Urban Ecology

An International Perspective on the Interaction Between Humans and Nature

Editors

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For Herbert Sukopp, Urban Ecology pioneer

and

Marsha Landolt (Dean of the University of Washington's Graduate School), Debra Nickel (first program administrator of the University of Washington's Urban Ecology Program), and Bob Reineke (pioneer post-doc in the University of Washington's Urban Ecology Program) who gave so much to our interdisciplinary graduate program in Urban Ecology, but died too young

An introduction to Urban Ecology as an interaction between humans and nature

Urban Ecology is the study of ecosystems that include humans living in cities and urbanizing landscapes. It is an emerging, interdisciplinary field that aims to understand how human and ecological processes can coexist in human-dominated systems and help societies with their efforts to become more sustainable. It has deep roots in many disciplines including sociology, geography, urban planning, landscape architecture, engineering, economics, anthropology, climatology, public health, and ecology. Because of its interdisciplinary nature and unique focus on humans and natural systems, the term "urban ecology" has been used variously to describe the study of humans in cities, of nature in cities, and of the coupled relationships between humans and nature. Each of these research areas is contributing to our understanding of urban ecosystems and each must be understood to fully grasp the science of Urban Ecology. Therefore, in *Urban Ecology: an international perspective on the interaction between humans and nature,* we introduce students and practitioners of urban ecology to its roots, bases, and prospects by way of a diverse collection of historical and modern foundational readings. We editors are urban ecologists from the United States, Italy, and Germany who together view these readings as a fair representation of the importance of both natural and social sciences to Urban Ecology.

In this book we collect important papers in the field of Urban Ecology that both set the foundations for the discipline and illustrate modern approaches, from a variety of perspectives and regions of the world. We do this by reprinting important publications, filling gaps in the published literature with a few targeted original works, and translating several key works originally published in German. Our hope is that this collection of thoughts will provide students, practitioners, and professionals with a rich background in some of the core facets of Urban Ecology.

As you study these readings, it may be useful to consider the city as a set of strongly interacting systems or *spheres*. The urban ecosystem includes abiotic spheres (the *atmosphere*, *hydrosphere*, *lithosphere*, and soil or *pedosphere*) and biotic spheres (often viewed as an interacting *biosphere* of urban plants and animals plus the socio-economic world of people, the *anthroposphere*; Fig. 1). The readings deal with each of these spheres, and also with the connections between and amongst them. These connections have been and continue to be viewed very differently by the authors of the articles in this collection. The relative importance of the spheres changes with one's research bias, but more importantly the way and extent that authors have represented the connections among the spheres (or even the degree of isolation of the spheres) differs vastly. Look for these differences as you read the collection. Our view is that the interrelated processes among the subsystems (spheres) must be studied and understood to understand the ecology of a city. This is what modern Urban Ecology strives to do.

We organize the readings in six related sections. Together they cover studies of the natural and anthropogenic aspects of urban ecosystems. As one moves forward in time, they increasingly focus on inter-relations of people and nature where they co-occur in urban places.

In Section I: *Urbanization and Human Domination of Earth*, four papers show why Urban Ecology is an important and growing discipline. They detail the trends of increasing human domination of Earth generally, and of urbanization, specifically. They review the extent of urbanization and its

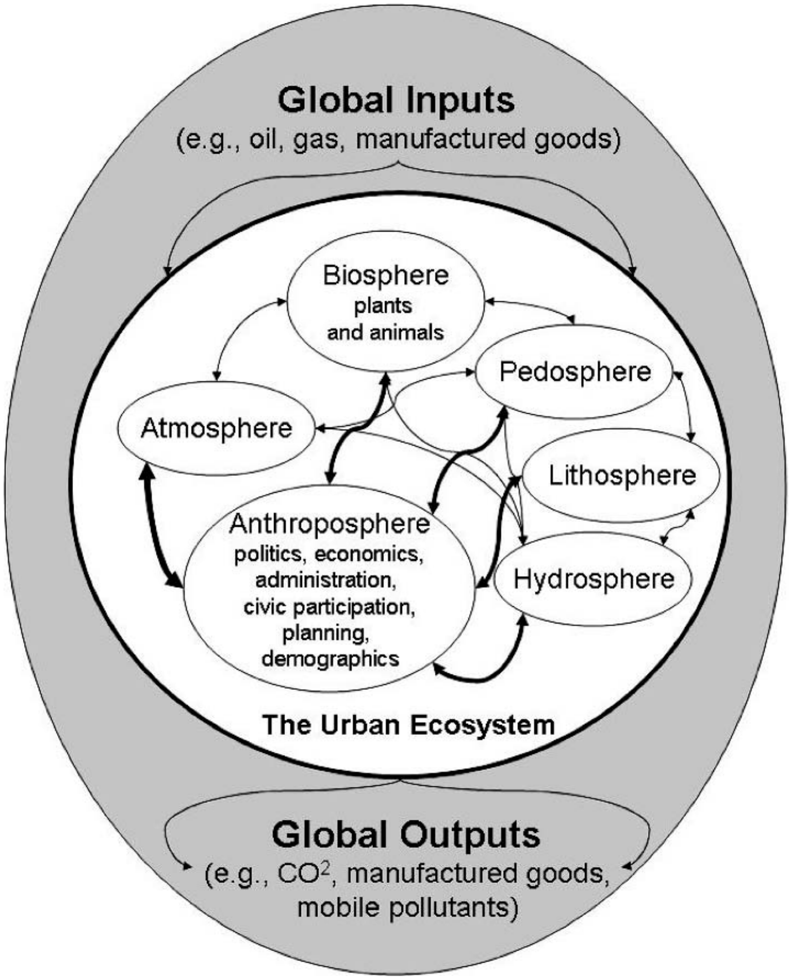

Fig. 1 Basic components of the urban ecosystem; some examples (arrows, width denotes relative magnitude) of relationships among them; and its global footprint (shaded area). The urban ecosystem, like other human-dominated systems, is characterized by the large direct and indirect of effects of the anthroposphere upon other biotic and abiotic systems and upon those systems' interactions

relation to other human endeavors that have changed the face of Earth. Together the papers examine the impacts of humans on basic ecological and evolutionary processes.

In Section II: *Conceptual Foundations of Urban Ecology,* five recent review articles report on the interdisciplinary synthesis that is modern Urban Ecology. Cities are viewed quite differently by early social ecologists like Burgess, by European pioneers like Sukopp, and by more recent emerging interdisciplinary American urban ecology teams. Again, as we move forward in time, cities are increasingly viewed as emergent phenomena with a new level of organization whose macroscopic properties and behavior are poorly predictable from knowledge of the properties of their constituent parts. This strongly suggests that cities are, in fact, an entirely new type of ecological entity - an entirely new level of complexity and organization - and that they must be studied as integrated systems. Precisely how these emergent phenomena have been studied has differed somewhat along disciplinary and geographic boundaries. Two major approaches in modern studies can be called "ecology *in* cities" and "ecology *of* cities" (Fig. 2). In Europe, we see a strong tendency to study the ecology of individual species or other taxa (but seldom *Homo sapiens*) within a city. The early

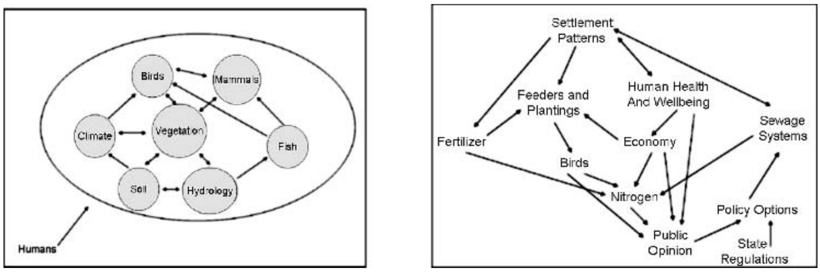

Fig. 2 An example of relationships in a typical study of ecology *in* the city (left) and ecology *of* the city (right)

American view championed by Burgess and others in Chicago focused on social processes and used ecological concepts to understand socio-economic dynamics in cities. On the other hand, early European urban ecology had been dominated by geographically constrained, traditional, autecological studies. It is only recently that urban ecology in both the USA and in Europe has expanded to include more complex interactions between human and natural systems across many scales.

In Section III: *The Atmosphere, Hydrosphere, and Pedosphere*, we present seven readings on the physical underpinnings of urban ecosystems. They cover the influence of urbanization on soils, hydrology, and climate, illustrate the effects of these changes on global climate and pollution, and consider various effects on human health.

Section IV: *The Biosphere* provides fourteen readings on specific environmental and ecological aspects of urban ecosystems. The studies cover ecological patterns, processes, and impacts, and link human effects on the abiotic environment to their effects on aquatic and terrestrial ecosystems. Research on birds in urban environments is highlighted because studies on the effects of urbanization on birds have a long and rich history in many parts of the world. We synthesize similarities and differences in how various animals and environments respond to urbanization while pointing out the importance of vegetative structure and composition (and therefore of soil formation and nutrient cycling) in helping to determine the structure of urban animal communities. The selections show how most studies *in* urban ecosystems focus on individuals, populations, or communities of plants and animals, while studies *of* urban ecosystems typically focus on ecosystem-level processes like nutrient cycling and energy flow.

Section V: *The Anthroposphere–Human Dimensions* explores the socio-economic aspects of urban ecosystems and links human settlement to ecosystem function, human health and well-being, and social justice through nine papers. Some important social science approaches are illustrated as the included authors investigate human decision-making and patterns of human settlement. Some articles explicitly consider sustainability of urban development at local and global scales. Our selections have been guided by our intent to synthesize social and economic drivers of various settlement patterns and to appraise the economic, ecological, and social sustainability of urban systems.

Section VI: *The Anthroposphere–Planning and Policy* presents eleven papers that together review the practical application of urban ecological knowledge to urban planning, conservation in and of urban ecosystems, and policy formulation. The suite of papers introduces the complexities of human urban institutions and the difficulties of managing them for local environmental health. They also present basic planning strategies intended to conserve or promote biological diversity in urbanizing areas. To help in making decisions at the policy and planning levels, these papers also discuss modeling strategies for appraising change in ecosystem function resulting from such decisions. Our synthesis builds on the readings to suggest general principles that would help increase the inclusion and use of urban ecological knowledge in the social arenas of planning and policy making

As we study the foundations of Urban Ecology, rarely do we see the various scholars in our field stand back and attempt to place cities into a larger ecological context. That larger-scale vision is now rapidly developing, and is the direction in which Urban Ecology, as a field, is clearly headed.

Cities are both drivers of, and driven by, ecological processes within and beyond their boundaries. It is no longer acceptable – indeed, it is highly counterproductive - to separate human and natural components in urban ecological studies. Cities are complex human phenomena, but they must be understood in new contexts:

First – cities should be studied both as social and biophysical phenomena. Like other complex phenomena, cities have generalizable and definable internal structures, functions, and processes that produce cities' emergent properties. Many of those internalities are not yet identified, much less understood.

Second – cities have incredibly large impacts on Earth's ecological processes, at all spatial scales yet studied, and with temporal scales yet to be determined. Cities have huge ecological "footprints" caused by their needs for goods, energy and services and their capacity to import natural resources from, while exporting their emissions and wastes to, distant regions. For these same reasons cities also have the potential to offer unique opportunities for resource conservation and environmental impact mitigation.

Our collection of writings suggests that there are at least three views of Urban Ecology as a field: (1) ecology and evolution of organisms that happen to live within city boundaries; (2) biological, political, economic, and cultural ecology of *Homo sapiens* in urban settings; (3) cities as emergent phenomena of coupled human and natural processes with implications for evolution and survival of our own and other species.

We believe that the third view – to which we ourselves subscribe – is the direction that the field can and must go. This view allows various aspects of the human enterprise and nature to be seen as interacting forces that shape measurable patterns and processes. Human factors are not isolated from other biotic or abiotic factors - together, as coupled human-natural systems, they both drive and are affected by the patterns and processes they create (Fig. 3). Ultimately our ability to build more resilient cities depends on a better understanding of the mechanisms that govern these interactions.

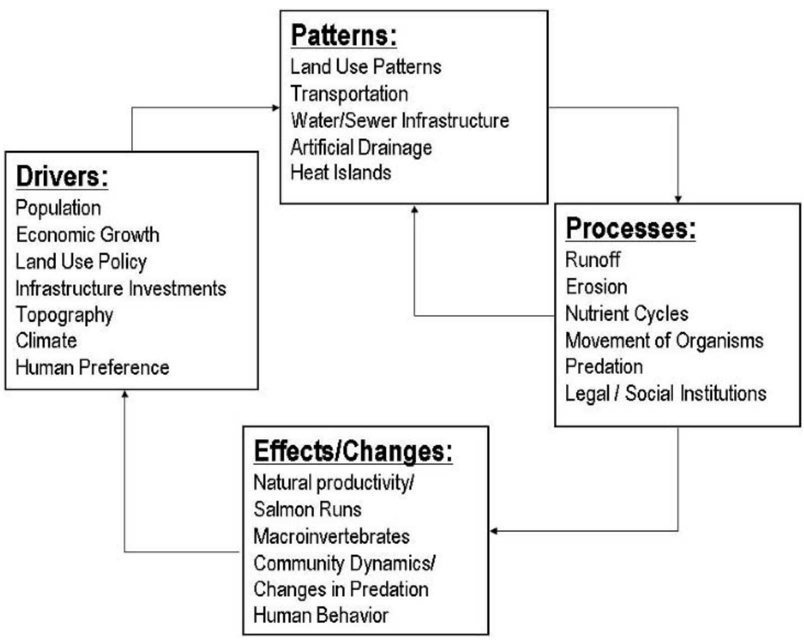

Fig. 3 A view of Urban Ecology that emphasizes the coupled relationships between humans and nature. Abiotic and biotic (anthropogenic and otherwise) drivers cause patterns and processes that urban ecologists measure. But these patterns and processes themselves affect interacting human and natural drivers by their effects and changes to the urban ecosystem

This is a productive time for the field of Urban Ecology. Groundwork by the many scholars represented in this reader has set the foundations for the development of new hypotheses about the similarities and differences of urban ecosystems and non-human-dominated ecosystems. There are many challenges and exciting discoveries to be made along this journey. Travelers will bring with them varied backgrounds, including theoretical and applied interests in the natural and social sciences. The field will advance as new theories are conceived, data are collected, simulation models are developed, and knowledge is used to inform planning and policy-making. We encourage theoreticians to fully conceptualize and model the complex web of interactions between humans and our ecosystem. Empiricists can test theory and document the dual nature of feedbacks amongst human health, economies, and cultures and the biotic and abiotic components of their ecosystems. Understanding coevolved relationships that often emerge from long-standing interactions between people and nature will be a special challenge that must be accomplished if a full understanding of the evolutionary aspects of urban ecology is to develop. Linking this basic knowledge to informed practice is perhaps the greatest challenge for Urban Ecology. Doing that will require a full integration of the natural and social sciences, careful distillation and application of complex ecological knowledge, and dedicated practitioners. These practitioners include planners, engineers, architects, landscape architects, and policy makers. Their actions quite literally shape our urban ecosystems. However these practitioners are also shaped by the responses of the ecosystem. We feel strongly that those who find this field interesting and who choose to participate in it through research or practice will benefit from understanding where the field has been and how it got to where it is today. Only then can we intelligently participate in moving the field forward. Providing that knowledge, and illustrating the theory, implications, and application of Urban Ecology is the purpose of this book.

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Section III The Atmosphere, Hydrosphere, and Pedosphere

Section IV The Biosphere

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Section I
Urbanization and Human Domination of Earth

This section documents the extraordinary impact of humans on Earth's ecosystems and discusses the implications of a human-dominated planet for ecology. The articles have been selected to provide the rationale and a context for Urban Ecology. Humans have transformed approximately one-third to one-half of Earth's land surface and use more than half of all accessible surface fresh water (Vitousek et al. 1997). Vitousek et al. (1997) provide a synthesis of key major trends indicating how planet Earth is changed as a result of human action. A few key trends are the nearly 30 percent increase in carbon dioxide concentration in the atmosphere since the beginning of the Industrial Revolution; the fixation of 60% of Earth's nitrogen by humanity; and the extinction of one-quarter of the bird species on Earth.

Palumbi (2001) shows us that humans are also affecting evolutionary trajectories, and hence dramatically accelerating evolutionary change in other species. One way humans effect evolutionary change is by introducing antibiotics, to which most bacterial diseases have evolved strong resistance (Palumbi 2001). The human-induced causes of evolutionary change also include the introduction of potent toxins, genetic engineering, generation and release into the environment of thousands of synthetic compounds, and selective fishing (Palumbi 2001). Rapid evolutionary change is becoming a hallmark of urban systems by adjusting the appearance and behavior of plants and animals to the novel pressures of urban life (Yeh 2004). The fact that evolution is visible and common in people's backyards may provide humans with an extremely clear window into a controversial issue (Marzluff in press). Palumbi (2001) argues that recognizing where possible the pace and pervasiveness of human dominated impacts on the planet's evolutionary trajectories is not only essential to mitigate the potential devastating impact that our species can have on others but can also greatly reduce the economic and social costs of evolution.

The dramatic environmental changes induced by humans are most evident in urbanizing landscapes. Berry's chapter from the *Earth as Transformed by Human Action*, published in 1990, provides an historical perspective on urbanization trends in the last 300 years. Berry asks when and where urban growth has happened and how can we explain the actual distribution and size of the cities worldwide. Berry synthesizes the main environmental impacts associated with urbanization known at the end of the 1980's for which, fifteen years later, there is increasing evidence and further understanding. But most important, Berry points to the different regional trajectories of urbanization and complex patterns that both natural and social scientists need to take into account when assessing the future human-induced environmental change. In particular Berry reminds us that urbanization is increasing most rapidly in the developing regions where the most dramatic environmental changes are still to come.

Measuring urbanization trends and their impacts on Earth's ecosystems requires a shared definition of "urban" between the natural and social sciences. In general, "urban" can be defined in terms of human population density and or dominant land cover. In the social sciences urban is typically defined based on population density. For example, the U.S. Bureau of the Census defines "urban" as an area with more than 2500 people (>620 individuals/km2). Ecologists on the other hand tend to use definitions in terms of dominant land cover. Based on these two dimensions, definitions of Urban, Suburban, Exurban, Rural and Wildland can be defined to represent different degrees of

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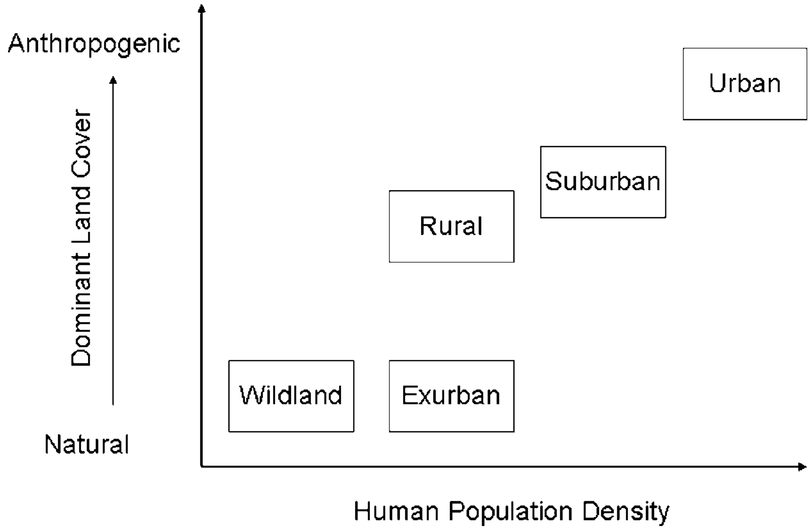

Fig. 1 A two dimensional definition of the urban-to-wildland gradient with commonly used categories of urbanization indicated

human domination on the landscape Mcintyre et al. (2000). This two-dimensional continuum could be represented as shown in Figure 1. While recognizing that no single definition of "urban" is possible, Mcintyre et al. (2000) propose that a quantitative description of what urban means is necessary and needs to take into account the dynamic and heterogeneous physical and social characteristics of an urban ecosystem. They propose to expand the dimensions to account for urban patterns, land uses, road network, governance and other factors. A more explicit and measurable definition of "urban" is considered critical to making progress in urban ecology.

The selection of the articles points out the challenges for the study of ecology that are posed by human domination of planet Earth, with increasing urbanization and associated resource consumption. As Vitousek et al. (1997) indicate, "...the global consequences of human activity are not something to face in the future - they are with us now." Furthermore most of these changes are accelerating. The planet's urban population has increased more than 10-fold over only the past century and most expected population growth in the next thirty years (approximately 2 billion people) will be concentrated in urban areas (UN 2003). But, in spite of the challenges that humans pose for Earth's ecosystems and the emerging need for a new paradigm for studying urban ecosystems, there is no shared theoretical framework to integrate fully humans into ecosystem studies. The changes imposed by humans on planet Earth clearly suggest that humans must be explicitly incorporated into ecological studies and equally clearly pose the question whether current ecological theory requires fundamental revisions to address human-dominated ecosystems.

