Review TRENDS in Ecology and Evolution Vol.21 No.4 April 2006

Full text provided by www.sciencedirect.com

A distinct urban biogeochemistry?

Jason P. Kaye[1], Peter M. Groffman[2], Nancy B. Grimm[3], Lawrence A. Baker[4] and Richard V. Pouyat[5]

[1]The Pennsylvania State University, Department of Crop and Soil Sciences, 116 ASI Building, University Park, PA 16802, USA
[2]Institute of Ecosystem Studies, Box AB, 65 Sharon Turnpike, Millbrook, NY 12545, USA
[3]School of Life Sciences, Arizona State University, Tempe, AZ 85287, USA
[4]Minnesota Water Resources Center, 173 McNeal Hall, 1985 Buford Avenue, St Paul, MN 55108, USA
[5]USDA Forest Service, Room 134, Technology Research Center Building, 5200 Westland Boulevard, University of Maryland, Baltimore County, Baltimore, MD 21227, USA

Most of the global human population lives in urban areas where biogeochemical cycles are controlled by complex interactions between society and the environment. Urban ecology is an emerging discipline that seeks to understand these interactions, and one of the grand challenges for urban ecologists is to develop models that encompass the myriad influences of people on biogeochemistry. We suggest here that existing models, developed primarily in unmanaged and agricultural ecosystems, work poorly in urban ecosystems because they do not include human biogeochemical controls such as impervious surface proliferation, engineered aqueous flow paths, landscaping choices, and human demographic trends. Incorporating these human controls into biogeochemical models will advance urban ecology and will require enhanced collaborations with engineers and social scientists.

New models for a new discipline

Urban (see Glossary) land area and the global urban population have expanded dramatically over the past three decades, with a concurrent expansion in urban ecological research to understand both the environment in which most people live and the feedbacks among urbanization and ecosystem structure and function. Although urban ecosystems are literally 'in our backyard', they are a frontier for ecology, and there is little previous research available to guide urban ecologists. With little data or theory of its own, it is tempting for researchers of this young discipline to borrow models designed for agricultural or unmanaged ecosystems and to apply them to the urban setting. However, implicit in such a transfer of knowledge is the hypothesis that urban ecosystems are qualitatively similar to other ecosystem types.

This hidden hypothesis has important implications for progress in urban ecology. If urban and non-urban ecosystems function similarly, then the new discipline will result in the extension of existing ecological theory to urban field sites. By contrast, if urban ecosystems are fundamentally different from non-urban ecosystems, then urban research has the potential to enrich the field of ecology with new theoretical advances. Collins *et al.* [1] argue that traditional ecological models might not transfer well to urban environments because they fail to represent human controls on ecosystem structure and function. Support for this idea has been documented for plant species diversity, which correlated with population density in Phoenix, AZ [2] and for vegetation cover, which was controlled by landscaping choices linked to social status in Baltimore, MD [3]. Our main purpose here is to determine through a review of the recent research whether urban biogeochemistry is distinct from non-urban biogeochemistry as a result of human-controlled energy and element fluxes. We illustrate variability among cities by comparing Phoenix and Baltimore, both of which were funded as long-term urban ecological research sites in 1997.

Input-output budgets for urban ecosystems

H.T. Odum [4] recognized long ago that an industrialized city expends 10–100 times more energy compared with most unmanaged ecosystems and that urban metabolism depends upon external energy and matter (Box 1). Two approaches to characterizing this dependence upon external inputs at the whole-ecosystem scale are the ecological footprint [5–8] and mass-balance [9–13].

Ecological footprints

The ecological footprint of a city is the total land area required to meet the demands of its population in terms of consumption and waste assimilation [5]. Previous research calculated footprints that were hundreds of times larger than that of the cities themselves [6], but we believe that these studies are flawed because they do not incorporate biophysical setting, population size

Glossary

Biogeochemistry: the study of transport and transformation of matter and energy in ecosystems.
City: a center of high human population density within an urban ecosystem. Synonymous with 'urban core' in this review.
Urban: pertaining to an urban ecosystem.
Urban ecosystems: areas where people live at high densities (<186 people km^{-2}) and in high numbers (usually a total population <50 000), or where the built infrastructure covers a large portion of the land surface. For some ecological studies, it is relevant to include suburbs and less densely populated areas that are connected to the urban core by energy and material flows. Based on [14].

Corresponding author: Kaye, J.P. (jpk12@psu.edu).
Available online 6 January 2006

www.sciencedirect.com 0169-5347/$ - see front matter © 2006 Elsevier Ltd. All rights reserved. doi:10.1016/j.tree.2005.12.006

Box 1. A conceptual model

Similar to non-urban biogeochemical models, a conceptual model of urban biogeochemical cycles (Figure I) includes major elemental pools (boxes), transformations of materials within those pools (dotted arrows), fluxes among pools (solid arrows), and controls on these fluxes (bowties). The model is generic with respect to elements (plug in your element of interest) and our choice of pools, fluxes, transformations and controls is not comprehensive. The urban ecosystem can be 'black boxed' such that only inputs and outputs to the entire system are measured (the largest box in Figure I), or internal components (land, atmosphere and hydrosphere) of the system can be studied.

Humans are likely to affect all element cycles in cities, but we focus here on ways in which humans control five dominant drivers of biogeochemical cycling. These drivers, shown with stick-figure bowties to denote human control are: (i) hydrology (humans build impervious surfaces and drainage networks that alter aqueous flow paths and elemental transport; Figure Ia); (ii) atmospheric chemistry (elevated CO_2, NO_x, ozone, organic aerosols and metals in urban atmospheres interact to alter plant growth and ecosystem carbon and nitrogen cycling; Figure Ib); (iii) climate (urban heat islands and irrigation alter evaporation, transpiration and probably other biogeochemical process rates; Figure Ic); (iv) nutrients (food and fertilizer cycling affect plant growth and interact with human-dominated hydrology and atmospheric chemistry; Figure Id); and (v) vegetation and land-use (humans impose a variable distribution of land use types within and around cities, and within each land-use type, they exert strong control over plant composition and, thus, ecosystem function; Figure Ie).

