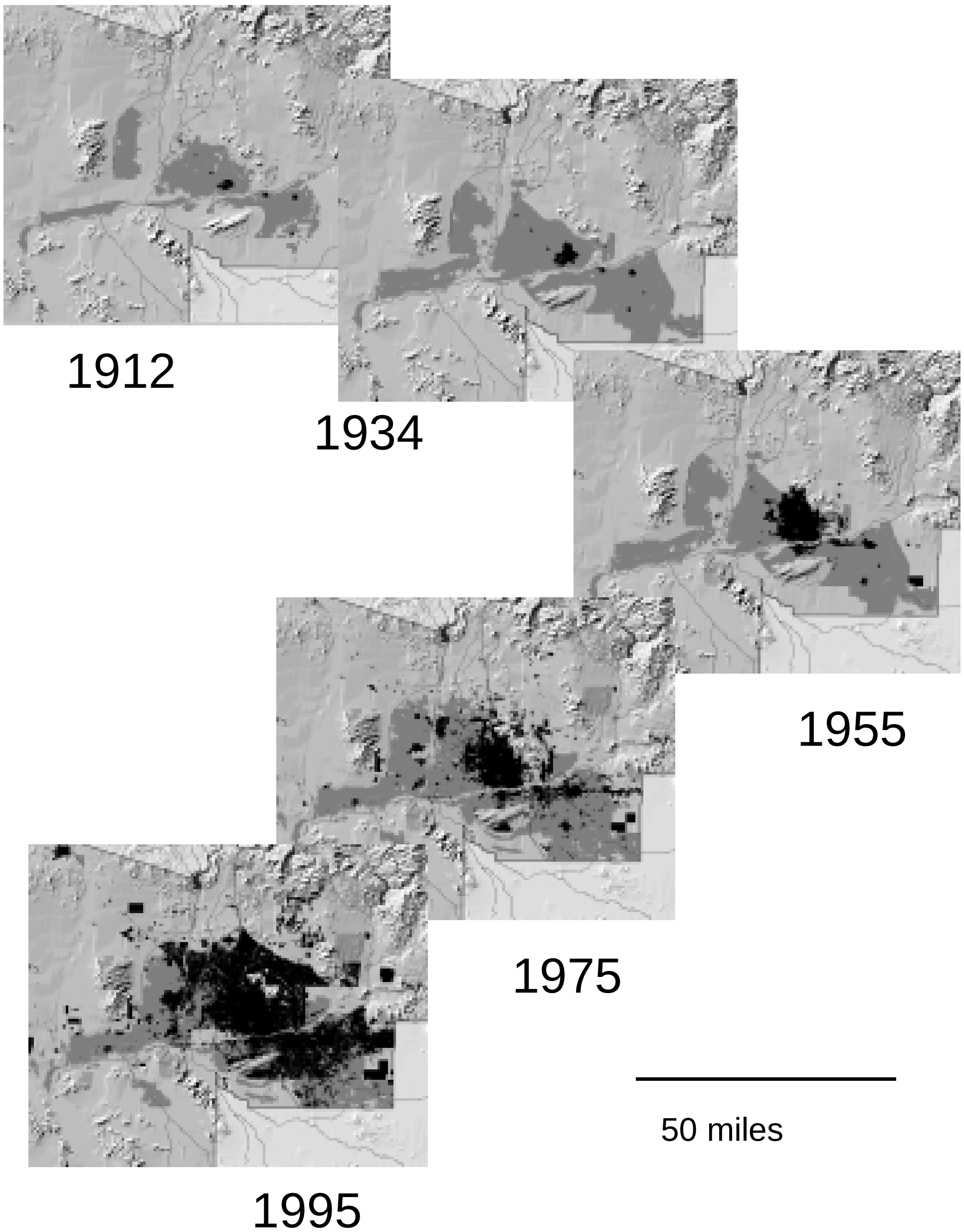

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