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Human Domination of Earth's Ecosystems

Peter M. Vitousek, Harold A. Mooney, Jane Lubchenco, Jerry M. Melillo

Abstract Human alteration of Earth is substantial and growing. Between one-third and one-half of the land surface has been transformed by human action; the carbon dioxide concentration in the atmosphere has increased by nearly 30 percent since the beginning of the Industrial Revolution; more atmospheric nitrogen is fixed by humanity than by all natural terrestrial sources combined; more than half of all accessible surface fresh water is put to use by humanity; and about one-quarter of the bird species on Earth have been driven to extinction. By these and other standards, it is clear that we live on a human-dominated planet.

Keywords: human domination · extinction · carbon cycle · nitrogen cycle · global change · land cover change

All organisms modify their environment, and humans are no exception. As the human population has grown and the power of technology has expanded, the scope and nature of this modification has changed drastically. Until recently, the term "human-dominated ecosystems" would have elicited images of agricultural fields, pastures, or urban landscapes; now it applies with greater or lesser force to all of Earth. Many ecosystems are dominated directly by humanity, and no ecosystem on Earth's surface is free of pervasive human influence.

This article provides an overview of human effects on Earth's ecosystems. It is not intended as a litany of environmental disasters, though some disastrous situations are described; nor is it intended either to downplay or to celebrate environmental successes, of which there have been many. Rather, we explore how large humanity looms as a presence on the globe—how, even on the grandest scale, most aspects of the structure and functioning of Earth's ecosystems cannot be understood without accounting for the strong, often dominant influence of humanity.

We view human alterations to the Earth system as operating through the interacting processes summarized in Fig. 1. The growth of the human population, and growth in the resource base used by humanity, is maintained by a suite of human enterprises such as agriculture, industry, fishing, and international commerce. These enterprises transform the land surface (through cropping, forestry, and urbanization), alter the major biogeochemical cycles, and add or remove species and genetically distinct populations in most of Earth's ecosystems. Many of these changes are substantial and reasonably well quantified; all are ongoing. These relatively well-documented changes in turn entrain further alterations to the functioning of the Earth system, most notably by driving global climatic change [1] and causing irreversible losses of biological diversity [2].

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J.M. Marzluff et al., *Urban Ecology,*

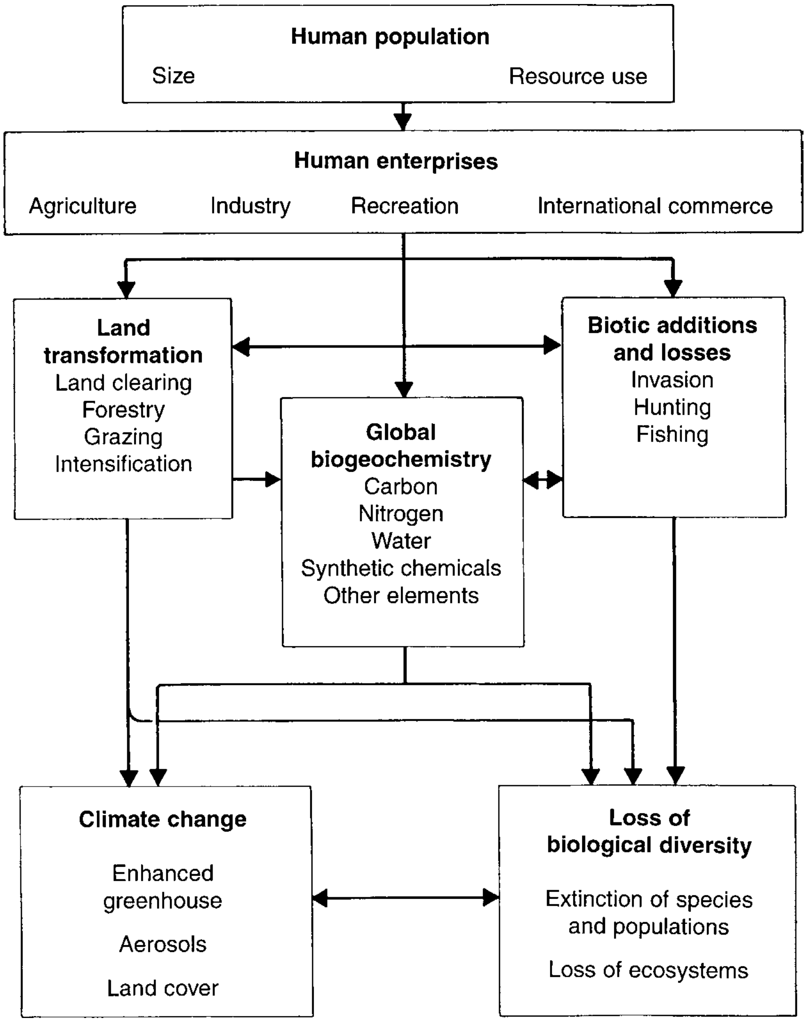

Fig. 1 A conceptual model illustrating humanity's direct and indirect effects on the Earth system [modified from [56]]

Land Transformation

The use of land to yield goods and services represents the most substantial human alteration of the Earth system. Human use of land alters the structure and functioning of ecosystems, and it alters how ecosystems interact with the atmosphere, with aquatic systems, and with surrounding land. Moreover, land transformation interacts strongly with most other components of global environmental change.

The measurement of land transformation on a global scale is challenging; changes can be measured more or less straightforwardly at a given site, but it is difficult to aggregate these changes regionally and globally. In contrast to analyses of human alteration of the global carbon cycle, we cannot install instruments on a tropical mountain to collect evidence of land transformation. Remote sensing is a most useful technique, but only recently has there been a serious scientific effort to use high-resolution civilian satellite imagery to evaluate even the more visible forms of land transformation, such as deforestation, on continental to global scales [3].

Land transformation encompasses a wide variety of activities that vary substantially in their intensity and consequences. At one extreme, 10 to 15% of Earth's land surface is occupied by row-crop agriculture or by urban-industrial areas, and another 6 to 8% has been converted to pastureland [4]; these systems are wholly changed by human activity. At the other extreme, every terrestrial ecosystem is affected by increased atmospheric carbon dioxide (CO_2), and most ecosystems have a history of hunting and other low-intensity resource extraction. Between these extremes lie grassland and semiarid ecosystems that are grazed (and sometimes degraded) by domestic animals, and forests and woodlands from which wood products have been harvested; together, these represent the majority of Earth's vegetated surface.

The variety of human effects on land makes any attempt to summarize land transformations globally a matter of semantics as well as substantial uncertainty. Estimates of the fraction of land transformed or degraded by humanity (or its corollary, the fraction of the land's biological production that is used or dominated) fall in the range of 39 to 50% [5] (Fig. 2). These numbers have large uncertainties, but the fact that they are large is not at all uncertain. Moreover, if anything these estimates understate the global impact of land transformation, in that land that has not been transformed often has been divided into fragments by human alteration of the surrounding areas. This fragmentation affects the species composition and functioning of otherwise little modified ecosystems [6].

Overall, land transformation represents the primary driving force in the loss of biological diversity worldwide. Moreover, the effects of land transformation extend far beyond the boundaries of transformed lands. Land transformation can affect climate directly at local and even regional scales. It contributes ~20% to current anthropogenic CO_2 emissions, and more substantially to the increasing concentrations of the greenhouse gases methane and nitrous oxide; fires associated with it alter the reactive chemistry of the troposphere, bringing elevated carbon monoxide concentrations and episodes of urban-like photochemical air pollution to remote tropical areas of Africa and South America; and it causes runoff of sediment and nutrients that drive substantial changes in stream, lake, estuarine, and coral reef ecosystems (7–10).

The central importance of land transformation is well recognized within the community of researchers concerned with global environmental change. Several research programs are focused on aspects of it [9, 11]; recent and substantial progress toward understanding these aspects has been made [3], and much more progress can be anticipated. Understanding land transformation is a difficult challenge; it requires integrating the social, economic, and cultural causes of land transformation with evaluations of its biophysical nature and consequences. This interdisciplinary approach is essential to predicting the course, and to any hope of affecting the consequences, of human-caused land transformation.

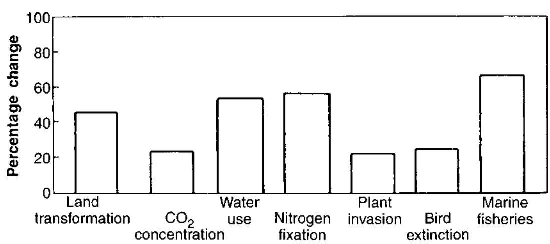

Fig. 2 Human dominance or alteration of several major components of the Earth system, expressed as (from left to right) percentage of the land surface transformed [5]; percentage of the current atmospheric CO_2 concentration that results from human action [17]; percentage of accessible surface fresh water used [20]; percentage of terrestrial N fixation that is human-caused [28]; percentage of plant species in Canada that humanity has introduced from elsewhere [48]; percentage of bird species on Earth that have become extinct in the past two millennia, almost all of them as a consequence of human activity [42]; and percentage of major marine fisheries that are fully exploited, overexploited, or depleted [14]

Oceans

Human alterations of marine ecosystems are more difficult to quantify than those of terrestrial ecosystems, but several kinds of information suggest that they are substantial. The human population is concentrated near coasts—about 60% within 100 km—and the oceans' productive coastal margins have been affected strongly by humanity. Coastal wetlands that mediate interactions between land and sea have been altered over large areas; for example, approximately 50% of mangrove ecosystems globally have been transformed or destroyed by human activity [12]. Moreover, a recent analysis suggested that although humans use about 8% of the primary production of the oceans, that fraction grows to more than 25% for upwelling areas and to 35% for temperate continental shelf systems [13].

Many of the fisheries that capture marine productivity are focused on top predators, whose removal can alter marine ecosystems out of proportion to their abundance. Moreover, many such fisheries have proved to be unsustainable, at least at our present level of knowledge and control. As of 1995, 22% of recognized marine fisheries were overexploited or already depleted, and 44% more were at their limit of exploitation [14] (Figs. 2 and 3). The consequences of fisheries are not restricted to their target organisms; commercial marine fisheries around the world discard 27 million tons of nontarget animals annually, a quantity nearly one-third as large as total landings [15]. Moreover, the dredges and trawls used in some fisheries damage habitats substantially as they are dragged along the sea floor.

A recent increase in the frequency, extent, and duration of harmful algal blooms in coastal areas [16] suggests that human activity has affected the base as well as the top of marine food chains. Harmful algal blooms are sudden increases in the abundance of marine phytoplankton that produce harmful structures or chemicals. Some but not all of these phytoplankton are strongly pigmented (red or brown tides). Algal blooms usually are correlated with changes in temperature, nutrients, or salinity; nutrients in coastal waters, in particular, are much modified by human activity. Algal blooms can cause extensive fish kills through toxins and by causing anoxia; they also lead to paralytic shellfish poisoning and amnesic shellfish poisoning in humans. Although the existence of harmful algal blooms has long been recognized, they have spread widely in the past two decades [16].

Alterations of the Biogeochemical Cycles

Carbon. Life on Earth is based on carbon, and the CO_2 in the atmosphere is the primary resource for photosynthesis. Humanity adds CO_2 to the atmosphere by mining and burning fossil fuels, the residue of life from the distant past, and by converting forests and grasslands to agricultural and other low-biomass ecosystems. The net result of both activities is that organic carbon from rocks, organisms, and soils is released into the atmosphere as CO_2.

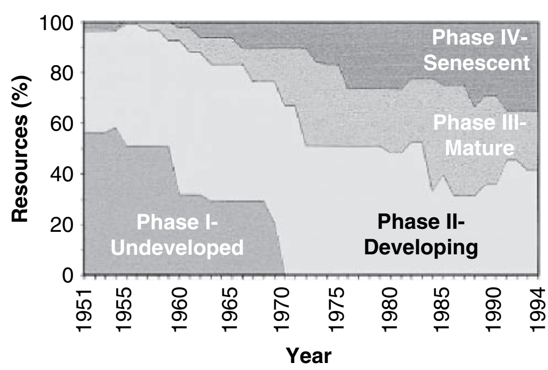

Fig. 3 Percentage of major world marine fish resources in different phases of development, 1951 to 1994 [from [57]]. Undeveloped = a low and relatively constant level of catches; developing = rapidly increasing catches; mature = a high and plateauing level of catches; senescent = catches declining from higher levels

The modern increase in CO_2 represents the clearest and best documented signal of human alteration of the Earth system. Thanks to the foresight of Roger Revelle, Charles Keeling, and others who initiated careful and systematic measurements of atmospheric CO_2 in 1957 and sustained them through budget crises and changes in scientific fashions, we have observed the concentration of CO_2 as it has increased steadily from 315 ppm to 362 ppm. Analysis of air bubbles extracted from the Antarctic and Greenland ice caps extends the record back much further; the CO_2 concentration was more or less stable near 280 ppm for thousands of years until about 1800, and has increased exponentially since then [17].

There is no doubt that this increase has been driven by human activity, today primarily by fossil fuel combustion. The sources of CO_2 can be traced isotopically; before the period of extensive nuclear testing in the atmosphere, carbon depleted in ^{14}C was a specific tracer of CO_2 derived from fossil fuel combustion, whereas carbon depleted in ^{13}C characterized CO_2 from both fossil fuels and land transformation. Direct measurements in the atmosphere, and analyses of carbon isotopes in tree rings, show that both ^{13}C and ^{14}C in CO_2 were diluted in the atmosphere relative to ^{12}C as the CO_2 concentration in the atmosphere increased.

Fossil fuel combustion now adds 5.5 ± 0.5 billion metric tons of CO_2-C to the atmosphere annually, mostly in economically developed regions of the temperate zone [18] (Fig. 4). The annual accumulation of CO_2-C has averaged 3.2 ± 0.2 billion metric tons recently [17]. The other major terms in the atmospheric carbon balance are net ocean-atmosphere flux, net release of carbon during land transformation, and net storage in terrestrial biomass and soil organic matter. All of these terms are smaller and less certain than fossil fuel combustion or annual atmospheric accumulation; they represent rich areas of current research, analysis, and sometimes contention.

The human-caused increase in atmospheric CO_2 already represents nearly a 30% change relative to the pre-industrial era (Fig. 2), and CO_2 will continue to increase for the foreseeable future. Increased CO_2 represents the most important human enhancement to the greenhouse effect; the consensus of the climate research community is that it probably already affects climate detectably

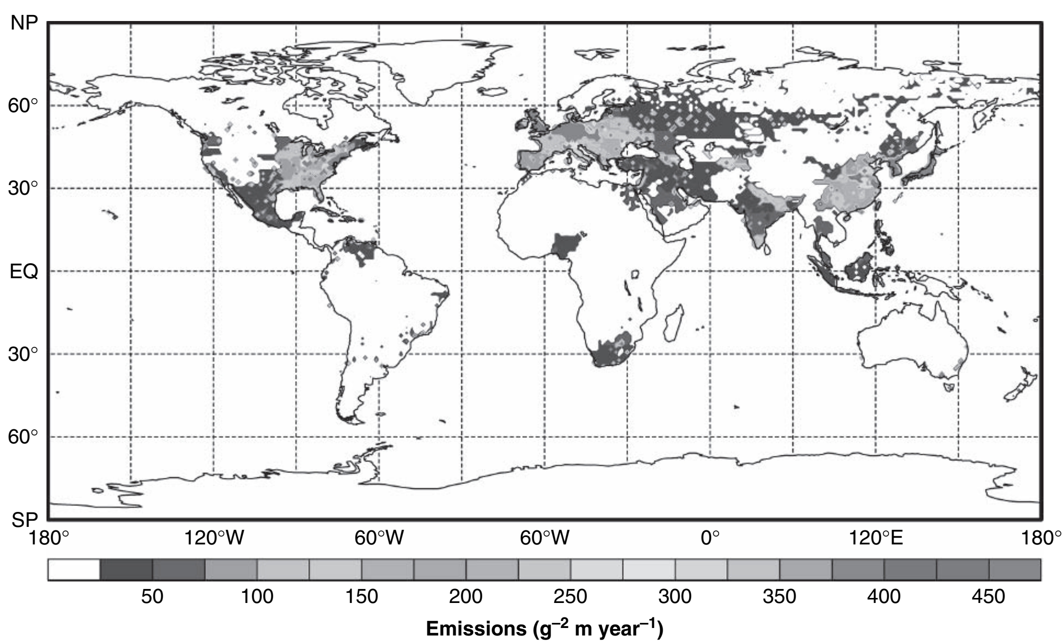

Fig. 4 Geographical distribution of fossil fuel sources of CO_2 as of 1990. The global mean is 12.2 g m^{-2} year^{-1}; most emissions occur in economically developed regions of the north temperate zone. EQ, equator; NP, North Pole; SP, South Pole. [Prepared by A. S. Denning, from information in [18]]

and will drive substantial climate change in the next century [1]. The direct effects of increased CO_2 on plants and ecosystems may be even more important. The growth of most plants is enhanced by elevated CO_2, but to very different extents; the tissue chemistry of plants that respond to CO_2 is altered in ways that decrease food quality for animals and microbes; and the water use efficiency of plants and ecosystems generally is increased. The fact that increased CO_2 affects species differentially means that it is likely to drive substantial changes in the species composition and dynamics of all terrestrial ecosystems [19].

Water. Water is essential to all life. Its movement by gravity, and through evaporation and condensation, contributes to driving Earth's biogeochemical cycles and to controlling its climate. Very little of the water on Earth is directly usable by humans; most is either saline or frozen. Globally, humanity now uses more than half of the runoff water that is fresh and reasonably accessible, with about 70% of this use in agriculture [20] (Fig. 2). To meet increasing demands for the limited supply of fresh water, humanity has extensively altered river systems through diversions and impoundments. In the United States only 2% of the rivers run unimpeded, and by the end of this century the flow of about two-thirds of all of Earth's rivers will be regulated [21]. At present, as much as 6% of Earth's river runoff is evaporated as a consequence of human manipulations [22]. Major rivers, including the Colorado, the Nile, and the Ganges, are used so extensively that little water reaches the sea. Massive inland water bodies, including the Aral Sea and Lake Chad, have been greatly reduced in extent by water diversions for agriculture. Reduction in the volume of the Aral Sea resulted in the demise of native fishes and the loss of other biota; the loss of a major fishery; exposure of the salt-laden sea bottom, thereby providing a major source of windblown dust; the production of a drier and more continental local climate and a decrease in water quality in the general region; and an increase in human diseases [23].

Impounding and impeding the flow of rivers provides reservoirs of water that can be used for energy generation as well as for agriculture. Waterways also are managed for transport, for flood control, and for the dilution of chemical wastes. Together, these activities have altered Earth's freshwater ecosystems profoundly, to a greater extent than terrestrial ecosystems have been altered. The construction of dams affects biotic habitats indirectly as well; the damming of the Danube River, for example, has altered the silica chemistry of the entire Black Sea. The large number of operational dams (36,000) in the world, in conjunction with the many that are planned, ensure that humanity's effects on aquatic biological systems will continue [24]. Where surface water is sparse or over-exploited, humans use groundwater—and in many areas the groundwater that is drawn upon is nonrenewable, or fossil, water [25]. For example, three-quarters of the water supply of Saudi Arabia currently comes from fossil water [26].

Alterations to the hydrological cycle can affect regional climate. Irrigation increases atmospheric humidity in semiarid areas, often increasing precipitation and thunderstorm frequency [27]. In contrast, land transformation from forest to agriculture or pasture increases albedo and decreases surface roughness; simulations suggest that the net effect of this transformation is to increase temperature and decrease precipitation regionally [7, 26].

Conflicts arising from the global use of water will be exacerbated in the years ahead, with a growing human population and with the stresses that global changes will impose on water quality and availability. Of all of the environmental security issues facing nations, an adequate supply of clean water will be the most important.

Nitrogen. Nitrogen (N) is unique among the major elements required for life, in that its cycle includes a vast atmospheric reservoir (N_2) that must be fixed (combined with carbon, hydrogen, or oxygen) before it can be used by most organisms. The supply of this fixed N controls (at least in part) the productivity, carbon storage, and species composition of many ecosystems. Before the extensive human alteration of the N cycle, 90 to 130 million metric tons of N (Tg N) were fixed biologically on land each year; rates of biological fixation in marine systems are less certain, but perhaps as much was fixed there [28].