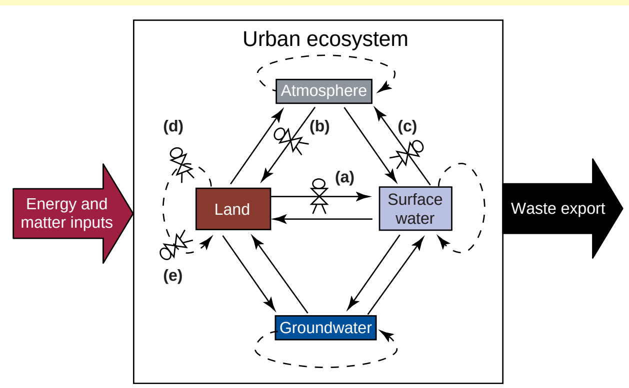

Figure I.

and per capita consumption rate into the footprint calculation. Until these factors are included, footprints depict negative impacts of cities without accounting for the probable efficiency of dense urban living [14]. For example, a rural county in Arizona has a smaller footprint than does Phoenix, although Phoenicians, with small houses on small lots, access to public transportation, and efficient distribution of food and manufactured products, are likely to have a lower per capita impact on biogeochemical cycles than rural residents. This contrast might be even more marked in older, more centralized cites such as Baltimore [15].

Recent footprint modifications separate out water, food, or waste footprints [7] and account for variable productivity of land [8,16]. For example, contrasting the water footprint to city area ratio for Phoenix metro to that of Baltimore-Washington (21 versus 4, respectively) illustrates the impact of local climates on footprints [7]. Our overall impression of footprint studies is that the concept has heuristic potential, but that footprints do not yet provide a realistic representation of urban biogeochemical cycles for the reasons described above. As such, we favor a mass-balance approach.

Mass-balance approaches

Mass balances quantify inputs and outputs of elements or materials (or these values converted to energy units), and yield information about whether a city is a source (input > output) or sink (input < output) for the material and how the material is transformed by urban activities (Box 1). Carbon mass balance shows that cities are major sources of CO_2 and large contributors to its global enrichment in the atmosphere [16,17], but that this is a consequence of both extremely large carbon imports (as fuels, including oil, gas, coal or biofuel [9]) and a lack of significant carbon sequestration. In Phoenix, most emitted CO_2 comes from cars, although human respiration is a measurable 1.6% of emissions [17]. Biomass reduction from regional land clearing for development also contributes CO_2 to the atmosphere, particularly in productive regions [18]. Urban vegetation represents a small carbon sink compared with fluxes associated with fossil-fuel burning [19]. In contrast to carbon, other elements might be accumulating in urban ecosystems. The few available studies show cities to be sinks for nitrogen (Figure 1 [11–13]), phosphorus [10,12] and, in some cases, metals (Box 2 [20,21]).

To predict changes in urban mass balance over time or among cities, urban ecologists must incorporate several human controls into new biogeochemical models. Population size influences material importation, transformation and waste generation, but these aspects of urban metabolism are also dependent upon biophysical factors (e.g. climate), urban form (e.g. compact versus sprawling), and social factors (e.g. household size [22] and affluence [23]). These factors all affect per capita consumption and represent examples where additional social-science information is needed to understand material use because the latter cannot be predicted from population size alone. Current trends of declining persons per household [22] and increasing flooring area per house [24] will increase energy use and associated material demand disproportionately to predictions based on population growth alone. Such socioeconomic changes could cause large increases in urban metabolism of Asian cities [23], particularly in China, where rapid economic growth is coupled with an already large population. Although social scientists have long debated the environmental and social costs and benefits of compact urban form [14,25,26], little ecological information is available to answer the key question of what the most ecologically sound configuration for millions of people inhabiting any area is. We suggest that sociological and ecological urban models would benefit from interdisciplinary collaborations addressing this question (Box 3).

Drivers of urban biogeochemistry

Even when the mass balances show cities to be a net sink, many urban material transfers impair water [26,27] and air quality (Figure 1 [28,29]). Thus, to learn whether and how urban biogeochemistry is distinct from non-urban