Human activity has altered the global cycle of N substantially by fixing N_2—deliberately for fertilizer and inadvertently during fossil fuel combustion. Industrial fixation of N fertilizer increased from <10 Tg/year in 1950 to 80 Tg/year in 1990; after a brief dip caused by economic dislocations in the former Soviet Union, it is expected to increase to >135 Tg/year by 2030 [29]. Cultivation of soybeans, alfalfa, and other legume crops that fix N symbiotically enhances fixation by another ~40 Tg/year, and fossil fuel combustion puts >20 Tg/year of reactive N into the atmosphere globally—some by fixing N_2, more from the mobilization of N in the fuel. Overall, human activity adds at least as much fixed N to terrestrial ecosystems as do all natural sources combined (Fig. 2), and it mobilizes 50 Tg/year more during land transformation [28, 30].

Alteration of the N cycle has multiple consequences. In the atmosphere, these include (i) an increasing concentration of the greenhouse gas nitrous oxide globally; (ii) substantial increases in fluxes of reactive N gases (two-thirds or more of both nitric oxide and ammonia emissions globally are human-caused); and (iii) a substantial contribution to acid rain and to the photochemical smog that afflicts urban and agricultural areas throughout the world [31]. Reactive N that is emitted to the atmosphere is deposited downwind, where it can influence the dynamics of recipient ecosystems. In regions where fixed N was in short supply, added N generally increases productivity and C storage within ecosystems, and ultimately increases losses of N and cations from soils, in a set of processes termed "N saturation" [32]. Where added N increases the productivity of ecosystems, usually it also decreases their biological diversity [33].

Human-fixed N also can move from agriculture, from sewage systems, and from N-saturated terrestrial systems to streams, rivers, groundwater, and ultimately the oceans. Fluxes of N through streams and rivers have increased markedly as human alteration of the N cycle has accelerated; river nitrate is highly correlated with the human population of river basins and with the sum of human-caused N inputs to those basins [8]. Increases in river N drive the eutrophication of most estuaries, causing blooms of nuisance and even toxic algae, and threatening the sustainability of marine fisheries [16, 34].

Other cycles. The cycles of carbon, water, and nitrogen are not alone in being altered by human activity. Humanity is also the largest source of oxidized sulfur gases in the atmosphere; these affect regional air quality, biogeochemistry, and climate. Moreover, mining and mobilization of phosphorus and of many metals exceed their natural fluxes; some of the metals that are concentrated and mobilized are highly toxic (including lead, cadmium, and mercury) [35]. Beyond any doubt, humanity is a major biogeochemical force on Earth.

Synthetic organic chemicals. Synthetic organic chemicals have brought humanity many beneficial services. However, many are toxic to humans and other species, and some are hazardous in concentrations as low as 1 part per billion. Many chemicals persist in the environment for decades; some are both toxic and persistent. Long-lived organochlorine compounds provide the clearest examples of environmental consequences of persistent compounds. Insecticides such as DDT and its relatives, and industrial compounds like polychlorinated biphenyls (PCBs), were used widely in North America in the 1950s and 1960s. They were transported globally, accumulated in organisms, and magnified in concentration through food chains; they devastated populations of some predators (notably falcons and eagles) and entered parts of the human food supply in concentrations higher than was prudent. Domestic use of these compounds was phased out in the 1970s in the United States and Canada, and their concentrations declined thereafter. However, PCBs in particular remain readily detectable in many organisms, sometimes approaching thresholds of public health concern [36]. They will continue to circulate through organisms for many decades.

Synthetic chemicals need not be toxic to cause environmental problems. The fact that the persistent and volatile chlorofluorocarbons (CFCs) are wholly nontoxic contributed to their widespread use as refrigerants and even aerosol propellants. The subsequent discovery that CFCs drive the breakdown of stratospheric ozone, and especially the later discovery of the Antarctic ozone hole and their role in it, represent great surprises in global environmental science [37]. Moreover, the response of the international political system to those discoveries is the best extant illustration that global environmental change can be dealt with effectively [38].

Particular compounds that pose serious health and environmental threats can be and often have been phased out (although PCB production is growing in Asia). Nonetheless, each year the chemical industry produces more than 100 million tons of organic chemicals representing some 70,000 different compounds, with about 1000 new ones being added annually [39]. Only a small fraction of the many chemicals produced and released into the environment are tested adequately for health hazards or environmental impact [40].

Biotic Changes

Human modification of Earth's biological resources—its species and genetically distinct populations—is substantial and growing. Extinction is a natural process, but the current rate of loss of genetic variability, of populations, and of species is far above background rates; it is ongoing; and it represents a wholly irreversible global change. At the same time, human transport of species around Earth is homogenizing Earth's biota, introducing many species into new areas where they can disrupt both natural and human systems.

Losses. Rates of extinction are difficult to determine globally, in part because the majority of species on Earth have not yet been identified. Nevertheless, recent calculations suggest that rates of species extinction are now on the order of 100 to 1000 times those before humanity's dominance of Earth [41]. For particular well-known groups, rates of loss are even greater; as many as one-quarter of Earth's bird species have been driven to extinction by human activities over the past two millennia, particularly on oceanic islands [42] (Fig. 2). At present, 11% of the remaining birds, 18% of the mammals, 5% of fish, and 8% of plant species on Earth are threatened with extinction [43]. There has been a disproportionate loss of large mammal species because of hunting; these species played a dominant role in many ecosystems, and their loss has resulted in a fundamental change in the dynamics of those systems [44], one that could lead to further extinctions. The largest organisms in marine systems have been affected similarly, by fishing and whaling. Land transformation is the single most important cause of extinction, and current rates of land transformation eventually will drive many more species to extinction, although with a time lag that masks the true dimensions of the crisis [45]. Moreover, the effects of other components of global environmental change—of altered carbon and nitrogen cycles, and of anthropogenic climate change—are just beginning.

As high as they are, these losses of species understate the magnitude of loss of genetic variation. The loss to land transformation of locally adapted populations within species, and of genetic material within populations, is a human-caused change that reduces the resilience of species and ecosystems while precluding human use of the library of natural products and genetic material that they represent [46].

Although conservation efforts focused on individual endangered species have yielded some successes, they are expensive—and the protection or restoration of whole ecosystems often represents the most effective way to sustain genetic, population, and species diversity. Moreover, ecosystems themselves may play important roles in both natural and human-dominated landscapes. For example, mangrove ecosystems protect coastal areas from erosion and provide nurseries for offshore fisheries, but they are threatened by land transformation in many areas.

Invasions. In addition to extinction, humanity has caused a rearrangement of Earth's biotic systems, through the mixing of floras and faunas that had long been isolated geographically. The magnitude of transport of species, termed "biological invasion," is enormous [47]; invading species are present almost everywhere. On many islands, more than half of the plant species are nonindigenous, and in many continental areas the figure is 20% or more [48] (Fig. 2).

As with extinction, biological invasion occurs naturally—and as with extinction, human activity has accelerated its rate by orders of magnitude. Land transformation interacts strongly with biological invasion, in that human-altered ecosystems generally provide the primary foci for invasions, while in some cases land transformation itself is driven by biological invasions [49]. International commerce is also a primary cause of the breakdown of biogeographic barriers; trade in live organisms is massive and global, and many other organisms are inadvertently taken along for the ride. In freshwater systems, the combination of upstream land transformation, altered hydrology, and numerous deliberate and accidental species introductions has led to particularly widespread invasion, in continental as well as island ecosystems [50].

In some regions, invasions are becoming more frequent. For example, in the San Francisco Bay of California, an average of one new species has been established every 36 weeks since 1850, every 24 weeks since 1970, and every 12 weeks for the last decade [51]. Some introduced species quickly become invasive over large areas (for example, the Asian clam in the San Francisco Bay), whereas others become widespread only after a lag of decades, or even over a century [52].

Many biological invasions are effectively irreversible; once replicating biological material is released into the environment and becomes successful there, calling it back is difficult and expensive at best. Moreover, some species introductions have consequences. Some degrade human health and that of other species; after all, most infectious diseases are invaders over most of their range. Others have caused economic losses amounting to billions of dollars; the recent invasion of North America by the zebra mussel is a well-publicized example. Some disrupt ecosystem processes, altering the structure and functioning of whole ecosystems. Finally, some invasions drive lossess in the biological diversity of native species and populations; after land transformation, they are the next most important cause of extinction [53].

Conclusions

The global consequences of human activity are not something to face in the future—as Fig. 2 illustrates, they are with us now. All of these changes are ongoing, and in many cases accelerating; many of them were entrained long before their importance was recognized. Moreover, all of these seemingly disparate phenomena trace to a single cause—the growing scale of the human enterprise. The rates, scales, kinds, and combinations of changes occurring now are fundamentally different from those at any other time in history; we are changing Earth more rapidly than we are understanding it. We live on a human-dominated planet—and the momentum of human population growth, together with the imperative for further economic development in most of the world, ensures that our dominance will increase.

The papers in this special section summarize our knowledge of and provide specific policy recommendations concerning major human-dominated ecosystems. In addition, we suggest that the rate and extent of human alteration of Earth should affect how we think about Earth. It is clear that we control much of Earth, and that our activities affect the rest. In a very real sense, the world is in our hands—and how we handle it will determine its composition and dynamics, and our fate.

Recognition of the global consequences of the human enterprise suggests three complementary directions. First, we can work to reduce the rate at which we alter the Earth system. Humans and human-dominated systems may be able to adapt to slower change, and ecosystems and the species they support may cope more effectively with the changes we impose, if those changes are slow. Our footprint on the planet [54] might then be stabilized at a point where enough space and resources remain to sustain most of the other species on Earth, for their sake and our own. Reducing the rate of growth in human effects on Earth involves slowing human population growth and using resources as efficiently as is practical. Often it is the waste products and by-products of human activity that drive global environmental change.

Second, we can accelerate our efforts to understand Earth's ecosystems and how they interact with the numerous components of human-caused global change. Ecological research is inherently complex and demanding: It requires measurement and monitoring of populations and ecosystems; experimental studies to elucidate the regulation of ecological processes; the development, testing, and validation of regional and global models; and integration with a broad range of biological, earth, atmospheric, and marine sciences. The challenge of understanding a human-dominated planet further requires that the human dimensions of global change—the social, economic, cultural, and other drivers of human actions—be included within our analyses.

Finally, humanity's dominance of Earth means that we cannot escape responsibility for managing the planet. Our activities are causing rapid, novel, and substantial changes to Earth's ecosystems. Maintaining populations, species, and ecosystems in the face of those changes, and maintaining the flow of goods and services they provide humanity [55], will require active management for the foreseeable future. There is no clearer illustration of the extent of human dominance of Earth than the fact that maintaining the diversity of "wild" species and the functioning of "wild" ecosystems will require increasing human involvement.

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Humans as the World's Greatest Evolutionary Force

Stephen R. Palumbi

Abstract In addition to altering global ecology, technology and human population growth also affect evolutionary trajectories, dramatically accelerating evolutionary change in other species, especially in commercially important, pest, and disease organisms. Such changes are apparent in antibiotic and human immunodeficiency virus (HIV) resistance to drugs, plant and insect resistance to pesticides, rapid changes in invasive species, life-history change in commercial fisheries, and pest adaptation to biological engineering products. This accelerated evolution costs at least \$33 billion to \$50 billion a year in the United States. Slowing and controlling arms races in disease and pest management have been successful in diverse ecological and economic systems, illustrating how applied evolutionary principles can help reduce the impact of human-kind on evolution.

Keywords: evolution · antibiotic · resistance · speed of evolution · human domination · economics

Human impact on the global biosphere now controls many major facets of ecosystem function. Currently, a large fraction of the world's available fresh water, arable land, fisheries production, nitrogen budget, CO_2 balance, and biotic turnover are dominated by human effects [1]. Human ecological impact has enormous evolutionary consequences as well and can greatly accelerate evolutionary change in the species around us, especially disease organisms, agricultural pests, commensals, and species hunted commercially. For example, some forms of bacterial infection are insensitive to all but the most powerful antibiotics, yet these infections are increasingly common in hospitals [2]. Some insects are tolerant of so many different insecticides that chemical control is useless [3]. Such examples illustrate the pervasive intersection of biological evolution with human life, effects that generate substantial daily impacts and produce increasing economic burden.

Accelerated evolutionary changes are easy to understand—they derive from strong natural selection exerted by human technology. However, technological impact has increased so markedly over the past few decades that humans may be the world's dominant evolutionary force. The importance of human-induced evolutionary change can be measured economically, in some cases, and is frequently seen in the exposure of societies to uncontrollable disease or pest outbreaks. Attempts to slow these evolutionary changes are widespread but uncoordinated. How well do they work to slow evolution? Can successes from one field be generalized to others?

The Pace of Human-Induced Evolution

Paul Müller's 1939 discovery that DDT killed insects won him the 1948 Nobel Prize, but before the Nobel ceremony occurred, evolution of resistance had already been reported in house flies [3, 4]. By the 1960s, mosquitoes resistant to DDT effectively prevented the worldwide eradication of

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Table 1 Dates of deployment of representative antibiotics and herbicides, and the evolution of resistance. Source [75]

Evolution of Resistance to Antibiotics and Herbicides		
Antibiotic or herbicide	Year deployed	Resistance observed
	Antibiotics	
Sulfonamides	1930s	1940s
Penicillin	1943	1946
Streptomycin	1943	1959
Chloramphenicol	1947	1959
Tetracycline	1948	1953
Erythromycin	1952	1988
Vancomycin	1956	1988
Methicillin	1960	1961
Ampicillin	1961	1973
Cephalosporins	1960s	late 1960s
	Herbicides	
2,4-D	1945	1954
Dalapon	1953	1962
Atrazine	1958	1968
Picloram	1963	1988
Trifluralin	1963	1988
Triallate	1964	1987
Diclofop	1980	1987

malaria [5], and by 1990, over 500 species had evolved resistance to at least one insecticide [6]. Insects often evolve resistance within about a decade after introduction of a new pesticide [7], and many species are resistant to so many pesticides that they are difficult or impossible to control [3]. Similar trajectories are known for resistant weeds [8], which typically evolve resistance within 10 to 25 years of deployment of an herbicide (Table 1).

Bacterial diseases have evolved strong and devastating resistance to many antibiotics. This occurs at low levels in natural populations [9] but can become common within a few years of the commercial adoption of a new drug (Table 1). For example, virtually all Gram-positive infections were susceptible to penicillin in the 1940s [2, 10] but in hospitals today, the vast majority of infections caused by important bacterial agents like *Staphylococcus aureus* are penicillin-resistant, and up to 50% are resistant to stronger drugs like methicillin [11]. Treatments that used to require small antibiotic doses now require huge concentrations or demand powerful new drugs [10]. But such solutions are short-lived. For example, vancomycin, one of the only treatments for methicillin-resistant infections, has been overcome by some of the most frequent infectious agents in hospitals [2, 12]. Antibiotics also generate evolution outside hospitals. Resistant strains are common on farms that use antibiotics in livestock production [13] and have been found in soils and groundwater affected by farm effluents [14].

Retroviruses with RNA genomes evolve even more quickly than bacteria [15]. Every year, vaccinations against influenza must be reformulated, making prediction of next year's viral fashion one of preventative medicine's chief challenges [16]. The virus that causes AIDS, human immunodeficiency virus–1, evolves so quickly that the infection within a single person becomes a quasi-species consisting of thousands of evolutionary variants [15]. Over the course of months or years after HIV infection, the virus continually evolves away from immune system suppression [17, 18]. Evolution in the face of antiviral drugs is just as rapid. For example, the drug nevirapine reduces viral RNA levels for only about 2 weeks [19]. Thereafter, mutations in the HIV reverse transcriptase gene quickly arise that confer drug resistance, and the HIV mutants have a doubling time of 2 to 6 days [19]. This rapid evolution is repeated with virtually all other antiretroviral drugs when given singly, including the inexpensive antiviral drugs zidovudine (azidothymine, AZT), lamivudine (3TC), didanosine (ddI) and protease inhibitors like indinavir (20–24).

Rapid evolution caused by humans is not restricted to disease or pest species. Under heavy fishing pressure, fish evolve slower growth rates and thinner bodies, allowing them to slip through gill nets [25, 26]. In hatchery populations of salmon, there is strong selection for dwarf males that return from sea early, increasing their survival [25]. Invading species, transported by humans, have been known to rapidly change to match local selection pressures [27]. For instance, house sparrows, introduced to North America in 1850, are now discernibly different in body size and color throughout the United States [28]. In some cases, species introduced by humans induce evolution in species around them. For example, after the subtidal snail *Littorina littorea* invaded coastal New England in the late 1800s, native hermit crabs [*Pagurus longicarpus* (Say)] quickly evolved behavioral preference for their shells. The crabs also evolved body and claw changes that fit them more securely in these new, larger shells [29]. Even more quickly, introduced predatory fish have caused rapid evolution of life-history traits and color pattern in their prey species [30, 31]. Rates of human-mediated evolutionary change sometimes exceed rates of natural evolution by orders of magnitude [30].

Causes of Evolution

These examples demonstrate pervasive and rapid evolution as a result of human activity. In most cases, the causes of this evolutionary pattern are clear: if a species is variable for a trait, and that trait confers a difference in survival or production of offspring, and the trait difference is heritable by offspring, then all three requirements of evolution by natural selection are present. In such cases, the evolutionary engine can turn, although evolutionary directions and speed can be influenced by factors such as drift, conflicting selection pressure, and correlated characters [31].

The overwhelming impact of humans on evolution stems from the ecological role we now play in the world, and the industrialization of our agriculture, medicine, and landscape. Successful pesticides or antibiotics are often produced in massive quantities. DDT, for example, was first used by the Allied Army in Naples in 1943, but by the end of World War II, DDT production was proceeding on an industrial scale. Currently, we use about 700 million pounds of pesticide a year in the United States [7]. Antibiotic production is also high, with 25 to 50% going into prophylactic use in live-stock feed [13].

Inefficient use of antibiotics has been cited as a major cause of antibiotic resistance. Partial treatment of infections with suboptimal doses leads to partial control of the infecting cell population and creates a superb environment for the evolution of resistant bacteria. Up to one-third of U.S. pediatricians report overprescribing antibiotics to assuage patient concerns, particularly in cases of viral childhood congestions that cannot respond to the drug [32]. Failing to complete a course of antibiotics is associated with increased emergence of resistant tuberculosis and HIV infections [33, 34], and differences in antibiotic use may partly explain differences among nations in antibiotic resistance rates [2].

Spread of antibiotic resistance has been accelerated by transmission of genes between bacterial species [13]. Recently, biotechnology has applied this acceleration to other species as well, and a new human-mediated mechanism for generating evolutionary novelty has emerged—insertion of exogenous genes into domesticated plants and animals. Taken from bacteria, plants, animals, or fungi, these genes convey valuable commercial traits, and they are placed into new host genomes along with genes that control expression and in some cases allow cell lineage selection [35, 36]. Examples include the insertion of genes for insecticidal proteins [37], herbicide tolerance [38, 39] or novel vitamins [40] into crop plants; growth hormone genes into farmed salmon [41]; and hormone production genes into livestock "bioreactors" [42]. These efforts effectively increase the rate of generation of new traits—akin to increasing the rate of macromutation. When these traits cross from domesticated into wild species, they can add to the fuel of evolution and allow rapid spread of the traits in natural populations [43]. Genetic exchange from crops has already enhanced the weediness of wild relatives of 7 of the world's 13 most important crop plants [44], although no widespread escape of an engineered gene into the wild has been reported yet.

The Economics of Human-Induced Evolution

Evolution is responsible for large costs when pests or disease organisms escape from chemical control. Farmers spend an estimated $12 billion on pesticides per year in the United States [7]. Extra costs due to pest resistance, such as respraying fields, may account for about 10% of these direct expenditures [45, 46]. Despite the heavy use of chemical pesticides, 10 to 35% of U.S. farm production is lost to pest damage [45]. If even 10% of this loss is due to activities of resistant insects (and the figure may be far higher), this represents a $2 billion to $7 billion yearly loss for the $200 billion U.S. food industry. The development of resistance in diamondback moths to *Bacillus thuringiensis* (Bt) toxin in 1989 [47] foreshadows the decline in use of the world's largest selling biopesticide and the need for new approaches (Fig. 1). The price of developing a single new pesticide, about $80 million in 1999 [7], is an ongoing cost of agricultural business. Even higher development costs (about $150 million per product) are incurred by pharmaceutical companies [p. 157 in [7]]. In both sectors, evolution sparks an arms race between human chemical control and pest or disease agent, dramatically increasing costs that are eventually paid by consumers [7, 11]. For example, the new drugs linezolid and quinupristin-dalfopristin were recently approved by the U.S. Food and Drug Administration (FDA) for use on vancomycin-resistant infections [48]. Previously, vancomycin had been used to overcome methicillin resistance [10], and methicillin was itself a response to the failure of penicillin treatment [13]. This development cascade (Fig. 2) has been on going since the birth of the chemical-control era and represents a poorly quantified cost of evolution.