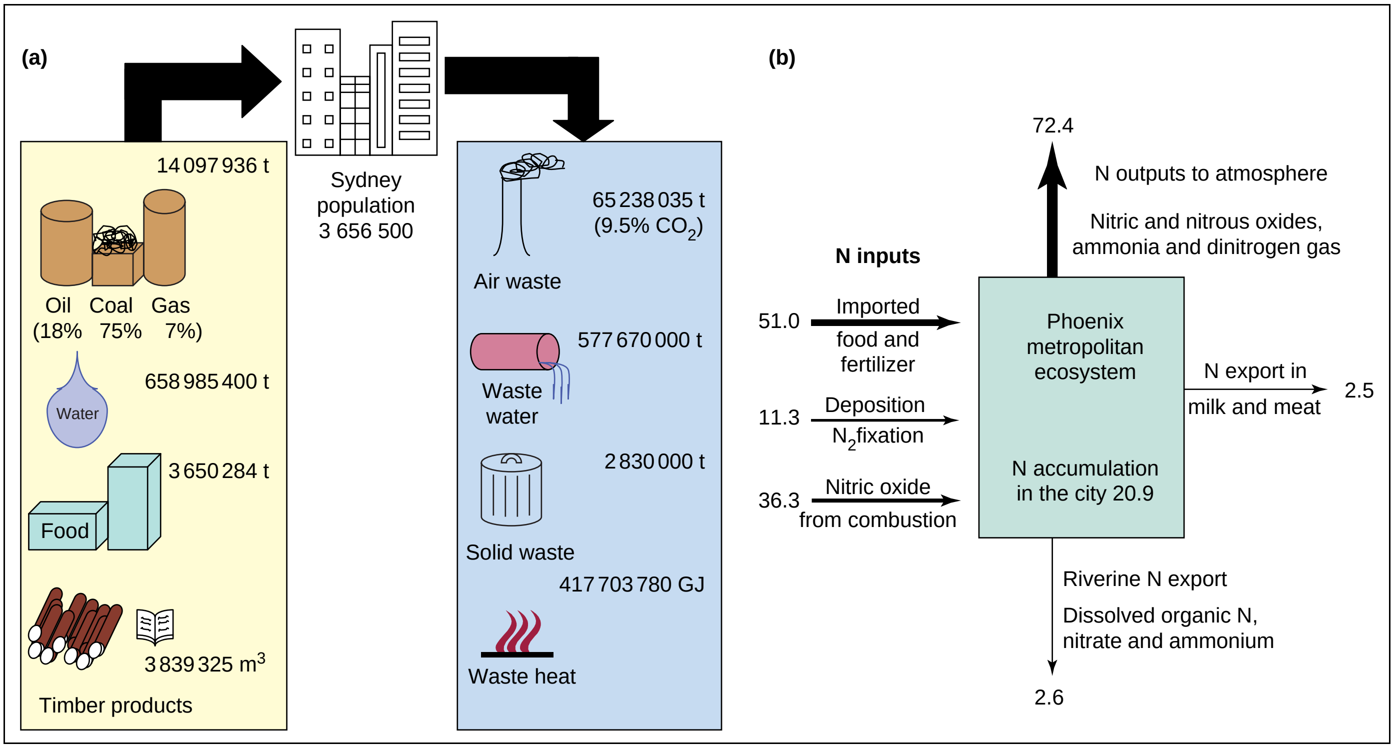

Figure 1. An urban metabolism diagram for Sydney, Australia in 1990 **(a)** and the nitrogen mass balance for Phoenix, Arizona in the 1990s [**(b)**; units are Gg N y^{-1}]. In these examples, approaches quantifying inputs and outputs, without standardizing to energy units or mass units of a single element (a) are contrasted with mass fluxes of a single element (b). The contrast shows that single-element studies can be balanced to reveal specific points of element accumulation and loss. In this case, the Phoenix N mass balance led directly to suggestions that could decrease urban air and water N pollution [11]. We suggest that material-balance studies such as (a) could be more useful in urban ecosystem ecology if the stoichiometric ratios of elements in the material flows (rather than just bulk material mass) were calculated because the relative proportion of nutrient elements will determine how downstream recipient systems respond to these waste products [65]. Wastewater does not include stormwater in (a). Panel (a) reproduced, with permission, from [71].

ecosystems, one must consider not only city-wide mass balance, but also the principal drivers of elemental transport and transformation. Non-urban biogeochemical research has identified several interconnected drivers (or controls) of biogeochemical cycles, including hydrology, atmospheric chemistry, climate, nutrients, vegetation composition and land-use. Here, we consider how humans affect these drivers (Box 1) and whether traditional models can accommodate human influences on biogeochemistry.

Hydrology

Humans change hydrology by altering water supply and drainage in their settlements and by constructing impervious surfaces that change the way in which water moves through urban ecosystems (Figure 2). Impervious surface cover within and among urban ecosystems varies greatly, from $<10\%$ in low-density residential areas to $>80\%$ in some urban cores [30]. The dominant effect of impervious surfaces is to increase surface runoff and decrease infiltration of precipitation. In many natural ecosystems, $>90\%$ of water flow from uplands to streams is by subsurface flow. In a humid-region city such as Baltimore, increasing impervious surfaces to 10–20% of the total surface area doubles surface runoff compared with a forested area, with shorter lag times between precipitation input and discharge, and higher flood peak discharges during storms [31]. Aridland streams in Phoenix have naturally low infiltration to runoff ratios and short lag times, so hydrographs might not show such dramatic shifts following urbanization [32]. The biogeochemical consequences of altered infiltration to runoff ratios can include a reduced influence of soil and plant processes on water chemistry [33], reduced stream baseflow [34], increased stream temperatures [35] and lower water tables, especially in riparian areas thought to be hotspots of nutrient removal between terrestrial and aquatic components of the landscape [36,37].

Cities have engineered infrastructure to import water and remove waste and drainage water. In cities with arid climates or seasons, imported water for irrigation can exceed precipitation, leading to increased terrestrial primary productivity and nutrient cycling [38–41]. Engineered wastewater flow paths include onsite (septic) or municipal sewage systems where water is treated and then discharged to surface waters or recycled as irrigation. In some cases, municipal water discharge raises groundwater and streamflow levels as well as nutrient concentrations [42]. Urban water supply and drainage systems can be leaky [43], and movement of surface water into sewers contributes to riparian drying, whereas leakage from sewers and discharges from combined sewer-stormwater overflows contaminate surface waters with pathogens and nutrients [44]. Engineered stormwater drainage also accelerates downstream material transport [36] and replaces heterogeneous, nutrient-retentive streams with highly engineered (and probably less nutrient-retentive) ones [31,32]. Confronted with human engineering of impervious surfaces, water sources and drainage, traditional ecological models fail routinely in urban

Box 2. Industrial ecology

The discipline of industrial ecology has been developing mainly in the field of engineering [66], but it has a lot to offer urban ecologists. One of the main tools of industrial ecologists is 'material flow analysis', which is a mass-balance model for an element. However, the elements of interest are usually metals, and the pools and fluxes include those required for manufacturing, economic consumption, waste and recycling. Thus, although the tool of mass balance is identical to that used by ecologists, its application to different elements in human-dominated systems is thus far unique to industrial ecology.

Recent material flow analyses show wide variations among regions. Europe and China are sinks for zinc [20] and copper [21], whereas Latin America is a source. In Europe, 5-kg zinc capita^{-1} y^{-1} are imported, 2-kg zinc capita^{-1} y^{-1} are lost or exported, and 3-kg zinc capita^{-1} y^{-1} accumulate in the system [21]. For copper, the net accumulation is 6-kg capita^{-1} y^{-1} in Europe [20]. Although these zinc and copper budgets were not conducted for individual cities, the main sinks for these elements are electrical and civil engineering of urban infrastructure.