More direct expenses stem from the increase in drug payments and hospitalization necessary to treat resistant diseases. There are approximately 2 million hospital-acquired infections in the

Fig. 1 In this field of water cress, the world's biggest selling biopesticide, *Bacillus thuringiensis* (Bt) toxin, was overcome by the evolution of resistance in diamondback moths. This pesticide is engineered into millions of acres of crop plants, and so the ability of insects to evolve resistance has created anxiety in the biotechnology industry

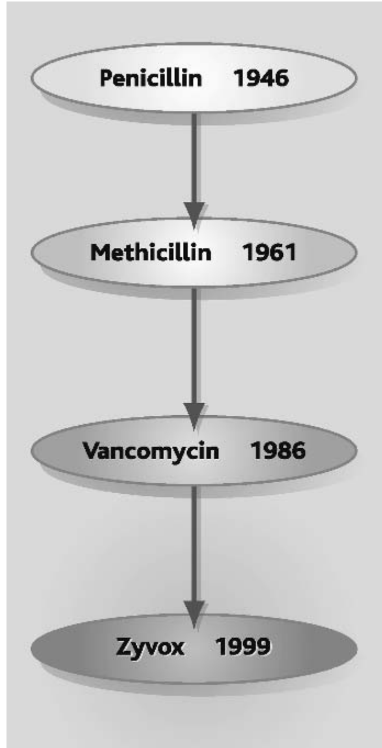

Fig. 2 Developmental cascade of antibiotics used to treat dangerous *Staphylococcus* infections. Dates reflect evolution of resistance to each drug, requiring search for more powerful alternatives

United States each year [data from 1995 [11, 49]], a quarter of which are caused by antibiotic-resistant *S. aureus* [2]. Half of these are penicillin-resistant strains that require treatment with methicillin at a cost of $2 billion to $7 billion [11, 49]. The other half are methicillin-resistant infections, and they cost hospitals an estimated $8 billion per year to cure [11]. Community-acquired, antibiotic-resistant staph infections more than double these costs [49, 50]. These figures are for a single type of infection and do not include other well-known drug-resistant bacteria. For example, in the United States up to 22% of hospital-acquired infections of *Enterococcus faecium* are resistant to vancomycin, and combating such infections drives the price of evolution even higher.

Similar conservative tabulations can be made for the cost of HIV treatment. The current standard of care in the United States is to treat HIV with massive doses of at least three drugs [51]. Because treatment with the inexpensive antiretroviral drug AZT would successfully halt HIV if it did not evolve resistance, the need for more powerful drugs is due to HIV evolution. Drug and treatment prices vary but have recently been estimated at $18,300 per year per patient in the United States [52]. If half the 700,000 HIV patients [53] in the United States receive this level of care, these costs amount to $6.3 billion per year [52]. Costs of lost labor, disruption of health services, development of new drugs, and medical research are not included in this figure, and so the actual cost of HIV evolution is far higher.

The annual evolution bill in the United States approaches $50 billion for these examples (Table 2), and probably exceeds $100 billion overall. However, the social price of evolution is far higher. Skyrocketing costs of treating resistant diseases create a situation where effective medical treatment may be economically unattainable for many people. Thus, evolution expands the class of diseases that are medically manageable but economically incurable.

Table 2 Examples of the costs of human-induced evolution in insect pests and several disease organisms in the United States

Costs of Human-Induced Evolution in some Insect Pests and Diseases	
Factor	$U.S. billions per year
Additional pesticides	1.2
Loss of crops	2 to 7
S. aureus	
Penicillin-resistant	2 to 7
Methicillin-resistant	8
Community-acquired resistant	14 to 21
HIV drug resistance	6.3
Total for these factors	33 to 50

Ways of Slowing Evolution

Responding to the pervasive reach of evolution in medicine and agriculture, health specialists and agricultural engineers have developed an impressive series of innovative methods to slow the pace of evolution. A large body of theory guides deployment of some of these attempts (54–59). Other methods, circulated as guidelines for clinical practices or farming strategies, often appear to be developed through a combination of trial and error and common sense. Independent of their theoretical underpinnings, the following examples show that successful methods often slow evolution for clear evolutionary reasons and that these approaches may be generalizable to other systems.

Drug overkill and HIV triple-drug therapy. Overkill strategies, the combination of treatments to kill all infectious or invading pests, are common. For example, treatment with a drug cocktail that includes a protease inhibitor and two different reverse transcriptase inhibitors is the Cadillac of AIDS treatment strategies [51]. This approach has been successful longer than any other, because it not only reduces viral levels but also slows the evolution of resistance. The evolutionary biology hidden in this strategy is simple: a strong, multiple-drug dose leaves no virus able to reproduce, and so there is no genetically based variation in fitness among the infecting viruses in this overwhelming drug environment. Without fitness variation, there is no evolutionary fuel, and evolution halts. Lack of HIV variation for growth in this regime is responsible for reduced evolutionary rate and probably drives the current success of triple-drug treatment. However, sequential treatment with single drugs or voluntary drug cessation can foster the evolution of drug resistance [33], which appears to be increasing [60, 61]. This suggests that the triple-drug overkill strategy will not halt HIV evolution forever but may provide only a brief window for the development of more permanent solutions, such as HIV vaccines.

Overkill strategies have been echoed in pesticide management programs, where they are often termed “pyramiding” [62], and in treatment of bacterial infections [11]. However, their use is limited by drug toxicity: extreme doses can have physiological or ecosystem side effects.

Direct observation therapy. Tuberculosis infects one-third of the world’s population [10, 34], and is difficult to treat because it requires 6 months of medication to cure. Partial treatment has resulted in evolution of multidrug resistance [34]. To combat this, drug doses are brought individually to patients, who are observed while they take the drugs. This direct-observation therapy has been used to improve patient compliance during the whole treatment regimen, reducing evolution of resistance by ensuring a drug dose long enough and severe enough to completely eradicate the infection from each person. Direct-observation therapy has been credited with snuffing out emerging tuberculosis epidemics and dramatically reducing costs of medical treatment [10].

Withholding the most powerful drugs. The antibiotic vancomycin has been called the “drug of last resort,” because it is used only when other, less powerful antibiotics fail [10]. Withholding the most powerful drugs lengthens their effective life-span [11], because overall selection pressure exerted by the drug is reduced, slowing the pace of evolution. Although successful in reducing the evolution of resistance to vancomycin by some bacteria, the strategy depends on low use rates in all sectors of the antibiotic industry, including livestock and prophylactic use [13]. Failure to include these sectors in the strategy will engineer its failure.

Screening for resistance before treatment. Screening infections for sensitivity to particular antibiotics before treatment allows a narrow-range antibiotic to be used instead of a broad-spectrum antibiotic. Reduced use of broad-spectrum antibiotics slows evolution of resistance as in the mechanism above. Genotyping of viruses in an HIV infection and prediction of the antiviral drugs to which they are already resistant improves drug usefulness [63]. Similarly, farmers are advised to check their fields after pesticide treatment and then to change the chemical used in the next spraying if many resistant individuals are discovered. Screening for pest susceptibility reduces use of chemicals for which resistance has begun to evolve.

Cyclic selection due to changing chemical regimes. Farmers are encouraged to follow several simple rules to reduce herbicide resistance: (i) do not use the same herbicide 2 years in a row on the same field, and (ii) when switching herbicides, use a new one that has a different mechanism of action [64]. These guidelines slow evolution through a rapid alteration of selection pressure that sequentially changes the selective landscape. Mutants favored in one generation are not favored in the next, because one mutation is not likely to provide resistance to two herbicides with different mechanisms. Similar cyclic selection regimes have been proposed to limit resistance in intensive-care units [11, 59] and agricultural fields [62]. Mosaic selection, in which different chemicals are used in different places at the same time [65] is a spatial version of this tactic.

Integrated pest management. Integrated pest management (IPM) may include chemical control of pests, but does not rely on it exclusively, and is credited with better pest control and with slower evolution of resistance [62]. Slow evolution can come from two sources. First, the multiple control measures used in IPM reduce reliance on chemical treatments, thereby reducing selection for chemical resistance. Second, physical control of populations (e.g. through baiting, trapping, washing, or weeding) reduces the size of the population that is exposed to chemical control. Smaller populations have a reduced chance of harboring a mutation, thereby slowing the evolution of resistance. The term IPM is common only in insect management, but the strategy has appeared independently in hospitals where hand-washing, instead of prophylactic antibiotic use, is encouraged and in weed management, where resistant weeds are pulled by hand.

Refuge planting. Biotechnology has introduced insecticidal toxin genes into numerous crop species, but resistance to toxins produced by these genes has already evolved in pests, threatening the commercial use of this technology (66–68). To reduce the potential for evolution, crop engineers have instituted a program of refuge planting to slow the success of resistant insects [69]. If farmers plant a fraction of a field with non-toxin-producing crop varieties, and allow these to be consumed by insects, a large number of nonresistant pests are produced. These can then mate with the smaller number of resistant individuals emerging from fields of plants producing insecticidal proteins, greatly reducing the number of offspring homozygous for the resistance alleles. In cases where resistance is recessive, refuges slow the spread of resistant alleles [69], although they require high crop losses in the refuge plantings. This mechanism functions by reducing the inheritance of resistance through increases in the proportion of breeding individuals without resistance alleles.

Engineering evolution. Using evolution to our advantage may also be possible, although this is seldom attempted [p. 215 in [70]]. One illustrative exception is the use of the drug 3TC to slow the mutation rate of HIV and thereby, perhaps, to limit its ability to rapidly evolve resistance to other drugs [24]. An ongoing use of evolutionary theory is the prediction of which influenza strains to use for future vaccines [15]. Another is the use of chemical control where resistance includes a severe metabolic cost, making resistant individuals less fit when the chemicals are removed [71]. In such cases, the potential of evolution to lower pest fitness in the absence of a pesticide may be a method of using the power of evolution to our advantage. An unintended evolutionary outcome may be the escape of antibiotic, herbicide, or pesticide resistance genes to natural populations, possibly making them less susceptible to pesticides in the environment. In some agricultural settings, artificial selection for pesticide resistance has been used to protect populations of beneficial insects [72].

This summary shows that successful control of evolution has followed many different strategies, and that the methods currently used impact all three factors driving evolutionary change (Table 3). However, seldom have all three evolutionary prerequisites been manipulated in the same system, and seldom has the engineering of the evolutionary process been attempted in a systematic way. Instead, in every new case, human-mediated evolution tends to catch us by surprise, and strategies to reduce or stop it are invented from scratch. For example, cyclic selection has been invented at least three times (for control of insects, bacteria, and HIV), IPM at least three times (insects, weeds, and bacteria), and drug overkill at least twice (HIV and tuberculosis).

Overall, three ways to adjust selective pressures are widely used in pest and health management: application of multiple simultaneous chemicals or "pyramiding," cyclic application of different chemicals, and using different chemicals in different places or "mosaic application." Although the principles are exactly the same in all fields, seldom has the literature from one field been used to inform the other [73]. Some strategies that are very successful in one arena have not been tried in others (e.g., no direct-observation therapy has been tried on farms). Yet, the commonality of successful methods (Table 3) suggests that lessons in evolutionary engineering from one system may be useful in others and that it may be possible to control evolution far more successfully than is currently practiced. Mathematical models of evolutionary engineering provide some guidance about practical field methods [54, 62], but this exchange between prediction and practice has only been common in pest management [65] and antibiotic resistance [59]. A critical need is the inclusion of evolutionary predictions in the current debate on global HIV policy. Most important, it is seldom realized that a pivotal goal is slowing the evolution of resistance and that, without this, all successful pest and disease control strategies are temporary [62, 70, 74].

Table 3 The success of evolutionary engineering: mechanisms that reduce evolution can and do work on all three parts of the evolutionary engine

Mechanisms That Work to Slow Evolution	
Method of slowing evolution	Example
Reduce variation in a fitness-related trait	
Drug overkill with multiple drugs	Triple-drug therapy for AIDS
	Pesticide pyramiding
Ensure full dosage	Direct observation therapy of tuberculosis
Reduce appearance of resistance mutations	Engineer RT gene of HIV-1
Reduce pest population size	Integrated pest management of resistant mutants
	Nondrug sanitary practices
Reduce directional selection	
Vary selection over time	Herbicide rotation
	Vary choice of antibiotics, pesticides or antiretrovirals
Use nonchemical means of control	Integrated pest management
Limit exposure of pests to selection	Withhold powerful drugs, e.g., restricted vancomycin use
Avoid broad-spectrum antibiotics	Test for drug or pesticide susceptibility before treatment of infections or fields
Reduce heritability of a fitness-related trait	
Dilute resistance alleles	Refuge planting

Conclusions and Prospects

Rapid evolution occurs so commonly that it is, in fact, the expected outcome for many species living in human-dominated systems [62]. Evolution in the wake of human ecological change should be the default prediction and should be part of every analysis of the impact of new drugs, health policies, pesticides, or biotechnology products. By admitting the speed and pervasiveness of evolution, predicting evolutionary trajectories where possible, and planning mechanisms in advance to slow evolutionary change, we can greatly reduce our evolutionary impact on species around us and ameliorate the economic and social costs of evolution [70]. Ignoring the speed of evolution requires us to play an expensive catch-up game when chemical control agents and medications fail. Because our impact on the biosphere is not likely to decline, we must use our knowledge about the process of evolution to mitigate the evolutionary changes we impose on species around us.

Note added in proof: In two recent papers [76, 77], the genetic basis of resistance to BT toxins has been discovered in nematodes and lepidopterans. In both cases, mutations at single genes appear to confer substantial resistance, and might also provide cross resistance to different BT toxins. Without efforts to mediate this evolutionary potential, strong selection in diverse plant pests at a single locus may generate field resistance to transgenic Bt-producing crops or to commercially used sprays of Bt toxin.

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Urbanization

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Keywords: urbanization · world cities · population growth · pollution · climate · stream flow · industrial revolution

How has the concentration of the world's population in urban settlements changed in the past 300 years? Where and when has urban growth occurred, and why? What has happened to the distribution of cities by size? The first purpose of this chapter is to answer these questions by laying out the evidence on urban growth since 1700.

Has this urbanization resulted in environmental change? Is further urbanization likely to do so in the years ahead? Neither the social nor the physical sciences have answered these questions, yet the broad outlines of an answer certainly can be sketched. Until the middle of the twentieth century, urbanization levels were too low and the number of large cities was too small for there to be anything other than local climatic and hydrologic impacts. To the extent that urbanization produced environmental modification, it was in urban-centered gradients of agricultural land use and mineral and forest exploitation as urban demands diffused into the surrounding countryside. As late as 1900, there were barely 43 cities in the world exceeding 500,000 population, of which only 16 exceeded 1,000,000.[1] But since 1950, the number of large cities has increased very rapidly – close to 400 now exceed 1,000,000. Sprawling metropolitan areas have formed even larger agglomerations, and some very large urban regions with populations in the tens of millions have emerged. The question that arises in these cases is whether or not changes in the biosphere are unfolding at a regional scale that, in turn, might have global impacts. There is little to suggest that historic urban developments were active agents in climatic change. There is significant evidence that the modern metropolis has climatic and hydrologic consequences that increase with city size and urban densities. There is at least the suggestion that these consequences may be compounded at a regional scale in the largest agglomerations. But if our analysis is correct, regional-scale impacts may be more likely in the years ahead in Third World nations, where very large urban agglomerations are emerging, rather than in the most economically advanced countries, where a transformation is unfolding that is resulting in dispersed and relatively low-density urban networks. The very regions in which environmental alterations are most likely are those regions in which increasing shares of the world's population are concentrating.

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Urbanization in 1700: City-Centering of World Economies

The world of 1700 was largely agrarian. Urban populations were less than 10% of the whole. Yet in Fernand Braudel's view, this world was divided into a number of *city-centered world economies*: "economically autonomous sections of the planet able to provide for most of their own needs, sections to which their internal links and exchanges gave a certain organic unity" (Braudel 1984: 22). World economies, he said, centered on *world cities* that were in perpetual political and economic rivalry with each other, some rising and some falling. Each world city was surrounded by an immediate *core region* within which modification of the earth was greatest, a fairly developed *middle zone*, and a vast and relatively untouched *periphery* (Braudel 1984: 39). The core contained the concentration of everything that was most advanced and diversified, lying at the heart of the middle zone, the settled area of the state.

Thus, in the seventeenth century. Amsterdam was the "warehouse of the world" and the United Provinces were the middle zone. In this zone, urbanization levels rose to more than 30% (Wrigley 1987: 183): a high degree of agricultural specialization in cash crops for both the urban consumer and the industrial market developed; agriculture became close to gardening; ingenious methods of crop rotation were developed that were also to transform the English agricultural landscape; new drainage technologies were developed that enabled Holland's cultivable area to be increased and the British fenland to be settled; and the new middle-class spirit of Protestantism linked to capitalism was fostered, carrying with it associated ideas of man's dominion over nature. The closer to Amsterdam, the greater the degree of cash-crop specialization and the greater the extent of environmental modification to support agricultural development. The pattern was universal; the further from the world city, the less the clearance of the woodlands for ships' timbers, fuelwood, and farming. The more the city grew, the more intense the modification of the core and the wider the ring of diffusion into the middle zone. Furthest from the world city was the periphery, "with its scattered population, representing on the contrary backwardness, archaism, and exploitation by others" (Braudel 1984: 39; also Wallerstein 1974).

Tertius Chandler's statistics provide graphic evidence of the city-centeredness of Europe's world economies of 1700 (Chandler 1987; see also Table 1). What is impressive is both the sharpness of the primacy of most of Europe's world cities, many times the sizes of the other towns within their world economies, and the smallness of the capitals themselves, even though they accounted for a large share of the total urban population. Europe's six significant world economies centered on physically compact world cities that ranged in population from 200,000 to 700,000. Their surrounding core regions were equally small, as were the zones of active environmental modification.

This pattern was repeated elsewhere in the world. In China, new Manchu rulers had by 1700 restored the state apparatus of the Ming Dynasty. The Ching state (1644–1911), managed by competitively selected literati, engaged in economic planning to assure adequate supplies and effective distribution of foodgrains, presiding over a flexible market structure linking urban areas to the rural economy. Water management for both irrigated agriculture and transportation was one of the central administration's main duties, and was the measure of the efficiency of the state (Wittfogel 1957). The key component of the Chinese urban system was the establishment of a capital city – the emperor's seat and supreme political and spiritual authority of the empire – dominating and controlling the entire kingdom and concentrating the power of the bureaucracy. Beneath the capital city was an echelon of military-administrative centers, and beneath them the *Hsien* (county) capitals, which fulfilled the administrative roles of tax collection, military garrison, and dispensing public functions (Eisenstadt and Shachar 1987: 130).[2]

Similarly, in India, the Mogul Empire had been firmly established under the rule of Akhbar (1556–1605), who instituted a well organized central administration that was the cornerstone of governance over the next centuries and the basis of primate-city dominance. A similar dominance was evidenced in Japan even before reunified national political authority was asserted during the

Table 1 Urban Centers of the World Economies in 1700

France		**Britain-Holland**	
Paris	550,000	London	550,000
Lyon	97,000	Dublin	80,000
Marseille	75,000	Edinburgh	35,000
Rouen	63,000	Norwich	29,000
		Bristol	25,000
		Amsterdam	210,000
		Leiden	56,000
		Rotterdam	55,000
		Haarlem	48,000
		The Hague	36,000
Ottoman Empire		**Spanish Empire**	
Constantinople	700,000	Naples	207,000
Cairo	175,000	Palermo	124,000
Smyrna	135,000	Milan	113,000
Adrianople	85,000	Madrid	105,000
Damascus	70,000	Seville	80,000
Aleppo	67,000	Brussels	70,000
Bursa	60,000	Antwerp	67,000
Mecca	50,000	Mexico City	85,000
Baghdad	50,000	Potosi	82,000
Bucharest	50,000	Puebla	63,000
Belgrade	40,000		
Salonika	40,000		
German States		**Portugal**	
Hamburg	63,000	Lisbon	188,000
		Oporto	23,000
Austria		**Russia**	
Vienna	105,000	Moscow	114,000
Prague	48,000		
China		**Japan**	
Peking	650,000	Yedo	688,000
Hangchow	303,000	Osaka	380,000
Canton	200,000	Kyoto	350,000
Sian	167,000	Kanazawa	67,000
Soochow	140,000	Five more over	50,000
Nanking	140,000		
Wuchang	110,000	**Korea**	
Kingtehchen	100,000	Seoul	158,000
Niaghsia	90,000	Pyongyang	55,000
12 more over	50,000		
Moghul Empire		**Persia**	
Ahmedabad	380,000	Isfahan	350,000
Aurangabad	200,000	Tabriz	75,000
Dacca	150,000	Qazvin	60,000
Srinagar	125,000		
Patna	100,000	**Siam**	
Benares	75,000	Ayutia	150,000
Agra	70,000		
Delhi	60,000		

Source of data: Chandler 1987.