A major insight from material flow analyses is that metals that are immobile in non-urban ecosystems can become highly mobile with human cycling [67]. For example, in Stockholm, annual mercury fluxes exceeded stocks by 10–100-fold [68]. A recent analysis of 77 elements revealed that human mobilization dominated the cycles of elements with low aqueous solubility in nature [67], suggesting that human impacts on element cycles can be predicted from environmental kinetics.

Because of the increased mobility of metals in urban ecosystems, most material flow analyses reveal significant leaks where metals enter the environment with probable implications for ecosystem dynamics and human health [20,21,66,67]. Atmospheric scientists have shown that metals that leak from technology resource cycles are not distributed evenly throughout the urban landscape [28]; one ecological study showed that increasing concentrations of metals in urban forest soils correlated with declines in microinvertebrate and fungal abundance [69], suggesting effects on non-metal biogeochemical cycling. It seems clear that potential linkages between industrial ecology, metal cycling, human health and urban ecology represent a suitable area for interdisciplinary collaboration.

ecosystems. A promising approach would be to combine engineering models, which do a good job moving water around in cities, with more traditional biogeochemical models that depict soil and vegetation processing of water.

Atmospheric chemistry

Altered atmospheric chemistry has been a human health concern for centuries because urban atmospheres have higher concentrations of carbon, nitrogen, aerosols, metals, and ozone relative to atmospheres unaffected by cities [28,45,46]. Sources of nitrogen to the atmosphere include oxidized compounds (NO_2, HNO_3 and aerosol NO_3^-) originally emitted as NO by combustion, and reduced compounds (NH_3 and NH_4^+) emitted by catalytic reduction and agriculture. Urban combustion also creates an elevated 'CO_2 dome' or plume with near-surface CO_2 concentrations varying by >100 ppm along urban-rural gradients [47]. Atmospheric organic carbon originates mainly from fossil-fuel combustion and cooking oil [28]. Many of these urban pollutants are precursors to ozone formation, but rates of photolytic ozone production and consumption in the atmosphere vary such that net increases in ozone can occur either within or downwind of the urban core [45].

Box 3. Three areas for future urban biogeochemical research

Urban engineering

For all the drivers that we explored, engineered impervious surfaces, aqueous flow paths and gaseous fluxes from fossil-fuel burning have a role in determining biogeochemical cycling rates. In addition, industrial ecology has shown that some elements that are immobile in non-urban environments cycle rapidly in urban areas as a result of engineering (Box 2). Engineers use systems-modeling approaches that are similar to the systems models that ecologists use; indeed, ecosystem modeling has historical roots in engineering [4]. Thus, it should be possible to incorporate models used to design and implement urban engineering projects into urban biogeochemical models to predict the effects of the built environment on nutrient and energy cycles. Eventually, these hybrid engineering-ecology models could be linked to the energy and material demands generated by human demographic trends and household actions discussed below.

Human demographic trends

Human population dynamics determine the demand for food, energy, and engineered structures, which we have identified here as major controls on urban hydrology, climate, nutrient cycling and atmospheric chemistry. Ecologists have abundant experience in modeling animal population dynamics and linking these dynamics to biogeochemistry [65]. Elemental fluxes through most large terrestrial mammals are small and, thus, ignored. In urban ecosystems, traditional ecological approaches are inappropriate because socioeconomics is the best predictor of demographic trends, and the biogeochemical cycling of elements through and by people is a substantial component of urban element budgets [11,12]. Correlative links between human population size and nutrient cycling are available [27], but a mechanistic understanding will require models that link demographics, diets and waste. Such models are beyond the scope of current ecological theory and must be developed through collaborations with social scientists.

Household-scale actions

In addition to the size of their populations, humans drive urban biogeochemical cycles with their actions, which are based in culture, attitudes and beliefs and are constrained by institutional and socioeconomic factors. Landscape design and management choices such as species composition [2,3] and lawn fertilization and irrigation regimes [38,39,55] can alter vegetation and nutrients as drivers of urban biogeochemical cycles. Urban housing choices such as subdivision style [26], household size [22], and floor area per house [24] will affect impervious surface area and energy demands that control urban hydrology, climate and atmospheric chemistry. Household recycling choices motivated by economics and aesthetics [70] can also affect biogeochemical cycles [66]. Thus, we are unlikely to generate accurate predictive models of urban biogeochemistry without incorporating the actions that people take in managing their landscapes and households, and we are unlikely to be able to predict those actions without understanding their variation as a result of culture, attitudes, and socioeconomic setting.

The multiple pollutants and nutrients in urban atmospheres interact in complex ways to alter urban biogeochemical cycles. Aerosols can reduce ground-level short wave radiation by 20% [48], and atmospheric nitrogen, carbon and metals are deposited to ecosystems within and downwind of the urban source [28,29,46]. The effects of elevated nitrogen deposition on eastern deciduous forests (such as those surrounding Baltimore) have been studied extensively and reviewed elsewhere [29], but much less is known about the effects of nitrogen deposition on the arid ecosystems surrounding Phoenix. In addition, little is known about the net plant stomatal responses to high concentrations of CO_2, nitrogen gases, ozone and