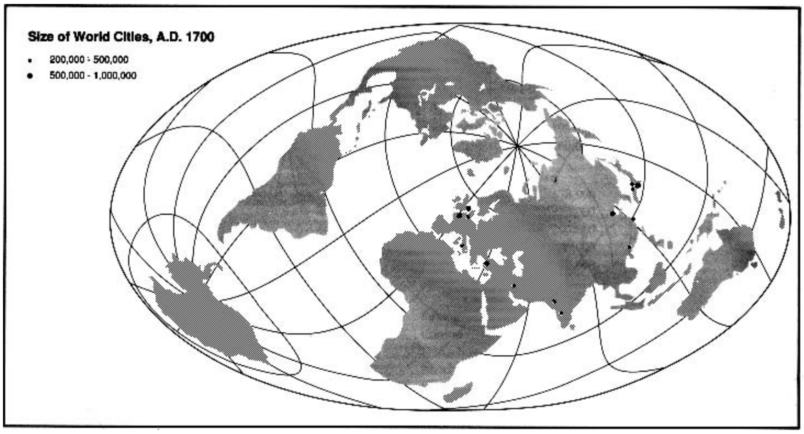


Fig. 1 The world cities in A.D. 1700

Tokugawa Shogunate (17th to 19th centuries). Japan's major cities were not simply political-administrative centers, but also centers of trade, tightly controlled by wholesale and retail monopolies and by the guilds. As in Europe, Asian world cities and the *ecumene* were small. The world urban map of 1700 was nearly empty. Only 5 cities exceeded 500,000 population, and only 34 exceeded 100,000.

Beyond Europe and Asia, in Central and South America, the largest urban places were those of the Spanish and Portuguese empires. Outside the Ottoman Empire, the largest of Africa's cities were Muslim – Algiers, Fez, Meknes, Tunis, and Sale-Rabat exceeded 50,000 in the Mahgreb, whereas the principal south Moslem centers were Katsina, Kazarganu, and Zaria. Only Oyo reached 50,000 in Black Africa. A few small dots on the map contained the majority of the world's urban population (Fig. 1).

Eighteenth-Century Quickening: Britain Emerges

The first example of urban-led economic growth that brought urbanization levels above 10% was that of the Netherlands from the early sixteenth century through their great age of economic supremacy in the seventeenth century (Braudel 1984; DeVries 1981; also see Fig. 2). The next example was that of Britain, whose navies and trading companies established their ascendancy during the eighteenth century as European nation-states reached outward to expand their world economies, establishing

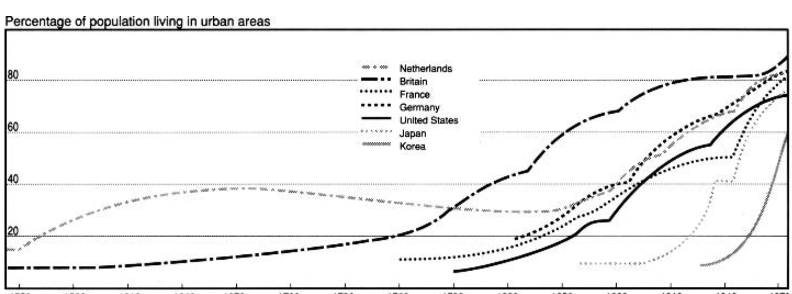


Fig. 2 Changing levels of urbanization, 1550–1980: The Netherlands, Britain, the United States, France, Germany, Japan, and Korea

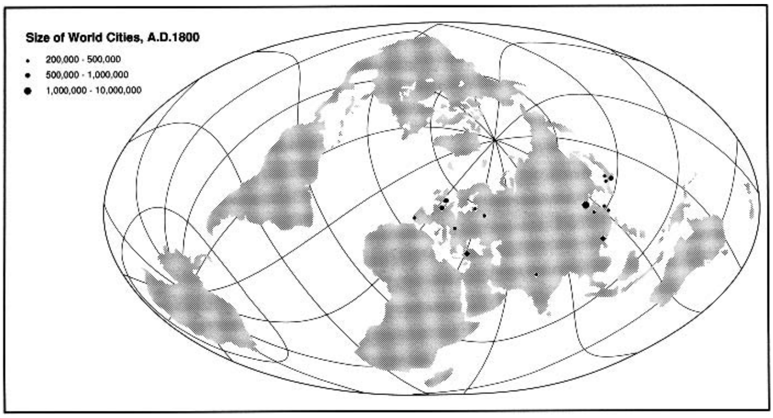


Fig. 3 The world cities in A.D. 1800

colonial outposts to exploit the resources of the peripheries and setting in motion the expansion of pioneer settlement frontiers. Radiating outward, there were waves of clearance, drainage, conversion, and extraction; with the extent of environmental modification – of human dominion over nature – patterned by gradients of accessibility to the world cities. London grew to be the largest city in Europe. Its food market radically changed the agricultures of the Kentish and East Anglian core. Its wealthy merchants bought country estates and hired the landscape gardeners who created England's rural landscape. The wealthy merchant classes of the core became the principal dissenters who set in motion Europeans' drive to master the North American wilderness. Yet by 1800, the world's urban map had scarcely changed (Fig. 3). Chandler's statistics show that the number of cities in the world with greater than 500,000 population had increased only from 5 to 6 and that the number of additional places exceeding 100,000 had increased only from 29 to 44 (Tables 2 and 3).

In contrast to the stationary state of Holland in the eighteenth century, it was Britain's growth that had been quickening. As a result, the urbanization of the population had been increasing since the mid-seventeenth century. But the pattern still was one of world-city concentration as mercantile expansion took place. Britain's urban percentage rose from 8.25% in 1600 to 17.0% in 1700, and London's share of the national population increased from 5% to 11.5% (Wrigley 1987: 163), justifying James I's fear that "soon London will be all England." The concentration of Britain's urban population in London increased from 60% in 1600 to 67% in 1700. The city's net population increase of 275,000 from 1650 to 1750, at a time when its death rates exceeded its birth rates, was achieved by absorbing the natural increase of a population of 5 million people, in which the surplus

Table 2 Changes in City-Size Distributions, 1700–1975

	Numbers of Cities Exceeding Specified Sizes (population in thousands)					
	>100	>200	>500	>1,000	>10,000	>20,000
1700[a]	34	14	5			
1800[a]	50	17	6	1		
1900[a]	287	142	43	16	1	
1950[b]	950	N.E.	179	76	N.E.	
1975[a]	N.E.	N.E.	422	191	7	1
2000[b]	1,699	N.E.	859	440	N.E.	N.E.

[a] Adapted from Chandler 1987: 521.
[b] United Nations 1980.
N.E.: not estimated.

Table 3 Major Cities in the World Economies of 1800

Britain[a]		**British Empire**		**Portugal**		**Austria-Hungary**	
London	861,000	Lucknow	240,000	Lisbon	237,000	Vienna	231,000
	(959,000)	Murshidabad	190,000	Oporto	67,000	Venice	146,000
Dublin	165,000	Benares	179,000			Prague	77,000
Glasgow	84,000	Hyderabad	175,000	**Holland**			
Edinburgh	82,000	Patna	170,000	Amsterdam	195,000	**Russia**	
Manchester	81,000	Calcutta	162,000	Rotterdam	60,000	Moscow	248,000
	(89,000)	Bombay	140,000			St. Petersburg	220,000
Liverpool	76,000	Surat	120,000	**Ottoman Empire**			
	(83,000)	Madras	110,000	Constantinople	570,000	**Japan**	
Birmingham	71,000	Dacca	106,000	Smyrna	125,000	Yedo	685,000
	(74,000)			Damascus	90,000	Osaka	383,000
				Adrianople	80,000	Kyoto	377,000
France				Aleppo	72,000	Nagoya	92,000
Paris	547,000	**French Empire**				Kanazawa	71,000
Lyon	111,000	Cairo	186,000	**Chinese Empire**			
Bordeaux	92,000			Peking	1,100,000	**Marathas and Rajputs**	
Marseille	83,000			Canton	800,000	Delhi	140,000
Rouen	80,000			Hangchow	387,000	Ujjain	100,000
Nantes	70,000			Soochow	243,000	Ahmedabad	89,000
				Sian	224,000	Baroda	83,000
Spain				Kingtehchen	164,000	Jodhpur	75,000
Madrid	182,000	**Spanish Empire**		Wuchang	160,000	Bharatpur	75,000
Barcelona	113,000	Mexico City	128,000	Tientsin	130,000	Nagpur	74,000
Seville	96,000	Manila	77,000	Foshan	124,000		
Cadiz	87,000			Chengdu	97,000	**Burma**	
Italian States				Langchow	90,000	Amarapura	175,000
Naples	430,000	**German States**		Changsha	85,000		
Rome	142,000	Berlin	172,000	Ningpo	80,000	**Korea**	
Palermo	135,000	Hamburg	117,000	Kaifeng	80,000	Seoul	194,000
Milan	122,000	Warsaw	75,000	Hsuchow	75,000	Pyongyang	68,000

Source: Chandler 1987.
[a] The alternative populations placed in parentheses are those appearing in Wrigley 1987: 160.

of births over deaths was 5 per 1,000 per annum (Wrigley 1987: 135–36). Capital-city concentration was a feature not only of English urban growth: during the eighteenth century fully 80% of European urban growth took place in its capital cities (DeVries 1981: 88).

It was Britain that first broke with the pattern of world-city urban concentration. From 1700 to 1800, the degree of urbanization in England increased from 17.0% to 27.5%, but London's share of the national population remained constant at around 11.0%, while its share of the urban population dropped from 67% to 40%. Urban growth accelerated outside London, with the main burst of expansion occurring in the last quarter of the century in such cities as Manchester, Liverpool, Birmingham, and Glasgow, and a second echelon in the 20,000-to-50,000 range that included Leeds, Sheffield, Newcastle, Stoke, and Wolverhampton. The English share of European urban growth had been 33% in the seventeenth century, but was over 70% in the second half of the eighteenth century (Wrigley 1987: 177), and this increased share was concentrated outside London in the newly industrializing north.

This was, of course, the initial wave of the Industrial Revolution, brought about by major advances in the cotton and iron industries, the first flush of factory building, significant improvements in waterborne transportation, and also colonial policies that systematically destroyed Indian cotton-textile production and guaranteed imperial markets to Lancashire's producers. As a result of the quickening of growth, there were already heavy pressures on land use for bread grains and pasturage early in the century, to which one response was the Enclosure Movement. There was need for timber for shipbuilding, oaks for the Royal Navy, and wood ash for the alkalies used in the bleaching process by the textile industry. But above all, an energy shortage afflicted the economy at midcentury, calling forth the key inventions of Cort and Watt: the development of a substitute (coal) for progressively scarcer wood (the production of 10,000 tons of charcoal-iron required the felling of 40,000 hectares of forest; Wrigley 1987: 79); the need to drain the coal mines (steam engine); and the need to transport the coal (canals). Coal output increased in Britain from 3 to 10 million tons in the eighteenth century, particularly in the northeast, with easy access by sea to the London market.

The origins of the factory system were in another crisis: the shortage of spinners to supply the hand-loom weavers. The water frame and spinning jenny came into use in the 1770s. Water-powered scribbling mills were introduced in the 1780s, taking over the tasks of teazing and carding, but they were as dispersed as the weaver-crofters. It was only after 1800 that factory production concentrated on the coal fields, rivers, and canals; when steam began to replace the power of the overshot water wheel; and when the scribbling mill, the power mule, the dyehouse, the fulling mill, the warehouse, and the cropping shop were incorporated under a single roof. Only then did industrial urbanization begin in earnest, coal-field-oriented, with housing developments confined to walking distance of the mills, and it was reinforced after the turn of the century by the railroad and the steamship.

Long Waves of Industrial Urbanization

The late-eighteenth-century burst of industrial growth was concentrated in Britain, and ended in the sharp depression that followed the Napoleonic Wars and the War of 1812. From initial acceleration to a peak in 1792, followed by a turnaround into deceleration to depression, the wave of growth lasted about 55 years. This 55-year pattern has been repeated three more times in modern history. Each growth upswing quickened urbanward migration; each slowdown was followed by a lower rate of urban growth. From the 1820s on, the rhythms were sharpest in the United States, where the urbanward migrants came not only from America's farms, but from Europe too. The waves of emigration to the New World that increased in good times and decreased in bad times flattened the 55-year rhythms of European urban growth.

Urban growth, of course, has two components: natural increase and net migration. It is in urbanward migration that the 55-year long-wave rhythmicity of urban growth is revealed. If this rhythmicity is compared with the long swings of economic growth (as measured by the average annual growth rates of real GNP and of real GNP per capita), it is clear that the urbanward migrants were simply responding to economic opportunity; the waves of urban growth in the United States were sympathetic and somewhat lagged responses to waves of economic growth (Fig. 4). Each burst of economic growth called forth a rush of urbanward migrants and raised the level of urbanization.

It was the Soviet economist Nikolai Kondratiev who first drew attention to the 55-year-long-wave phenomenon (Kondratiev 1935; like all such ideas, this one had antecedents, for example van Gelderen 1913). As enriched by growth theorists such as Schumpeter (1934), the theory of long waves centers on the role of key innovations that become the leading sectors in growth. In the eyes of these theorists, clusters of innovations produce accelerated expansion until markets are saturated; there is then a recessionary turning point, acceleration turns to deceleration, and deceleration turns to stagnation and collapse in a depression. Venture capitalists then look for new sources of profit,

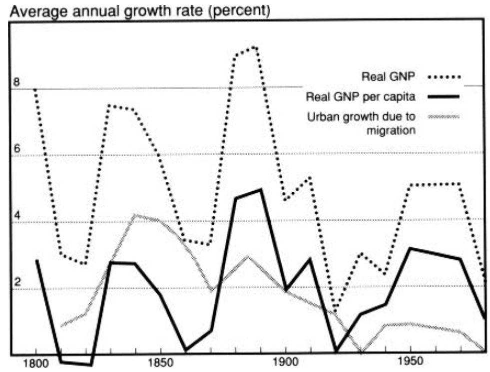


Fig. 4 Long waves of economic growth and urbanward migration in the United States, 1790–1980

investing in new technologies that become the leaders for the next wave of growth. The span from depression to depression averages some 55 years.

Following this argument, the late-eighteenth-century growth that was Britain's alone (the "first Kondratiev") was sparked by innovations in the water-powered textile industries. Britain's second wave of growth (the "second Kondratiev") was coal-based and steam-powered, marked by mechanization of the textile factories, initial railroadization, and the growth of the iron industry. It was in this wave that Britain became the "workshop of the world" and the center of the Atlantic economy. Around 1830, jerry-builders hit upon an ingenious house design that could save both land and building materials and that produced increasing urban concentration – the back-to-back house. Built in double rows under a single roof, with a standpipe for water supply at the end of the streets and a couple of earth closets devoid of privacy for each 150 people, this design could cram large numbers of houses into small spaces, without either sunlight or ventilation. The result was "...the despair of medical officers ...(with) a mortality rate greater by 15 or 20 percent ...the absence of a general system of sewerage, the imperfect conditions of the streets and roads, the confined courts, the open middens and cesspools, stagnant ditches and insufficient water supply" (F. Tillyard, quoted in Aldridge 1915).

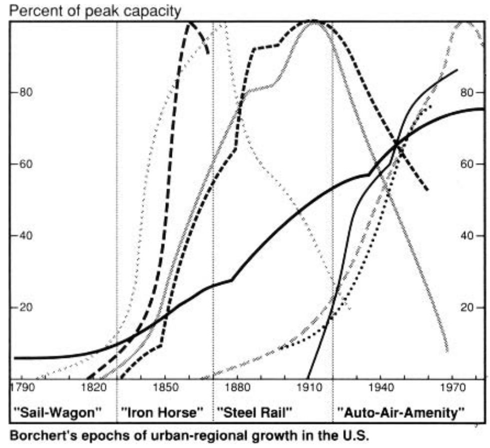
It was apparent that the new urbanization was creating unhealthy environments – environments that persisted long into the twentieth century and were responsible for the growth of the modern public health and urban planning professions. Even in 1913 in Birmingham

> 200,000 people were housed in 43,366 buildings of the back-to-back type already long condemned or injurious to health because of lack of ventilation. In the worst six wards, from 51 percent to 76 percent were back-to-backs. Even more serious was the fact that 42,020 houses had no separate water supply, no sinks, no drains, and 58,028 no separate w.c., the closets being communal and exposed in the courts. This meant that over a quarter of a million people lived in cavernous conditions. The real objection to back-to-back houses lies not so much in their method of construction as in the degrading and disgusting condition of their out-buildings, which frequently made decency impossible and inevitably tended to undermine the health and morals of the tenants (Bourneville Village Trust 1941: 16).

There was lagged emulation of both the best and the worst of Britain's second-Kondratiev growth, in which factory chimneys and smoky atmospheres were the mark of progress. Victorian England became the hub of a worldwide economy. Liverpool and Manchester became the first noncapital cities in the world ever to rival in size the capital cities of the world empires. Railroadization and both industrial and urban growth diffused outward, finally reaching the world's furthest peripheries late in the twentieth century (Berry et al. 1987: 415–16).

As the Victorian boom began in Britain, the first wave of United States industrial and urban growth also took place, but it was predominantly water-powered (Fig. 5; Borchert 1967); this was the epoch in which the northeastern mill towns grew. Only toward the end did the railroads and the iron industries expand. It was not until after the Civil War that the locus of initiative shifted to

Fig. 5 Changing levels of urbanization in the United States compared with the dominant mode of transportation, 1790–1980



the United States and to continental Europe as the leading growth regions (Fig. 6). Urban growth rates in Britain slowed down, and were further depressed by the magnitude of emigration. It was the third Kondratiev that was the principal epoch of coal-based steam power in the United States (later supplemented by gas and electricity), of steel rails and ships, and of the growth of the chemical industry. New steel-frame technologies enabled urban densities to be increased by going upward at the same time that the balloon frame enabled rapid construction of workers' housing further outward. New communications technologies enabled the head office to be separated from the factory floor, and the office skyscraper core of the modern central business district was born. Large-scale mass production gave rise to the essence of urbanization, "a process of population concentration" (Tisdale 1942), characterized by increasing size of cities, increasing population densities, and the increasing heterogeneity of their immigrant populations (Wirth 1938). Radiating rail lines and omnibuses, originally horse-drawn but soon steam- (and later electric-) powered, first enabled owners and managers (and later the workers) to commute to residences beyond the confusion and smoky atmospheres of the concentrated core. By the end of the nineteenth century, the modern metropolis as we know it had been born.

When, in 1899, he wrote his monumental study *The Growth of Cities in the Nineteenth Century*. Adna Ferrin Weber concluded that "the most remarkable social phenomenon of the present century is the concentration of population in cities" (Weber 1899: 1). Chandler's statistics reveal a fivefold increase in the cities exceeding 125,000 population and an eightfold increase of those exceeding 500,000 between 1800 and 1900 (Table 2). The ancient Asian urban hierarchies of 1800 were replaced by a map dominated by major cities within Northern-hemisphere industrial regions, linked by overseas gateways to colonial empires (Fig. 7).