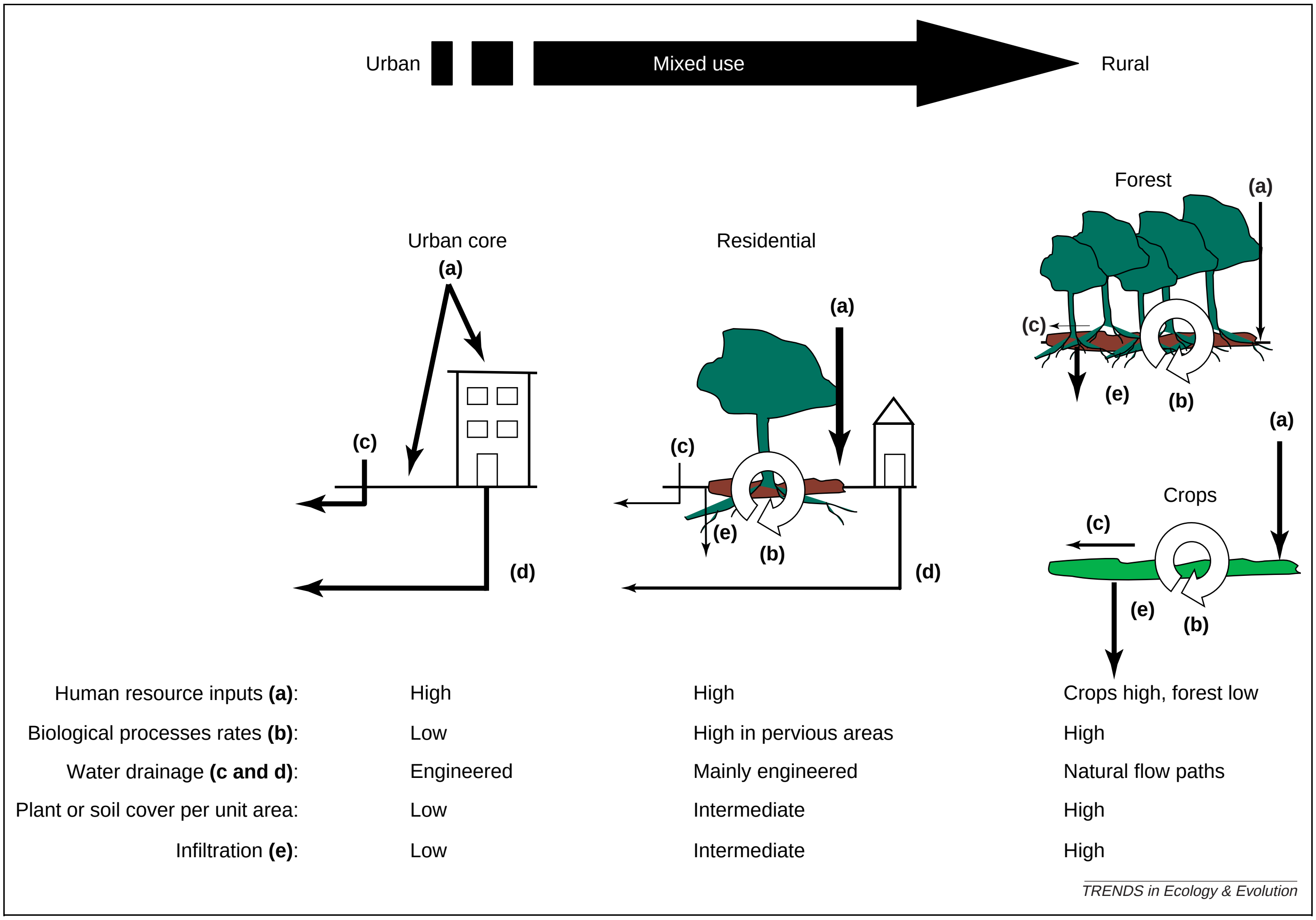

Figure 2. Framework showing how anthropogenic and natural ecosystem characteristics change along a continuum from urban (almost entirely human-made) to rural ecosystem types (those with the least human modification). At the urban core, engineered flow paths for surface water **(c)** and wastewater **(d)** disconnect material inputs **(a)** and flows from infiltration **(e)** and natural processing **(b)** that occur in forests and rural agricultural ecosystems. In these latter areas, the connectivity between the engineered and natural components of the ecosystem is low. In residential areas, the connectivity between engineered and natural components can be relatively high, depending on the spatial relationship of impervious and pervious surfaces. Management and environmental inputs in residential areas can be high per unit of pervious area (a). However, depending on site history, soil type and the concentration of flows, these areas can have surprisingly high cycling rates (b) for processing or storing these inputs.

aerosols in urban atmospheres. In one study along an urban-rural gradient in the New York City metropolitan area [45], declines in tree sapling growth (in pots) owing to high ozone exposure in rural and suburban areas were greater than potential stimulation of sapling growth from warmer temperatures and nitrogen enrichment in the urban core. Whereas some existing biogeochemical models predict impacts of elevated CO_2 on plant growth, they do not predict the net effects of multi-gas interactions that are common in urban atmospheres. Furthermore, predicting the relative abundances of these gases over time will require models that reflect the influences of human demographics and engineering technologies on energy use and emissions.

Climate

Temperature is an important driver of biogeochemical process rates, and urban air temperatures are generally warmer than rural air temperatures because impervious surfaces adsorb light energy during the day and release long wave radiation (heat) at night creating an urban heat island [48]. Some common characteristics of heat islands are (i) higher mean and minimum daily air temperatures, with little or no change in the maximum; (ii) a longer warm period within a given day; (iii) a longer warm season and a shorter cool season; and (iv) more frost-free days [49,50]. Urban temperatures are also affected by changes in evaporation caused by impervious surfaces or irrigation. Evaporative cooling can account for 25–41% of total urban radiation [51,52], and irrigated urban soils can be cooler in summer than are dry, non-urban soils [39].

Changes in air and soil temperatures can alter biogeochemistry by altering plant and microbial growth rates. In humid environments, such as Baltimore, the growing season in cities is about a week longer than in nearby rural areas [53], which can increase annual nutrient and carbon uptake by plants. By contrast, warming in arid Phoenix can suppress photosynthesis during the hottest part of the year [38].

Current biogeochemical models include temperature as an input variable, rather than as a human-controlled driver. However, there is a simple empirical relationship to predict the rise in minimum temperature in a city relative to the surrounding rural landscape as a function of population [51]. For larger cites (<1 million inhabitants), the maximum difference between urban and rural temperatures is 6–12 °C. Regression models of this type could be incorporated into biogeochemical models to predict temperature, but eventually urban biogeochemical models will need to predict human demographic trends to depict changes in heat islands over time. New models might also need to incorporate human water-use choices because people respond to high temperatures by using more water, which alters latent heat flux [38].

Nutrients

Urban ecosystems have been identified as sources of nutrient pollution to receiving waters in many areas worldwide [26,27]. Humans alter nutrient sources by changing atmospheric deposition rates and by importing fertilizer and food [27,54]. In Phoenix, food and fertilizer nitrogen imports were $\sim$20 Gg (gigagrams) y^{-1} each, and 36 Gg y^{-1} of atmospheric nitrogen were generated by combustion [11]; collectively, these human-mediated fluxes account for over 90% of N input (Figure 1). Urban fertilizer application and intensity are more variable than in agricultural ecosystems, with fertilized areas (lawns) occupying discrete portions of the landscape and application rate varying with the preferences of multiple land managers [55]. Imported human food is entrained into the engineered infrastructure, with waste products transported from the site of consumption via sewers and garbage [54]. A major exception can be imported pet food because pet waste is generally not collected, treated or exported [11].

Humans directly control the food and waste nutrient fluxes described above, but they also influence nutrient transformations in plants and soils. For example, the ability of riparian areas to convert soil nitrate to nitrogenous gases (denitrification) can be reduced in humid urban areas such as Baltimore [37] owing to the hydrological changes discussed above. Human activities also create hotspots for denitrification in stormwater detention basins, ditches, gutters, lawns and all places where water, nitrate and organic matter accumulate [39,56,57]. In Phoenix, where riparian zones have been eliminated, stormwater detention basins are especially important hotspots for denitrification [56]. Humans also create sinks for nutrients in landscaping because urban vegetation is often young and rapidly growing and, therefore, accumulating nutrients in biomass and soil organic matter [58].

In aggregate, nutrient sinks appear to be important in urban ecosystems because recent analyses have found that hydrological outputs can account for >30% of urban nitrogen inputs (Figure 1 [11,13]). The demographic trends that control food nutrient imports and the waste engineering that controls nutrient exports are not components of current ecological models. Because human fluxes dominate the mass of urban nutrient cycles, we expect that traditional biogeochemical models will fail to predict city-scale nutrient cycling (Figure 1).

Land use and vegetation cover

Whereas transitions between agricultural and native land-use change have been relatively well studied, conversions to urban land uses have received less attention. Major differences in urban land-use change include increases in variability, patchiness and irreversibility compared with other land-use types [59,60]. Abandoned agricultural land recovers many aspects of native ecosystem structure and function (i.e. the changes are somewhat reversible), whereas trajectories of recovery following urban land use are uncertain [60]. Urban variability results from human-caused disturbance (e.g. movements of soil and building construction) and is more pronounced during, rather than after, the land-development process. Patchiness is common to all ecosystems, but the spatial pattern of human activity results in landscapes that are continually subdivided into ever smaller patches, each with an individual land owner. These parcels and other human works are overlain on natural environmental patchiness, producing a complex landscape mosaic (Figure 2).

Plant communities in urban ecosystems are dynamic and diverse. Urbanization drives many local extinctions, but cities are also epicenters for intentional (e.g. landscaping) and unintentional introduction of non-native species [61]. Some non-native species are important pathogens or pests (e.g. Dutch elm disease or the Asian long-horned beetle), but they, along with imported plants, contribute to the high species richness of urban communities [2,62]. In arid Phoenix, intentional introductions, along with the addition of water, tend to strongly increase plant community diversity [2,61]. In humid-region cities, such as Baltimore, urbanization is likely to reduce plant diversity by reducing natural habitat (a hypothesis currently being tested in Baltimore). Current biogeochemical models do not include landscaping choices that determine urban plant community composition [3]. Furthermore, the exotic species that are common in the urban environment can alter nitrogen cycling, primary productivity and other biogeochemical processes [63,64], but these effects are often species-specific and difficult to predict.

Conclusion

On a molecular level, urban biogeochemistry should not be distinct from other ecosystems because the physical and chemical laws that govern biogeochemical reactions are universal. Yet, we show here that constant physicochemical laws do not enable a simple transfer of biogeochemical models to urban ecosystems because the drivers of biogeochemical reactions are under human control. Although human control is complex, all the drivers we examined appear to be linked to three classes of human activity: engineering, urban demographic trends and household-scale actions (Box 3). We suggest that these areas should be foci for future urban ecological research that will require urban ecologists to work with engineers, industrial ecologists and social scientists. Ultimately, the challenge is to integrate human choices and ecosystem dynamics into a seamless, transdisciplinary model of biogeochemical cycling in urban ecosystems.

In addition to interactions among disciplines, urban ecologists must conduct more research on interactions among biogeochemical cycles. We focused on five drivers as individual processes, but interactions among drivers (Box 1) and elements (Figure 1) are important and difficult to predict. Experiments that integrate multiple biogeochemical alterations to determine which factor(s) has/have the largest impact [45] are required. Urban research should include multi-element mass-balance studies to understand how human-dominated fluxes couple or decouple elemental cycles in cities [65]. By constructing mass balances at scales from the household to the city, human choice can be linked directly to biogeochemical cycling.

Acknowledgements

This work was supported in part by the Central Arizona-Phoenix LTER (NSF Grants no. DEB-9714833 and DEB-0423704; J.P.K., N.B.G. and L.A.B.), the Baltimore Ecosystem Study LTER (NSF Grants no. DEB-9714835 and DBI-0423476; P.M.G. and R.V.P.), the NSF Ecosystem Studies Program (DEB-0514382; N.B.G., and DEB-0514379; J.P.K.), and the NSF Biocomplexity in the Environment Program (EAR-0322065; J.P.K. and L.A.B.). We thank David Lewis and four anonymous reviewers for comments that improved the article.