Britain's urban map was dominated in 1900 by nine major cities and a constellation of smaller industrial towns:

London	6,480,000
Manchester	1,435,000
Birmingham	1,248,000
Glasgow	1,015,000
Liverpool	940,000
Newcastle	615,000
Leeds	430,000
Sheffield	402,000
Edinburgh	400,000

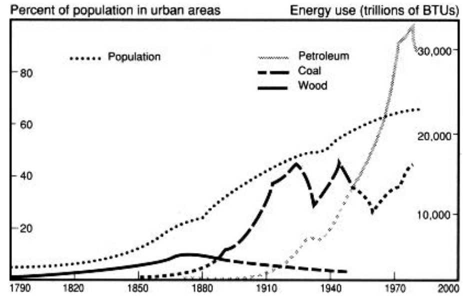


Fig. 6 Dominant sources of energy compared with the level of urbanization, 1790–1980

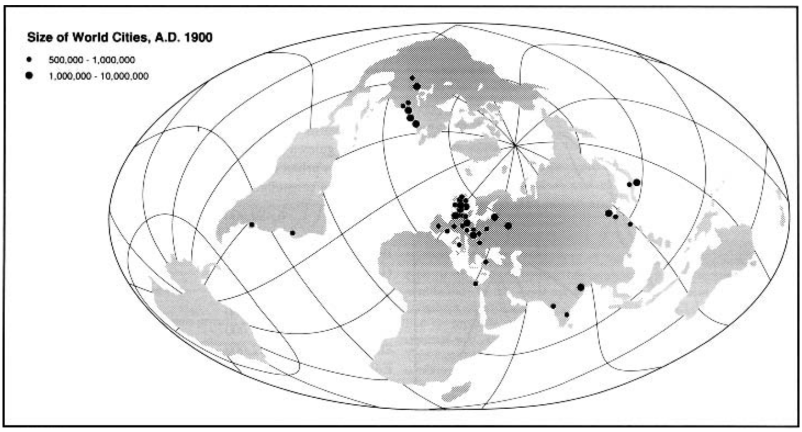


Fig. 7 The world cities in A. D. 1900

Eight urban complexes marked out Germany's industrial heartland:

Berlin	2,707,000
Hamburg	895,000
Ruhrgebiet	766,000
Dresden	540,000
Leipzig	532,000
Munich	499,000
Cologne	437,000
Breslau	427,000

Paris (3,330,000) still dominated France, but Lyon (508,000) and Marseille (410,000) had grown. Barcelona (552,000) now rivaled Madrid (539,000) in Spain, and Italy's first wave of industrialization had seen the rise of Milan (491,000), alongside Naples (563,000) and Rome (438,000). And instead of small mercantile centers on the eastern seaboard of North America in 1800, there now was the northeastern manufacturing belt marked out by major urban centers:

New York	4,242,000
Chicago	1,717,000
Philadelphia	1,418,000
Boston	1,075,000
St. Louis	614,000
Pittsburgh	562,000
Baltimore	508,000
Cincinnati	417,000

Standing in a dependent relationship to this heartland, radiating out across the national landscape, were resource-dominant regional hinterlands. But whereas within the United States the resources of the hinterlands had been brought within the nation's boundaries during the course of the nineteenth century by purchase and conquest, many of Europe's resource hinterlands were parts of overseas empires, and major cities had grown at the points of colonial penetration. To be sure, San Francisco (439,000) was a gateway to the West, but in the zones of active European settlement in the Southern Hemisphere, comparable gateway cities included Buenos Aires (806,000), Rio de Janeiro (744,000), Melbourne (485,000), and Sydney (478,000). Cairo's growth (595,000) reflected the flow of world commerce through the Suez Canal, Calcutta (1,085,000), Bombay (780,000), and Madras (505,000) were the gateways through which British Imperial domination of India was exercised, and even though Peking (1,100,000) remained China's largest city, external influence through Tientsin (700,000), Shanghai (619,000), Canton (585,000), and Hankow (480,000) dictated the dissolution of that imperial system. The balance of the world's urban map in 1900 was the more familiar one of the centers of the world empires: Europe's other capitals had grown [St. Petersburg (1,439,000), Moscow (1,112,000), Constantinople (900,000), Warsaw (724,000), Brussels (561,000), Amsterdam (510,000), and Copenhagen (462,000)]; and Tokyo (1,497,000) and Osaka (970,000) were still dominant in Japan; but there was a great gulf between them and lesser cities within their regions.

As impressive as these numbers may be, the change had not yet produced any truly urbanized societies, save perhaps Great Britain. Only 43 cities in the world exceeded 500,000 population in 1900, although 16 of them were now more than a million. Barely ten nations had more than 25% of their populations living in urban centers of more than 10,000 people in 1900 (Great Britain, Belgium, the Netherlands, Germany, France, the United States; Turkey-in-Europe, plus Australia, Argentina, and Uruguay) – a level of urbanization that was surpassed in 1985 by all but a few of the very poorest of the world's nations (Figs. 8 and 9).

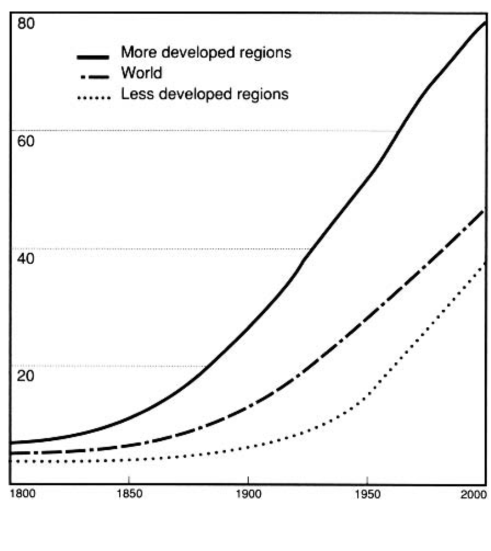


Fig. 8 Increases in the level of urbanization in the world's more- and less-developed regions. 1800–2000

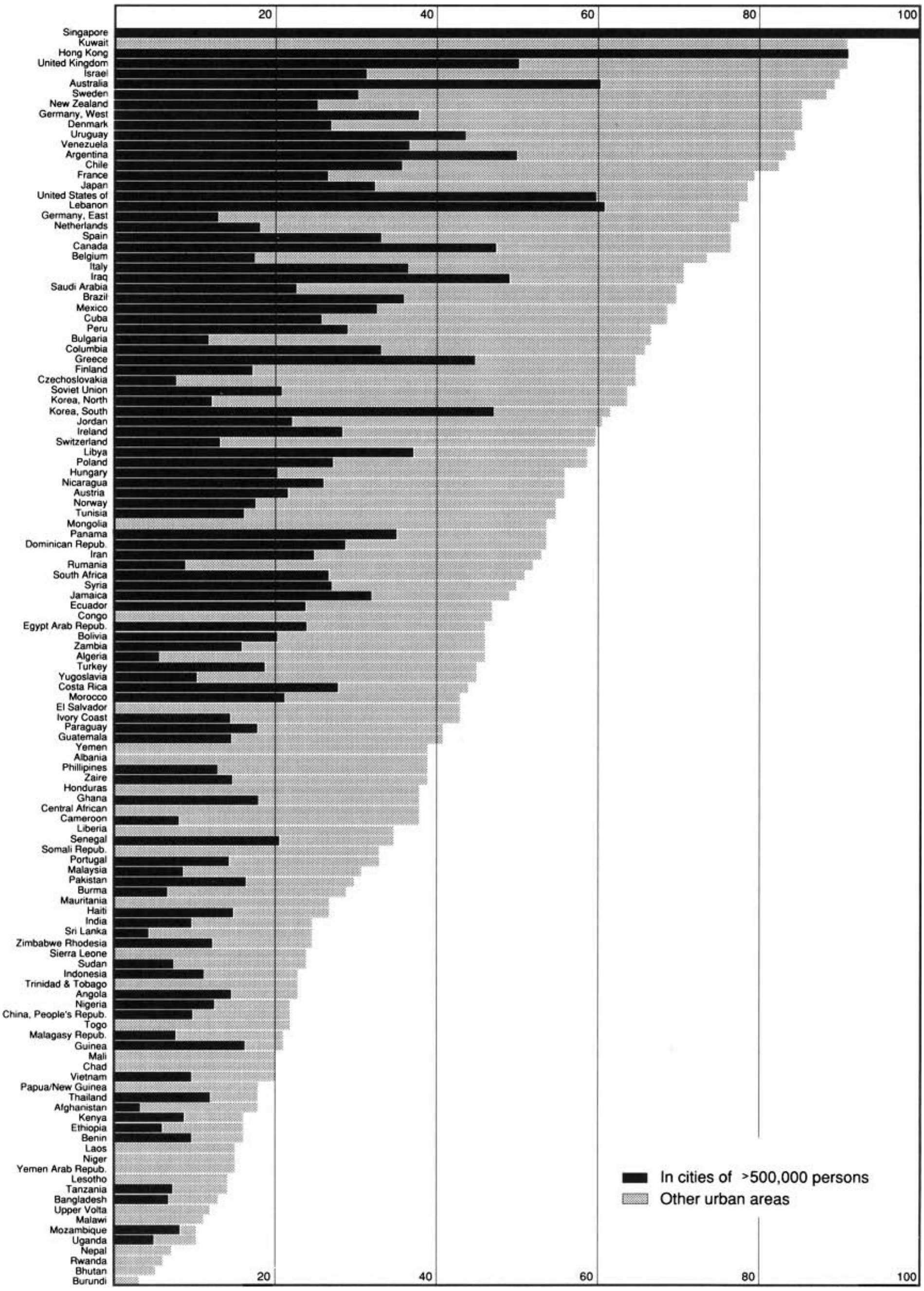


Fig. 9 World urbanization levels in 1985

What was significant about late-nineteenth-century growth was not that it urbanized the world, but that it produced a new scale and texture of world empires; the heartland (core) and hinterland (periphery) pattern of regional specialization in which multiechelon hierarchies of urban places played the critical structuring role. In the United States this involved the concentration of big cities in the northeastern manufacturing belt, the great heartland nucleation of industry and the national market, the focus of large-scale national-serving industry, the seedbed of new industry responding to the dynamic structure of national final demand, and the center of high levels of per capita income. Standing in a dependent relationship to the heartland, radiating out across the national landscape there developed resource-dependent regional hinterlands specializing in the production of raw material and intermediate outputs for which the heartland reached out to satisfy the input requirements of its great manufacturing plants. In the hinterlands, resource endowment was a critical determinant of the particular cumulative advantage of the region and hence its growth potential. In each case, the basic conditions of regional growth were set by the heartland. It was the lever for successive development of newer peripheral regions by reaching out to them as its input requirements expanded, and it thereby fostered specialization of regional roles in the national economy. The heartland experienced cumulative urban-industrial specialization, while each of the hinterlands found its comparative advantage based on narrow and intensive specialization in a few resource subsectors – diversifying only when the extent of specialization enabled the hinterland region to pass through that threshold scale of market necessary to support profitable local enterprise. Flows of raw materials inward and finished products outward articulated the whole. Large cities grew at the center of each region and at the top of each regional hierarchy – centers of activity and of innovation, focal points of the transport and communications networks, locations of superior accessibility, at which firms could most easily reap economies of scale and at which industrial complexes could obtain economies of localization and urbanization, encouraging labor specialization and efficiency in the provision of services.[4]

The urban-industrial system that evolved within the expanding national boundaries of the United States was duplicated globally by each of Europe's imperial powers. Each developed an economic heartland and each reached out globally for resource-dependent hinterlands, in the core-periphery pattern described by V. I. Lenin's "colonial model." Only imperial Russia mirrored the United States by bringing its peripheries within the frontiers of the nation-state. For Britain, France, Germany, Italy, Spain, and Portugal, the mother-country, urban-industrial cores reached out both for colonial raw materials and for safe colonial markets. Systems of imperial preference cemented the relationship, along with active European settlement of more temperate areas, from which the indigenous populations were relatively easily displaced. Urban-centered interconnections held the colonial networks together – networks that were to be disconnected in the twentieth century by two global wars separated by a profound depression. World War I disassembled the ancient Habsburg and Ottoman empires. World War II stemmed Germany's ultimately unsuccessful search for *lebensraum* and Japan's attempt to create its "greater East Asia co-prosperity sphere" by military means, but the price to the victors also was a loss of empire.

The Great Depression marked another technological watershed – from coal, steam, and rail to petroleum and the internal combustion engine. The base for new rounds of urban growth was the kind of city that had emerged in the nineteenth century, built on productive power, massed population, and industrial technology, and credited with the creation of a system of social life founded on entirely new principles. "Urbanization," wrote Tisdale (1942) "is a process of population concentration. It proceeds in two ways; the multiplication of the points of concentration and the increasing in size of individual concentrations. . . . Just as long as cities grow in size or multiply in number, urbanization is taking place. . . . Urbanization is a process of becoming. It implies a movement . . . from a state of less concentration to a state of more concentration." It was these concentrated urban environments that produced the local climatic and hydrologic alterations discussed in the section that follows.

In part, the reason for agglomeration was the concentration of large-scale production facilities at strategic points on efficient interregional transportation networks. It resulted partly from the specialization of functions that large-scale industry made possible, with external economies to be reaped within the agglomerations. But relatively poor intraregional transportation (still predominantly foot and horse until the Great Depression) meant that externalities could be captured only in the most central locations within the agglomerations. Supported by new building technologies, high-rise central business districts developed at the urban cores, surrounded by inner-city manufacturing, and then by high-density rings of workers' housing. Only the upper classes could escape the perceived ills of the core-oriented concentrations as street railways, tramways, and the omnibus provided access to more pleasant and lower-density environs.

A new concept was needed to capture the scale of the largest agglomerations. The authors of a report issued by the United States Bureau of the Census (1932) in the 1930s wrote that "the population of the corporate city frequently gives a very inadequate idea of the population massed in and around that city, constituting the greater city. [The boundaries of] large cities in few cases limit the urban population which that city represents or of which it is the center. If we are to have a correct picture of the massing or concentration of population in extensive urban areas it is necessary to establish *metropolitan districts* which will show the magnitude of each of the principal population centres." Spelling out the idea further, the Bureau of the Budget's Committee on Metropolitan Area Definition (1967) wrote: "The general concept of a *metropolitan area* is one of an integrated economic and social unit with a recognized large population nucleus."

The situation was both fluid and dynamic, however, and the form of the metropolis changed rapidly in the period of accelerating economic growth that followed World War II, facilitated by the new technologies that assumed ascendancy at this time. The concentrated industrial metropolis developed because proximity meant lower transportation and communication costs for those interdependent specialists who had to interact with each other frequently or intensively. But shortened distances meant higher densities and the costs of congestion, polluted environments, high rent, loss of privacy, and the like. The technological developments implemented in the fourth Kondratiev had the effect of reducing the constraints of geographic space and the costs of concentration. Modern transportation and communications made it possible for each succeeding generation to live farther apart, producing first, accelerated suburbanization and urban sprawl, and later, real deconcentration. In 1902 H. G. Wells had speculated about the possibility:

> Many of [the] railway-begotten "giant cities" will reach their maximum in the coming century [and] in all probability they . . .are destined to . . .dissection and diffusion . . .[*T*]*hese coming cities will not be, in the old sense, cities at all; they will present a new and entirely different phase of human distribution* [italics added] [T]he social history of the middle and latter third of the nineteenth century . . .[has been] the history of a gigantic rush of population into the magic radius of – for most people – four miles. . . .But . . .[n]ew forces, at present so potently centripetal in their influence, bring with them the distinct promise of a centrifugal application. . . .Great towns before this century presented rounded contours and grew as puff-ball swells; the modern Great City looks like something that has burst an intolerable envelope and splashed . . .the mere first rough expedient of far more convenient and rapid developments. . . .We are . . .in the early phase of a great development of centrifugal possibilities . . .[A] city of pedestrians is inexorably limited by a radius of about four miles . . .a horse-using city may grow out to seven or eight . . .[I]s it too much . . .to expect that the available area for even the common daily toilers of the great city of year 2000 . . .will have a radius of over one hundred miles? . . .[T]he city will diffuse itself until it has taken up . . .many of the characteristics of what is now country . . .[T]he country will take itself many of the qualities of the city. The old antithesis will . . .cease, the boundary lines will altogether disappear.[5]

These predictions were certainly realized in the United States. After 1950, growth of the service sector, increase in the number of "footloose" industries (including final processing of consumer goods using manufactured parts, and the aircraft, aerospace, and defense industries), rapid emergence of a "quaternary" sector of the economy (involving, for example, the research and development industry), expansion and interregional migration of the non-job-oriented population (for example, of retirees to Florida, Arizona, and California), rising governmental expenditures and overall rising real incomes, plus modern highways and the automobile – all served to produce yet another transformation of the economy and the urban system that confirmed H. G. Wells' forecasts. Not only did urban areas grow and disperse into sprawling metropolitan regions; advantages for economic growth were found during the fourth Kondratiev in former hinterland regions around the "outer rim" of the country as changing communications technology reduced the time and costs involved in previous heartland-hinterland relationships.

The changes were cumulative. First, regional growth within the context of the national pattern of heartland and hinter-land brought outlying regions to threshold sizes for local production of a wide variety of goods and services. But then, they developed alternative bases of expansion as changes in the definition of urban resources made their rapid advance, free of the traditional constraints of heartland-hinterland leverage, possible. Hence, the explosive metropolitan growth of the South, Southwest, and West, led by the tertiary and quaternary sectors. The outcome was that it became possible, by the end of the 1960s, to interpret the spatial structure of the United States as a pattern consisting of (1) metropolitan areas and (2) the intermetropolitan periphery. Except for thinly populated parts of the American interior, the inter-metropolitan periphery included "all the areas that intervened among metropolitan regions and that were the reverse image of the trend towards large scale concentrated settlement. . . .Like a devils' mirror . . .the periphery . . .developed a socioeconomic profile that perversely reflects the very opposite of metropolitan virility" (Friedmann and Miller 1965).

Even by 1960, much of the United States territory was covered by the daily commuting areas of its metropolitan centers, as the far-flung suburbs made possible by the automobile and by superhighway construction spread across the national landscape. The Greek planner Constantinos A. Doxiadis called these urban regions *daily urban systems* (Berry 1968). The coalescence of expanding metropolitan areas along the northeastern seaboard of the United States led Jean Gottman (1961) to coin a new term for the phenomenon – *megalopolis* – and a later author, somewhat facetiously, to call three such alleged developments "BosWash," "ChiPitts," and "SanSan," Peter Hall (1973) went on to document the extent of megalopolitan development elsewhere in the world, arguing that similar processes were unfolding in every economically advanced area.

It is these metropolitan regions that have been the subjects of extensive environmental analysis in the past half century, and it is from this analysis that we have been able to develop an understanding of the impacts of urbanization upon the biosphere.

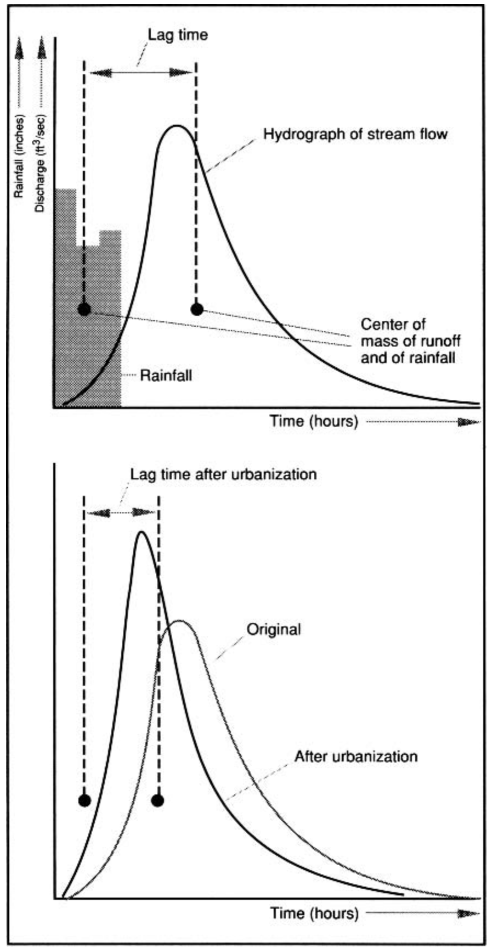
Environmental Effects of Metropolitan Growth

Urban modification of the atmospheric environment can occur at three geographic scales (Berry and Horton 1974):

1. Locally, by alterating in the nature of the effective surface: the replacement of the natural surface of soil, grass, and trees by the multiplicity of urban surfaces of brick, concrete, glass, and metal at different levels above the ground. These artificial materials change the nature of the reflecting and radiating surfaces, the heat exchange near the surface, and the aerodynamic roughness of the surface.
2. Regionally, by generating large amounts of heat artificially and by altering the composition of the atmosphere via emission of gaseous and solid pollutants. At certain times of the year in mid-latitude cities, artificial heat input into the atmosphere by combustion and metabolic processes may approach or even exceed that derived indirectly from the sun. The heat island that results serves as a trap for pollutants.
3. Potentially, globally, through urban contributions to the sulfur budget or to CO_2, and thus to the greenhouse effect, to global warming, and to sea-level changes that are likely to be of greatest consequence for major coastal cities.

Leopold (1968) records four interrelated but separable effects of local land-use changes on the hydrology: changes in peak-flow characteristics; changes in total runoff; changes in water quality; and changes in hydrologic amenities. Stream flows following rainstorms may be characterized by

Fig. 10 Unit hydrographs before and after urbanization



unit hydrographs that capture both the peakedness and the lags in the rainfall-discharge relationship (Fig. 10). After urbanization, runoff occurs more rapidly and with a greater peak flow than under nonurban conditions. Urbanization increases the impervious land area, and the urban area may be served by storm sewers. Both increase the peak discharge: maximum sewerage and imperviousness results in peak discharges that are more than six times greater than in unurbanized conditions. In their turn, sharper peak discharges increase flood frequencies (Fig. 11) and the ratio of overbank flows. Urbanization, then, increases the flood volume, the flood peak, and the flood frequency, and the flushing effect increases turbidity and pollutant loads, although sediment loads may fall and the channel response will therefore shift from aggradation to bank erosion (Wolman 1967). Water pollution, in its turn, changes the quality of the downstream resource, the ecology of the riverine environment, and the amenity value of the river bank or estuary. The effects become pronounced downstream of the larger cities, where natural flushing is incapable of preventing long-term damage.

At the scale of the metropolitan region, Landsberg's 1981 summary of the major changes in climates is, of course, well known (Table 4), but it needs to be discussed because it provides on-average estimates rather then insights into variations with city size.