References

1 Collins, J.P. *et al*. (2000) A new urban ecology. *Am. Sci.* 88, 416–425
2 Hope, D. *et al*. (2003) Socioeconomics drive urban plant diversity. *Proc. Natl. Acad. Sci. U. S. A.* 100, 8788–8792
3 Grove, J.M. *et al.* Characterization of households and its implications for the vegetation of urban ecosystems. *Ecosystems (N.Y., Print)* (in press)
4 Odum, H.T. (1971) *Environment, Power and Society*, Wiley-Interscience
5 Rees, W. and Wackernagel, M. (1996) Urban ecological footprints: why cities cannot be sustainable and why they are a key to sustainability. *Environ. Impact Asses. Rev.* 16, 223–248
6 Folke, C. *et al*. (1997) Ecosystem appropriation by cities. *Ambio* 26, 167–172
7 Luck, M.A. *et al*. (2001) The urban funnel model and the spatially heterogeneous ecological footprint. *Ecosystems (N.Y., Print)* 4, 782–796
8 Erb, K.H. (2004) Actual land demand of Austria 1926–2000: a variation on ecological footprint assessments. *Land Use Policy* 21, 247–259
9 Decker, E.H. *et al*. (2000) Energy and material flow through the urban ecosystem. *Annu. Rev. Energ. Environ.* 25, 685–740
10 Faerge, J. *et al*. (2001) Urban nutrient balance for Bangkok. *Ecol. Model.* 139, 63–74
11 Baker, L.A. *et al*. (2001) Nitrogen balance for the central Arizona-Phoenix (CAP) ecosystem. *Ecosystems (N.Y., Print)* 4, 582–602
12 Warren-Rhodes, K. and Koenig, A. (2001) Escalating trends in the urban metabolism of Hong Kong: 1971–1997. *Ambio* 30, 429–438
13 Groffman, P.M. *et al*. (2004) Nitrogen fluxes and retention in urban watershed ecosystems. *Ecosystems (N.Y., Print)* 7, 393–403
14 Pickett, S.T.A. *et al*. (2001) Urban ecological systems: linking terrestrial ecological, physical, and socioeconomic components of metropolitan areas. *Annu. Rev. Ecol. Syst.* 32, 127–157
15 Owen, D. (2004) Green Manhattan. *New Yorker* 80, 111–123
16 Imhoff, M.L. *et al*. (2004) Global patterns in human consumption of net primary production. *Nature* 429, 870–873
17 Koerner, B. and Klopatek, J. (2002) Anthropogenic and natural CO_2 emission sources in an arid urban environment. *Environ. Pollut.* 116, S45–S51
18 Boyle, C.A. and Lavkulich, L. (1997) Carbon pool dynamics in the Lower Fraser Basin from 1827–1990. *Environ. Manage.* 21, 443–455
19 Nowak, D.J. (1993) Atmospheric carbon reduction by urban trees. *J. Environ. Manage.* 37, 207–217
20 Spatari, S. *et al*. (2003) The contemporary European zinc cycle: 1-year stocks and flows. *Res. Conserv. Recyc.* 39, 137–160
21 Graedel, T.E. *et al*. (2004) Multilevel cycle of anthropogenic copper. *Environ. Sci. Technol.* 38, 1242–1252
22 Liu, J.G. *et al*. (2003) Effects of household dynamics on resource consumption and biodiversity. *Nature* 421, 530–533
23 Bai, X. (2003) The process and mechanism of urban environmental change: an evolutionary view. *Int. J. Environ. Pollut.* 19, 528–541
24 Wilson, A. and Boehland, J. (2005) Small is beautiful: U.S. house size, resource use, and the environment. *J. Ind. Ecol.* 9, 277–287
25 Audirac, I. *et al*. (1990) Ideal urban form and visions of the good life – Florida growth management dilemma. *J. Am. Plann. Assoc.* 56, 470–482
26 Nassaur, J. *et al*. (2004) Exurban residential subdivision development: effects on water quality and public perception. *Urban Ecosyst.* 7, 267–281
27 Boyer, E.W. *et al*. (2002) Anthropogenic nitrogen sources and relationships to riverine nitrogen export in the northeastern U.S.A. *Biogeochem.* 37/38, 137–169
28 Azimi, S. *et al*. (2005) Sources, distribution, and variability of hydrocarbons and metals in atmospheric deposition in an urban area (Paris, France). *Sci. Total Environ.* 337, 223–239
29 Fenn, M.E. *et al*. (2003) Ecological effects of nitrogen deposition in the western United States. *BioScience* 53, 404–420
30 Schueler, T. (1994) The importance of imperviousness. *Watershed Protect. Technol.* 1, 100–111
31 Paul, M.J. and Meyer, J.L. (2001) Streams in the urban landscape. *Annu. Rev. Ecol. Syst.* 32, 333–365
32 Grimm, N.B. *et al*. (2005) N retention and transformation in urban streams. *J.N. Am. Benthol. Soc.* 24, 626–642
33 Gold, A.J. *et al*. (2001) Landscape attributes as controls on groundwater nitrate removal capacity of riparian zones. *J. Am. Water Res. Assoc.* 37, 1457–1464
34 LeBlanc, R.L. *et al*. (1997) Modeling the effects of land use change on the water temperature in unregulated urban streams. *J. Environ. Manage.* 49, 445–469
35 Wang, L.H. and Kanehl, P. (2003) Influences of watershed urbanization and instream habitat on macroinvertebrates in cold water streams. *J. Am. Water Res. Assoc* 39, 1181–1196
36 Walsh, C.J. *et al*. (2005) The urban stream syndrome: current knowledge and the search for a cure. *J.N. Am. Benthol. Soc.* 24, 706–723
37 Groffman, P.M. *et al*. (2002) Soil nitrogen cycling processes in urban riparian zones. *Environ. Sci. Technol.* 36, 4547–4552
38 Martin, C. and Stabler, L.B. (2002) Plant gas exchange and water status in urban desert landscapes. *J. Arid Environ.* 51, 235–254
39 Kaye *et al*. (2004) Methane and nitrous oxide fluxes from urban soils to the atmosphere. *Ecol. Appl.* 14, 975–981
40 Milesi, C. *et al*. (2003) Assessing the impact of urban land development on net primary productivity in the southeastern United States. *Remote Sens. Environ.* 86, 401–410
41 Imhoff, M.L. *et al*. (2000) The use of multisource satellite and geospatial data to study the effect of urbanization on primary productivity in the United States. *IEEE Trans. Geosci. Remote* 38, 2549–2556
42 Sherlock, M.D. *et al*. (2002) Physical controls on septic leachate movement in the vadose zone at the hillslope scale, Putnam County, New York, USA. *Hydrol. Process.* 16, 2559–2574
43 Lerner, D.N. (2002) Identifying and quantifying urban recharge: a review. *Hydrogeol. J.* 10, 143–152
44 US EPA (1995) *Final Report: Sanitary Sewer Overflow Workshop*, US Environmental Protection Agency
45 Gregg, J.W. *et al*. (2003) Urbanization effects on tree growth in the vicinity of New York City. *Nature* 424, 183–187
46 Lovett, G.M. *et al*. (2002) Atmospheric deposition to oak forests along an urban-rural gradient. *Environ. Sci. Technol.* 34, 4294–4300
47 Idso, C.D. *et al*. (2001) An intensive two-week study of an urban CO_2 dome. *Atmos. Environ.* 35, 995–1000
48 Arnfield, A.J. (2003) Two decades of urban climate research: a review of turbulence, exchanges of energy and water, and the urban heat island. *Int. J. Climatol.* 23, 1–26
49 Baker, L.A. *et al*. (2002) Urbanization and warming of Phoenix (Arizona, U.S. A.): impacts, feedbacks, and mitigation. *Urban Ecosyst.* 6, 183–203
50 Toddhunter, P.E. (1996) Environmental indices for the Twin Cities metropolitan area (Minnesota, U.S. A.) urban heat island-1989. *Climate Res.* 6, 59–69