The most dramatic effect of metropolitan growth is the creation of the urban heat island, which serves as a trap for atmospheric pollutants. Ceteris paribus, the temperature differential between the city core and the rural periphery increases with city size; the differences are small and ephemeral in places of 250,000 or less population, but are both substantial and longer-lasting in larger places. The

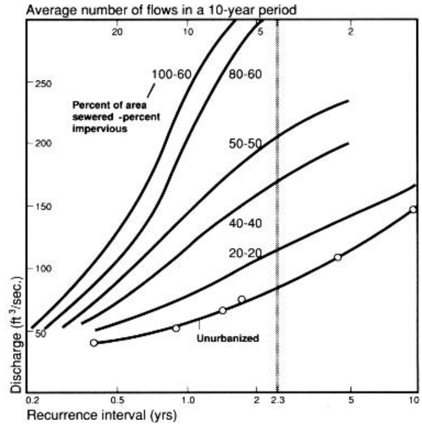


Fig. 11 Increasing flood frequencies as urbanization progresses

Table 4 On-Average Effects of Urbanization on the Climate of Cities

Element	Compared to rural environs
Contaminants	
Condensation nuclei	10 times more
Particulates	50 times more
Gaseous admixtures	5–25 times more
Radiation	
Total on horizontal surface	0–20% less
Ultraviolet, winter	30% less
Ultraviolet, summer	5% less
Sunshine duration	5–15% less
Cloudiness	
Clouds	5–10% more
Fog, winter	100% more
Fog, summer	30% more
Precipitation	
Amounts	5–15% more
Days with less than 5 mm	10% more
Snowfall, inner city	5–10% less
Snowfall, lee of city	10% more
Thunderstorms	10–15% more
Temperature	
Annual mean	0.5–3.0° C more
Winter minimums (average)	1–2° C more
Summer maximums	1–3° C more
Heating degree days	10% less
Relative Humidity	
Annual mean	6% less
Winter	2% less
Summer	8% less
Wind Speed	
Annual mean	20–30% less
Extreme gusts	10–20% less
Calm	5–20% more

Source: Landsberg 1981.

heat island expands and intensifies as the city grows, and stronger and stronger winds are needed to overcome it. Wind speeds of 5 $\mathrm{m/sec^{-1}}$ can eliminate the heat island in a city of 250,000, but speeds of 10 $\mathrm{m/sec^{-1}}$ are required when the size reaches 1,000,000, and 14 $\mathrm{m/sec^{-1}}$ at 10,000,000. Yet the surface roughness of the city serves to reduce wind speeds and inhibit this ventilation: average wind speed may be reduced as much as 30% by the big city. In the larger cities over 10,000,000, the mean annual minimum temperature may be as much as 4° F higher than that of the surrounding rural periphery. This difference is much greater in summer than in winter.

The causes are twofold, both of which are seasonally dependent. (1) In summer, the tall buildings, pavement, and concrete of the city absorb and store large amounts of solar radiation, and less of this energy is used for evaporation than in the country because of the high runoff. The stored energy is released at night, warming the urban air. (2) In winter, manmade energy used for heat and light produces the warming, yet the blanket of emissions reduces incoming radiation by as much as 20%. If the BosWash megalopolis reaches a population of 50–60 millions by the year 2000, it will be characterized by heat rejection of 65 $\mathrm{cal/cm^2/d}$. In winter, this is 50%, and in summer, 15%, of the heat received by solar radiation on a horizontal surface. In Manhattan, the heat produced by combustion alone in winter has been estimated to be two and one-half times the solar energy reaching the ground. This energy is trapped by the blanket of pollutants over the city, including particulates, water vapor, and carbon dioxide, and is reemitted downward to warm the ambient air.

In addition to the heat island, other climatic effects of urbanization – all increasing with city size – include greater cloudiness, fog, dust, and precipitation, but lower humidity. And as wind dissipates the heat island, a downwind urban heat plume is detectable in the atmosphere. Along this plume, there are increased precipitation, thunderstorm and hail probabilities. Beyond such regional-scale consequences, urban activities are a major source of CO_2 and of the fluorocarbons that, in combination, may affect future global climates and sea levels.

Urbanization and Environment in the Years Ahead

The local- and regional-scale environmental effects of urbanization are all big-city, high-density, maximum-imperviousness consequences of million-plus, core-oriented, high-rise concentrations. Between 1975 and 2000, the number of million-plus cities is expected to more than double, from 190 to 440 (Table 2). Where they coalesce into larger agglomerations, the environmental effects converge at even broader regional scale – one graphic example of which is provided by a schematic

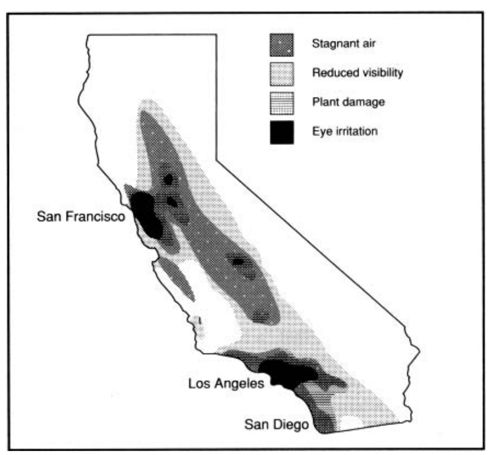


Fig. 12 The extent of air pollution in California

Table 5 The World's Urban Population, 1950–1985 (in thousands)

	1950		1960		1970		1975		1980		1985	
	pop.	%	pop.	%	pop.	%	pop.	%	pop.	%	pop.	%
World total	726,673	100	1,019,847	100	1,368,169	100	1,573,913	100	1,809,439	100	2,084,844	100
Percentage of total pop.	28.9		33.7		37.2		39.0		41.0		43.2	
Annual growth rate (%)			3.4		3.0		2.8		2.8		2.9	
Africa	32,434	4.5	50,416	4.9	80,644	5.9	103,832	6.6	134,951	7.5	174,829	8.4
Percentage of total pop.	14.8		18.4		22.8		25.6		28.8		32.1	
Annual growth rate (%)			4.5		4.8		5.2		5.4		5.3	
Latin America	67,465	9.3	106,520	10.4	162,075	11.8	197,250	12.5	238,283	13.2	285,274	13.7
Percentage of total pop.	41.2		49.5		57.3		61.2		64.7		67.8	
Annual growth rate (%)			4.7		4.3		4.0		3.9		3.7	
North America	106,018	14.6	133,280	13.1	159,493	11.7	170,167	10.8	181,433	10.0	194,871	9.3
Percentage of total pop.	63.8		67.1		70.5		72.0		73.7		75.4	
Annual growth rate (%)			2.3		1.8		1.3		1.3		1.4	
East Asia	112,638	15.5	199,855	19.6	276,808	20.2	322,530	20.5	371,199	20.5	425,010	20.4
Percentage of total pop.	16.7		24.5		28.2		30.3		32.7		35.3	
Annual growth rate (%)			5.9		3.3		3.1		2.9		2.7	
South Asia	112,507	15.5	158,717	15.6	234,924	17.2	286,228	18.2	352,827	19.5	437,409	21.0
Percentage of total pop.	15.9		18.3		21.2		22.8		24.8		27.2	
Annual growth rate (%)			3.5		4.0		4.0		4.3		4.4	
Europe	217,205	29.9	256,023	25.1	302,276	22.1	323,465	20.6	340,785	18.8	357,588	17.2
Percentage of total pop.	55.4		60.2		65.8		68.2		70.5		72.6	
Annual growth rate (%)			1.7		1.7		1.4		1.0		1.0	
Oceania	7,741	1.1	10,451	1.0	13,680	1.0	15,519	1.0	17,245	1.0	19,098	0.9
Percentage of total pop.	61.2		66.2		70.8		73.4		75.7		78.0	
Annual growth rate (%)			3.0		2.7		2.6		2.1		2.1	
USSR	70,765	9.7	104,589	10.3	138,270	10.1	154,923	9.8	172,715	9.5	190,765	9.2
Percentage of total pop.	39.3		48.8		56.7		60.9		64.8		68.2	
Annual growth rate (%)			4.0		2.8		2.3		2.2		2.0	

Source: United Nations 1980.

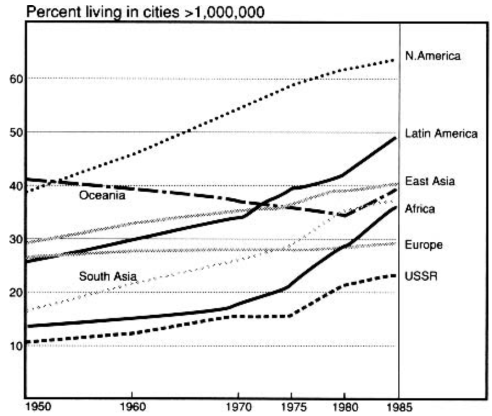


Fig. 13 Increases in the percentage of urban population living in million-plus cities

illustration of the extent of air pollution in California (Fig. 12; Berry and Horton 1974: 83). But the regional distribution of large agglomerations is changing rapidly, suggesting a new locus for such regional-scale environmental effects. Urbanization is increasing most speedily in Latin America, Africa, and South Asia (Table 5), and it is there that the most rapid increases in the proportion of the population concentrated in million-plus cities are taking place (Fig. 13). Between 1950 and 1985, Europe and North America's share of the world's urban population dropped from 45% to 26%, even while the numbers in their cities grew from 325 to 500 million. In the rest of the world, the urban population jumped from 400 million to almost 1,600 million. During the next Kondratiev, much of the world can be expected to urbanize as completely as the more developed regions, and what this means is that the Third World will have large numbers of very large urban regions. At the end of the Great Depression, the world's more developed regions were barely 40% urbanized, but by the 1980s the percentages were in the 70s. Today, the world's less developed regions are approaching the 40% level, and their urban growth is still accelerating.

For the next half century, much of the world will be experiencing the process of population concentration that has already ended in North America and western Europe. In the economically advanced regions, urbanization is not a stationary process, however; there is change in the nature of change, the environmental consequences of which are unclear. As metropolitan regions have rushed outward, urban densities and inner-city populations have dropped, leaving behind the most disadvantaged people in and around the former cores, inheriting the urban environments produced by earlier rounds of growth. Major reversals of patterns and reshaping of urban systems have been unfolding: what I have called a *process of counterurbanization* and what others have called *polarization reversal* or the *urban turnaround* (Ogden 1985; Richardson 1980; Vining and Kontuly 1978; Vining and Strauss 1977). Thus, in 1976 I wrote that "a turning point has been reached in the American experience. Counterurbanization has replaced urbanization as the dominant force shaping the nation's settlement patterns. . . .The process of counterurbanization has as its essence decreasing size, decreasing density, and decreasing heterogeneity: *counterurbanization is a process of population deconcentration; it implies a movement from a state of more concentration to a state of less concentration*." Counterurbanization is occurring in most of the world's advanced economies, often helped along by planning policies designed to stem big-city growth, and in the socialist world, by attempts to create the "city of socialist man" (Berry 1981).

One explanation for the turnaround is the existence of urban disamenities: that premiums have to be paid to do business in larger urban agglomerations, where the social costs and environmental burdens of urban living are greater. On the one hand, larger urban areas have been places of greater productivity, but on the other, they have been the locus of growing disamenities. If the disamenities

Table 6 Nodes of the Global Polycenter

Region (Urban complex)	Headquarters	Population
North America		
New York	90	17,100,000
Chicago	28	7,600,000
Los Angeles	22	8,900,000
San Francisco	15	4,400,000
Philadelphia-Wilmington	14	5,200,000
Dallas–Fort Worth	14	2,400,000
Houston	13	2,100,000
St. Louis	10	2,200,000
Detroit	9	4,800,000
Pittsburgh	9	2,000,000
Asia		
Tokyo	88	23,000,000
Osaka-Kobe	37	15,500,000
Seoul	9	6,800,000
Europe		
London	63	10,500,000
Paris	39	9,400,000
Ruhrgebiet	21	5,500,000
Frankfurt	13	1,600,000
Randstadt	9	2,000,000
Rome	4	3,600,000

keep growing and technological change reduces the productivity advantages of agglomeration, the point will be reached where a turnaround will occur and growth will disperse, and as growth disperses, the disamenities should be ameliorated. Some have argued that what is emerging are *urban civilizations without cities*. New technologies are compressing time and space and accelerating change, lessening the need for face-to-face contact because of instantaneous electronic communications in the new age of the computer. Populations are therefore moving into high-amenity areas newly endowed with electronic access, reducing the local- and regional-scale pressures that result in environmental modification at those scales. A new kind of much more harmonious relationship with natural environments that are valued appears to be emerging in such civilizations.

This is, however, not the end of the story. Other forces are beginning to exert their influences: (1) the reemergence of external economies as primary locational factors; (2) the rise of flexible production systems; (3) as a consequence, the reagglomeration of production in new regions in which the settlement pattern is polycentric; (4) the connection of these polycentric urban networks into a global system or polycenter organized by a limited number of complexes of multinational headquarters. One way to characterize the resulting urban systems is as dynamic networks. No longer are vertical hierarchies arranged regionally into heartlands and hinterlands; instead, the “cores” are centers of creativity and entrepreneurial activity wherever they may be located, and are linked into transnational networks. The important decisions are made in some 500 major private corporations, whose headquarters are dispersed throughout 19 great urban regions. These regions, tightly interlinked, constitute the polycenter of the global urban network (Table 6 and Fig. 14). This polycenter controls networks of interdependent specialists, and wherever such networks have penetrated, the old models of neat urban hierarchies topped by big cities, of heartlands and hinterlands, and of metropolitan and nonmetropolitan spaces have vanished. Yet we do not know if the environmental effects of the new-form settlements will be regional-scale disruptions of the biosphere (California style), or if they will signal an amelioration of the worst of the environmental effects of the large, core-oriented metropolis.

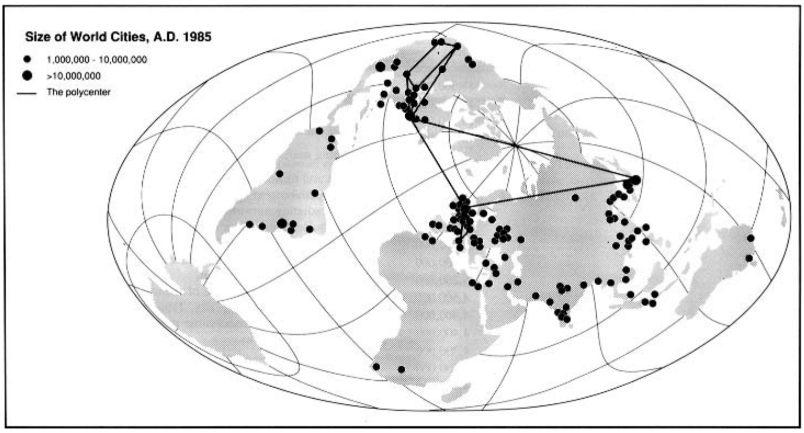


Fig. 14 The polycenter and the periphery. A. D. 1985

Meanwhile, in the world periphery beyond the polycenter, there is a growing list of very large cities in which the process of population concentration accompanied by local- and regional-scale environmental modification is being repeated. In 1975, Chandler's list included 19 urban regions that already had exceeded 4 million in population (Table 7). Another 46 had exceeded 2 million. As the dispersed polycenter continues to evolve in the First World, it is in these massive urban agglomerations, peripheral to the main chanels of global interdependence, that the greatest modifications of the biosphere will occur, changing the regional environments within which a growing proportion of the world's population will live marginal lives, pressed to the threshold of subsistence. The scale of Third World urban growth is such that even if First World environmental impacts are significantly reduced, the reductions will be swamped by the increases occurring elsewhere. It is from the Third World's economic growth and urban concentration that the most serious regional threats to the global environment will come.

Table 7 Urban Regions outside the Global Polycenter with Populations Exceeding 4,000,000 in 1975

Mexico City	11,300,000
Moscow	10,700,000
Sao Paulo	10,000,000
Buenos Aires	8,400,000
Cairo	8,400,000
Rio de Janeiro	8,300,000
Shanghai	8,000,000
Calcutta	7,800,000
Bombay	7,000,000
Manila	5,400,000
Jakarta	5,300,000
Peking	5,200,000
Tientsin	4,600,000
Karachi	4,400,000
Delhi	4,400,000
Bangkok	4,300,000
Leningrad	4,200,000
Tehran	4,300,000
Madrid	4,100,000

Source: Chandler 1987: 511.

Notes

1. The city sizes used in this chapter are those reported by Chandler 1987.
2. Chinese cities were built according to cosmological principles derived from Han Confucianism (Wheatley 1971). They were always walled cities, constructed according to a preconceived plan, in a regular and formalized pattern, an ecological symbolization of the cosmic order: first, as an *axis mundi* symbolizing the powerful centripetal forces in the universe; second, with the most important buildings arranged along a cardinally oriented ceremonial axis (the "celestial meridian writ small"; Wheatley 1971: 456); third, centered on a sacred enclave; fourth, walled in the form of a square; fifth, facing in the propitious southerly direction. The physical design was embedded in a cosmology that incorporated explicit views about the wholeness of humankind and nature (Eisenstadt and Shachar 1987: 140). In contrast to the emergent Western views of humans' dominion over nature that structured environmental attitudes as growing urban demands in Europe sparked both agricultural development and imperial expansion. Han Confucianism advanced ideas of the organic unity of humankind and nature, ineluctably and physically inter-dependent, with an order that had to be maintained by tradition and ritual.
3. The key elements of Japanese urban design also were determined by the Chinese urban model. At the core of the capital was the emperor's residence and the principal courts and temples, encircled by downwardly sloping gradients of status arranged with respect to a cardinally oriented square-grid design, reinforcing the fundamental principles of centrality and concentration, of the stability of social hierarchies, and of the unity of existence.
4. The overall pattern was one of the city-centered organization of economic activities along three dimensions:

    1. A heartland-hinterland arrangement of industrial and resource regions linked by intermetropolitan flows.
    2. A system of cities within each region, arranged in a hierarchy according to the functions performed by each.
    3. Corresponding areas of urban influence or urban fields surrounding, each of the cities in the system.

    Within this framework, impulses of economic change and of environmental modification and exploitation were transmitted simultaneously along three planes:

    1. Outward from heartland centers to those of the regional hinterlands on a national scale.
    2. From each regional capital to centers of lower level in the hierarchy, in a pattern of "hierarchical diffusion" within each of the regions.
    3. Outward from urban centers into their urban fields, radiating "spread effects" into the surrounding countryside.

    The resulting spatial patterns were these:

    1. The size and functions of a central city, the size of its urban field, and the spatial extent of development and environmental "spread effects" radiating outward from it were proportional.
    2. Since impulses of economic change were transmitted in order from higher to lower centers in the urban hierarchy, continued innovation in large cities remained critical for extension of growth over the complete economic system.
    3. The resulting spatial incidence of economic growth was a function of distance from the central city. Troughs of economic backwardness lay in the most inaccessible peripheries of the lowest-level centers in the hierarchy.
    4. The growth potential of any area situated along an axis between two cities became a function of the intensity of interaction between them, which in turn was a function of their relative location and the quality of transportation arteries connecting them.

5. Similar ideas had been expressed by Adna Weber in 1899: "the 'rise of the suburbs' it is, which furnishes the solid basis of a hope that the evils of city life, so far as they result from overcrowding, may in large part be removed. If concentration of population seems destined to continue, it will be a modified concentration which offers the advantages of both city and country life . . .a complete fusion of their different modes of life and combination of the advantages of both, such as no country in the world has ever seen." (Weber 1899: 475)

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Urban Ecology as an Interdisciplinary Field: Differences in the use of "Urban" Between the Social and Natural Sciences

Nancy E. McIntyre, K. Knowles-Yánez, and D. Hope

"If you wish to converse with me, define your terms." —attributed to Voltaire, The Home Book of Quotations: Classical and Modern, Fourth edition (B. Stevenson, ed.), p. 428, Dodd, Mead and Co., New York, NY, 1944

Abstract Though there is a growing appreciation of the importance of research on urban ecosystems, the question of what constitutes an urban ecosystem remains. Although a human-dominated ecosystem is sometimes considered to be an accurate description of an urban ecosystem, describing an ecosystem as human-dominated does not adequately take into account the history of development, sphere of influence, and potential impacts required in order to understand the true nature of an urban ecosystem. While recognizing that no single definition of "urban" is possible or even necessary, we explore the importance of attaching an interdisciplinary, quantitative, and considered description of an urban ecosystem such that projects and findings are easier to compare, repeat, and build upon. Natural science research about urban ecosystems, particularly in the field of ecology, often includes only a tacit assumption about what urban means. Following the lead of a more developed social science literature on urban issues, we make suggestions towards a consistent, quantitative description of urban that would take into account the dynamic and heterogeneous physical and social characteristics of an urban ecosystem. We provide case studies that illustrate how social and natural scientists might collaborate in research where a more clearly understood definition of "urban" would be desirable.