51 Brazel, A.J. *et al*. (2000) The tale of two cities: Phoenix and Baltimore urban LTERs. *Climate Res.* 15, 123–135
52 Grimmond, C.S.B. and Oke, T.R. (1995) Comparison of heat fluxes from summertime observations in the suburbs of four North American cities. *J. Appl. Meteorol.* 34, 873–889
53 White, M.A. *et al*. (2003) Satellite evidence of phenological differences between urbanized and rural areas of the eastern United States deciduous broadleaf forest ecosystems. *Ecosystems (N.Y., Print)* 5, 260–277
54 Bleken, M.A. and Bakken, L.R. (1998) The nitrogen cost of food production: Norwegian society. *Ambio* 26, 134–142
55 Law, N.L. *et al*. (2004) Nutrient input from residential lawn care practices. *J. Environ. Manage.* 47, 737–755
56 Zhu, W.X. *et al*. (2004) Urban nitrogen biogeochemistry: status and processes in green retention basins. *Biogeochem.* 71, 177–196
57 Groffman, P.M. and Crawford, M.K. (2003) Denitrification potential in urban riparian zones. *J. Environ. Qual.* 32, 1144–1149
58 Qian, Y. *et al*. (2003) Long-term effects of clipping and nitrogen management in turfgrass on soil organic carbon and nitrogen dynamics: the CENTURY model simulation. *J. Environ. Qual.* 32, 1695–1700
59 Luck, M. and Wu, J. (2002) A gradient analysis of the landscape pattern of urbanization in the Phoenix metropolitan area of USA. *Landscape Ecol.* 17, 327–339
60 Pouyat, R.V. *et al.* Urban land-use change effects on biogeochemical cycles. In *Terrestrial Ecosystems in a Changing World. Global Change and Terrestrial Ecosystems Synthesis Book*, (Canadell, J. *et al.*, eds) Springer (in press)
61 McKinney, M.L. (2002) Urbanization, biodiversity, and conservation. *BioScience* 52, 883–890
62 Nowak, D.J. *et al.* (2002) *Brooklyn's Urban Forest*, USDA Forest Service General Technical Report NE–290
63 Bohlen, P.J. *et al*. (2004) Ecosystem consequences of exotic earthworm invasion of north temperate forests. *Ecosystems (N.Y., Print)* 7, 1–12
64 Ehrenfeld, J.G. (2003) Effects of exotic plant invasions on soil nutrient cycling processes. *Ecosystems (N.Y., Print)* 6, 503–523
65 Sterner, R. and Elser, J. (2002) *Ecological Stoichiometry*, Princeton University Press
66 Graedel, T.E. and Allenby, R.B. (2003) *Industrial Ecology*, 2nd edn, Prentice Hall
67 Klee, R.J. and Graedel, T.E. (2004) Elemental cycles: a status report on human or natural dominance. *Annu. Rev. Environ. Nat. Res.* 29, 69–107
68 Sviden, J. and Jonsson, A. (2001) Urban metabolism of mercury turnover, emission and stock in Stockholm, 1795–1995. *Water Air Soil Pollut.* 1, 179–196
69 Pouyat, R.V. *et al*. (1994) Environmental effects of forest soil-invertebrate and fungal densities in oak stands along an urban-rural land use gradient. *Pedobiologia (Jena)* 38, 385–399
70 Yi, Y. *et al*. (1999) Factors affecting household-level environmental decision making: a three-country comparison of the determinants of household recycling. *Urban Ecosyst.* 3, 149–161
71 Newman, P.W.G. (1999) Sustainability and cities: extending the metabolism model. *Landscape Urban Plan.* 44, 219–226

AGORA initiative provides free agriculture journals to developing countries

The *Health Internetwork Access to Research Initiative* (HINARI) of the WHO has launched a new community scheme with the UN Food and Agriculture Organization.

As part of this enterprise, Elsevier has given 185 journals to *Access to Global Online Research in Agriculture* (AGORA). More than 100 institutions are now registered for the scheme, which aims to provide developing countries with free access to vital research that will ultimately help increase crop yields and encourage agricultural self-sufficiency.

According to the Africa University in Zimbabwe, AGORA has been welcomed by both students and staff. 'It has brought a wealth of information to our fingertips' says Vimbai Hungwe. 'The information made available goes a long way in helping the learning, teaching and research activities within the University. Given the economic hardships we are going through, it couldn't have come at a better time.'

For more information visit:
http://www.healthinternetwork.net

Elsevier.com – Dynamic New Site Links Scientists to New Research & Thinking

Elsevier.com has had a makeover, inside and out.

As a world-leading publisher of scientific, technical and health information, Elsevier is dedicated to linking researchers and professionals to the best thinking in their fields. We offer the widest and deepest coverage in a range of media types to enhance cross-pollination of information, breakthroughs in research and discovery, and the sharing and preservation of knowledge. Visit us at Elsevier.com.

Elsevier. Building Insights. Breaking Boundaries