Keywords: Urban · social science · ecology · definition of "urban"

Introduction

There is now compelling evidence that humans are altering virtually all of the Earth's ecosystems. Vitousek *et al*. (1997) noted that more than half of the Earth's fresh water is used by humans, nearly half of the land surface has been transformed by human action, more atmospheric nitrogen is fixed by human activities than by all natural terrestrial processes combined, and human activities are leading to significant losses of biodiversity. As a consequence of these actions, most (if not all) ecosystems can arguably be considered human-dominated ecosystems, regardless of whether humans actually occupy (live within) them. However, humans are also creating new ecosystems specifically for dwelling: these are urban ecosystems (Stearns and Montag, 1974). These new, synthetic ecosystems are unquestionably human-dominated, and yet urban ecosystems are both qualitatively and quantitatively different from other human-dominated ecosystems in terms of development, sphere of

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influence, and potential impacts. Understanding these differences hinges on our ability to distinguish between urban and human-dominated.

In 1900, only 9% of the world's human population lived in "urban environments" (so called by the World Bank, 1984). This figure had increased to 40% by 1980, 50% by 2000, and is expected to increase to over 66% by 2025 (World Bank, 1984; Simpson, 1993; Rodick, 1995; Brockerhoff, 1996). The increasing abundance of such environments has not gone unnoticed by ecologists. Indeed, the "urban landscape is a well-appreciated form of landscape" (Weddle, 1986) and offers numerous areas of scientific study since urban landscapes are often characterized by the presence of exotic flora and fauna, an imbalance between biotic immigration and extinction rates (Rebele, 1994), and the presence of pollution in air, water, and soil (Botkin and Beveridge, 1997). Until recently, however, relatively little ecological research was conducted in urban settings. Indeed, ecologists have primarily sought to understand their subjects of study in the absence of humans and have usually considered humans chiefly as agents of disturbance (Pickett and McDonnell, 1993; Costanza, 1996). A recent survey of research papers in the foremost ecological journals between 1993–1997 revealed that only 25 of 6157 papers (0.4%) dealt specifically with urban species or were carried out in an urban setting (Collins *et al.*, in review). Ecologists have recently begun to recognize this oversight, however, calling for more research into how overall ecosystem structure and function shape and are shaped by urban development (Matson, 1990; McDonnell and Pickett, 1990; Botkin and Beveridge, 1997; Walbridge, 1997; Parlange, 1998).

Such a pursuit is not as easy as it sounds, however. There are logistical problems incurred from working in urban settings, such as difficulty in obtaining permission to conduct large-scale experiments on private property, as well as vandalism to field equipment (see also Yalden, 1980). Similar problems are faced in most ecological studies. What is different about conducting research on how patterns of urbanization affect ecological processes, though, is in determining what constitutes an "urban ecosystem." By this, we mean recognition that an urban area is not simply a human-dominated area.

Fundamentally, a landscape defined as urban shows some effects of human influence. Taken literally, this could mean that the most remote sites could be called urban simply because humans have influenced a portion of their area at some point in time (e.g., by the presence of a secluded vacation cabin or even aboriginal ruins). Clearly, this description of urban is too broad to be very useful, and it confounds the differences between human-dominated and truly urban ecosystems. There is thus an evident need to remove the uncertainty with which ecologists define urban ecosystems and to correct oversights regarding definitions (or lack thereof) of what it means to be "urban."

In this paper, we will examine definitions of the term "urban" as used by ecologists and look towards the social sciences for guidance in creating a more quantitative description of what an urban ecosystem is. In recognizing that urbanization is both an ecological and a social phenomenon (thereby recognizing that urban ecology is an interdisciplinary field), we will compare and contrast definitions of "urban" as used by ecologists and social scientists and will provide some case studies of how interdisciplinary studies have defined and used "urban." This process will illuminate both strengths and gaps in current descriptions of urban systems. It will also provide both ecologists and social scientists with guidelines to quantify the urban setting of their own research projects. Although we recognize that no single definition of "urban" is necessarily more correct than another, we will demonstrate that demographic, economic, political, perceptive, and cultural criteria, when used in conjunction with geophysical and biological criteria, provide a more complete and useful definition.

How Ecologists have used the Term "Urban": a Review

Ecology studies the relationships between organisms and their environments. To determine how ecologists have described urban environments, we reviewed 63 ecological journal papers (Appendix 1). Using the Institute of Scientific Information's Web of Science database (http://www.isinet.com), we located research papers with "urban," "urbanization" (including British spelling of "urbanisation"), "city," "cities," "metropolis," or "suburb" in the title, abstract or keywords. For each reference, we noted whether "urban" was defined (and if so, what the definition was). We included only papers that considered how urban environmental features affect the abundance and distribution of organisms other than humans (the traditional focus of ecology), excluding papers that dealt strictly with environmental health/medicine, epidemiology, pest management, or ecotoxicology in urban settings. Excluding papers from these disciplines made our review potentially more relevant to both basic and applied research questions about urban ecosystems in general, whereas environmental health/medicine, epidemiology, pest management, and ecotoxicology are often localized in focus to meet specific applications. We did not exclude papers that assessed biodiversity along gradients of pollution, for example, merely because such papers included a variable that happened to be potentially of interest to environmental health/medicine, epidemiology, pest management, or ecotoxicology. Rather, we included papers based on the criterion of whether the primary research focus was ecological. While we acknowledge that other natural sciences (and, generally speaking, other sciences) may provide important insights in the study of urban ecosystems, we feel that ecology is the foremost and most inclusive of these, based simply on the definition of ecology. As with any review, particularly on a topic as diverse as urban ecology, it is likely that we overlooked some appropriate papers. However, the sheer volume and variety of papers that we did uncover should provide a reasonable assessment of how ecologists have used the term "urban."

Types of Ecological Studies in Urban Environments

Cicero (1989) identified four types of ecological studies in urban environments, of which we found examples of each (detailed below): (1) comparison of different land-use types within an urban setting (e.g., Woolfenden and Rohwer, 1969b; DeGraaf and Wentworth, 1981), (2) comparison of an urban area with a nearby "natural" area (e.g., Woolfenden and Rohwer, 1969a; Emlen, 1974; Guthrie, 1974), (3) gradient analysis (e.g., Klausnitzer and Richter, 1983; Ruszczyk, 1986; Ruszczyk *et al*., 1987; Sustek, 1987, 1992, 1993; Ruszczyk and de Araujo, 1992; Goszczynski *et al*., 1993; Pouyat *et al*., 1994; Medley *et al*., 1995; Blair, 1996; Natuhara and Imai, 1996; Blair and Launer, 1997; McDonnell *et al*., 1997), and (4) studies of urban development dynamics by monitoring a single area over time (e.g., Erz, 1964; Walcott, 1974; Rosenberg *et al*., 1987). In addition to these four types of urban ecological studies, there is also ecological "footprint" analysis (Rees, 1996; Wackernagel and Rees, 1997).

Comparisons within an urban setting—these studies acknowledged that a city is heterogeneous in space, meaning that an urban environment is a mosaic of areas that have different physical properties and uses (Hohtola, 1978; Dulisz and Nowakowski, 1996; Hadidian *et al*., 1997; Sewell and Catterall, 1998). In terms of comparing different land-use types within an urban area, relatively few studies calculated structural variables such as percent vegetative cover, average building height, or housing density (e.g., Ruszczyk *et al*., 1987; Edgar and Kershaw, 1994; Friesen *et al*., 1995; Blanco and Velasco, 1996; Natuhara and Imai, 1996). More usually, the researchers defined the urban environment in terms of generic land-use categories, such as "residential yard" or "park" (e.g., Hooper *et al*., 1975; Weber, 1975; Huhtalo and Jarvinen, 1977; Hohtola, 1978; Goszczynski *et al*., 1993; Jokimaki and Suhonen, 1993; Mirabella *et al*., 1996; Hadidian *et al*., 1997; Sewell and Catterall, 1998).

"Urban" vs. "natural"—these studies made comparisons between urban areas and undeveloped rural areas outside a city (Woolfenden and Rohwer, 1969a; Emlen, 1974; Guthrie, 1974). These studies assumed that the undeveloped areas represented regions outside human influence, with "urban" characterized by the presence of humans and "natural" by their absence. "Urban" and "natural"

environments were thus viewed by ecologists as being at opposite ends of a spectrum, representing a dichotomy of history, structure, function, and value (McDonnell *et al*., 1997; Walbridge, 1997).

Gradient analysis—in these studies, the ecological effects of urbanization were assessed along a gradient (usually simply the distance from a city's geographic center and not quantified as a gradient of variables such as housing density, air quality, etc.). With a gradient approach, the potential difficulty in assigning clear boundaries to urban ecosystems is recognized implicitly. In its most literal sense, gradient analysis invokes a spatially continuous linear gradient from urban core to rural exterior, as envisaged in simple models such as the concentric theory of urban structure and growth (Park *et al*., 1925). Biotic, physical, and social variables (e.g., human population density, housing density, traffic volume, air quality, species richness) are then correlated with position along this gradient. Distance from the urban center often provides a useful first cut for determining whether spatial correlations exist between increasing human activity and ecological response variables, with more proximate explanatory variables and mechanisms left for more detailed follow-up studies.

Many cities, however, particularly those that have developed in the last 10–20 years, consist of multiple cores, have hard boundaries, and grow by rapid, leap-frog development over remnants of undeveloped open space. Moreover, some urban-related effects do not decrease in intensity in a simple linear or concentric pattern from a single center. For example, the patterns of urban air pollution derived from fossil fuel combustion frequently occur in "complex terrains," where synoptic and topographically controlled airflows lead to patterns in the concentrations of nitrous oxide, fine particulates, and low-level ozone that may show little correspondence to the spatial distribution of urban core areas (Plaza *et al*., 1997; Sillman, 1999; Ellis *et al*., in press). In such situations, there is not a simple linear decrease in urbanization with distance from the city center. A more indirect form of gradient analysis is therefore necessary, whereby ecological variables are measured and used to construct a gradient of urbanization from sites dispersed across an urban area (e.g., Medley *et al*., 1995; McDonnell *et al*., 1997). In such cases, having easily quantifiable, independent criteria by which the urban gradient is defined is particularly important. Although gradient analysis is useful in pointing out that the influences of urbanization are continuous and not binary, the lack of gradient quantification we observed in most studies was disturbing. In addition, applications of gradient analysis to date have simply correlated environmental disturbances to aggregated or very simple measures of urbanization (Alberti, 1999).

Urban succession—Very few studies examined how ecological patterns and processes in urban mosaics change over time or attempted to quantify rates of change in the urban setting and compare how these rates differed from the same processes occurring in semi-natural and natural ecosystems (Walcott, 1974). In those studies that did consider a temporal component to urbanization, urbanization was seen as a form of disturbance, creating a "natural experiment" (*sensu* Diamond, 1986). Disturbance from urbanization was then treated in a similar fashion to effects from fire or flood, allowing comparisons to be made between disturbed and undisturbed areas or over time in a single area (pre-disturbance vs. post-disturbance) (Erz, 1964; Rosenberg *et al*., 1987).

Ecological "footprint" analysis—ecological economics attempts to view "socioeconomic system(s) as a part of the overall ecosphere, emphasizing carrying capacity and scale issues in relation to the growth of the human population and its activities" (Costanza *et al*., 1997). As an extension of ecological economics, ecological "footprint" analysis seeks to convert the material and energy flows required to sustain the human population and industrial metabolism in a defined area (regional, national, or local, e.g., a city) into a land/ecosystem area equivalent (Folke *et al*., 1997; Rees, 1996). That is, rather than considering the impact of a concentrated population only on its immediate surroundings, ecological "footprint" analysis takes into account the effects of humans on both their immediate surroundings and areas of influence much further afield. In "footprint" analysis, cities are typically defined as areas dominated by "consumed or degraded land", i.e., the "built environment" (Wackernagel and Rees, 1997). Defining boundaries in this way has been a source of criticism, particularly where a large urban region cuts across natural ecosystem boundaries (van den

Bergh and Verbruggen, 1999). More consistent and interdisciplinary ways of defining what is urban would benefit "footprint" analysis and help to resolved problems of comparability of data between disciplines.

These five types of urban ecological studies represent different approaches to a common subject. Shortcomings with how "urban" was defined were found in all five types.

Some Problems with Definitions of "Urban" in Ecology

For the vast majority of the papers we reviewed, the definition of what was urban was simply assumed, not defined explicitly, in much the same way that the definition of "forest" is assumed to be known to readers. Where the system was defined at all, it was usually done using general and indefinite terms (Table 1). For example, consider this statement: "The term 'urbanization' refers to development . . . such as road and building construction, and other changes of land use from rural to residential and industrial that result in an increase of impermeable surface, accumulation of toxic substances, increase of domestic wastewater load, and increase in water demand due to increased human population . . ." (Kemp and Spotila, 1997). In the field of urban landscape planning, a typical definition of "urban" is similar to that given by Hendrix *et al*. (1988): "All residential land at densities greater than one dwelling unit per acre, all commercial and public institutions, railyards,

Table 1 Examples of how "urban" has been defined in ecology and in various social sciences, with some strengths and weaknesses of each definition

Discipline	Reference	Definition of "urban"	Strengths	Weaknesses
Ecology	Emlen, 1974	Area consisting of "houses and lawns"	Mentions specific features	does not include density
Ecology	Erskine, 1992	"Built-up" area	Brief	Vague
Ecology	Odum, 1997	Area that uses at least $100,000\,\text{kcal/m}^2/\text{yr}$	Based on international currency	Difficult to measure accurately
Sociology	U.S. Bureau of the Census	Area with > 2500 people (620 individuals/km^2)	Precise, includes population density	Arbitrary
Sociology	United Nations, 1968	Area with > 20,000 people	Precise	Arbitrary; does not include density
Economics	Mills and Hamilton, 1989	"A place [with] a minimum population density [and] also a minimum total population"	Accounts for both human abundance and density	Does density not specify the minimum density or total population
Environmental psychology	Herzog and Chernick, 2000	Area with a "driveway between buildings . . .[and] paved parking areas with older buildings in the background"	Emphasizes presence of human-built structures (especially those associated with transportation)	Does not include direct presence of humans (only indirect) or human density
Planning	Hendrix *et al*., 1988	"All residential land at densities greater than one dwelling unit per acre, all commercial and public institutions, railyards, truckyards and highways"	Includes goods-and service-providers, transportation elements, and dwellings at a specified density	Density is arbitrary, boundaries are unspecified

truckyards and highways." Such definitions are thin and do not allow results to be compared adequately between different urban systems that may have different attributes. In another typical case, "urban" was defined as simply "built-up" (Erskine, 1992). In still another paper, "urban" referred to any area under human influence, such as a managed orchard in a rural setting (Majzlan and Holecova, 1993). Such variety in how these terms were used indicates a lack of consensus on what is meant by urban, resulting in a lack of rigor in the use of terminology (see Mack, 1999 for a discussion of how lax usage of terminology in ecology in general has created problems in the processes of peer review and publication).

Our review thus revealed that most ecological studies of urban environments have treated cities as another biome, an anthropogenic analog of a desert or temperate forest (Botkin and Beveridge, 1997). However, whereas "natural" biomes are defined by ecologists on the basis of variables such as temperature, precipitation, soil, and dominant vegetation types, urban environments appear to be determined solely on the basis of human presence. Urban studies assumed that such an environment would simply be recognized without needing to be defined clearly, with characteristics of the urban environment therefore usually not quantified. More often a qualitative (categorical) approach was used, usually based on simply the presence of human constructions. This vagueness precludes comparative studies from being made with any degree of precision.

However, there were some rare and relatively recent exceptions to this lack of urban quantification (e.g., Ruszczyk *et al*., 1987; Ruszczyk and de Araujo, 1992; Edgar and Kershaw, 1994; Friesen *et al*., 1995; Medley *et al*., 1995; Blanco and Velasco, 1996; Natuhara and Imai, 1996; and Blair and Launer, 1997). These studies measured a variety of physical (and some socioeconomic) variables, usually along a gradient of urbanization (usually density of people or buildings). We also noted a related trend in how urban ecosystems were considered over time. Earlier studies were largely descriptive and examined plants or animals that happened to occur in cities, whereas more recent studies have been more quantitative and were established within an urban ecology framework (ecology *of* urban ecosystems as opposed to ecology *in* urban ecosystems, *sensu* Fisher, 1997). Despite this trend, there is still a dearth of published studies that deal explicitly with urban ecology. Approximately half of the papers (46%) we reviewed were written before 1990, indicating that there has been interest in urban ecology for some years. Indeed, as early as 1935, there was a plea for ecologists to integrate human effects into their discipline (Adams, 1935). Serious scholarly interest in urban ecology thus evidently occurred rather early on in ecology, but preoccupation with the world wars, limited funding, and a lack of technological sophistication needed to address issues at city-sized spatial scales (e.g., remote sensing technology) may have precluded advancement of the field until recently (see also Adams, 1940; Lindeman, 1940). Accelerating human population growth, however, has made the issue immediately pressing (Pickett and McDonnell, 1993).

Most strikingly noted, however, was the fact that the majority of the studies we examined lacked an explicit understanding of the socioeconomic components of an urban system. Unlike ecology and other natural sciences, the social sciences were developed expressly for the purpose of examining human systems. Therefore, we culled definitions and techniques from a variety of social sciences that will help ecologists integrate social variables in ways that are appropriate for their own unique research aims. Although there exists much social science literature that possesses only a tacit understanding of what urban means (as in much of the ecological literature), we present attributes that characterize urban settings that are routinely included in social science studies and suggest a place for them in ecological studies.

Taking Cues from the Social Sciences

The social sciences (including, but not limited to, anthropology, political science, economics, planning, sociology, and environmental psychology) provide many examples of how urban areas have been defined in their respective disciplines (see, for example, Macura, 1961). Social scientists routinely distinguish among the terms "urban place," "urbanized area," "metropolitan statistical area," and "city," all of which are considered "urban" (Mills and Hamilton, 1989). The qualitative analysis of definitions of "urban" in social sciences provided here is meant to be neither comprehensive nor representative of all social science understandings of what is urban. Rather, our intent is to introduce the reader to some representative definitions of what is urban to social scientists and to introduce the idea that even within the social sciences, how and why "urban" is defined varies.

There are some standardized definitions of "urban" that are often used in the social sciences, but even these standardized definitions vary (Table 1). For example, the U.S. Bureau of the Census defines "urban" as an area with more than 2500 people (>620 individuals/km^2; http://www.census.gov/population/censusdata/urdef.txt). The United Nations, on the other hand, defines "urban" as an area with more than 20,000 people (United Nations, 1968, p. 38). These numbers are somewhat arbitrary and thus necessarily incur problems (e.g., is an area with 19,999 people substantially different from a truly "urban" site?), but they do provide useful boundaries. Indeed, "urban" does not necessarily need to refer to a concretely quantifiable category; the important point in defining what is "urban" is to provide a relative benchmark with which to compare studies.

Differences in the description of "urban" within the social sciences focus on various points of interest, often to the exclusion of attributes of interest to another discipline (Table 1). For example, an economist may refer to an urban area as "a political unit that generally contains more than 25,000 individuals . . .[where] increases in urbanization just mean increases in the share of the population living in these political units" (Glaeser, 1998, p. 2). This definition of urban is based on human population density within a given political unit. Another definition, however, may refer to a city more casually and state that "a city is just a dense agglomeration of people and firms" (Glaeser, 1998, p. 140), a definition that also notes density (although its exact measure is left unspecified), but with a nod towards the institutions that make up the urban framework.

A regional planner may take a more descriptive approach in defining what is urban, paying more attention to the structural divisions of urban areas:

> Our image of a city consists not only of people but also of buildings—the homes, offices, and factories in which residents and workers live and produce. This built environment forms contours which structure social relations, causing commonalities of gender, sexual orientation, race, ethnicity, and class to assume spatial identities. Social groups, in turn, imprint themselves physically on the urban structure through the formation of communities, competition for territory, and segregation . . .(Fainstein, 1994, p. 1)

A sociologist may use an even broader description that focuses on the presence of certain signposts that we culturally associate with cities (e.g., presence of centers for performing arts) or density-dependent differences in relationships among people (e.g., greater number of contacts with others, more secondary rather than primary relationships, and degrees of division and specialization of labor; Forman and Godron, 1986).

Results from empirical research in environmental psychology suggest that people perceive and conceptualize "urban" and "natural" environments differently (Kaplan *et al*., 1972; Kaplan, 1987; Herzog, 1989). Results of studies that focused on the perceived content of urban and natural environments and research on the relationships between environmental complexity and preference for certain environments (e.g., Kaplan *et al*., 1972; Kaplan, 1987) suggest that the mental categorization of landscapes into groupings such as "urban" and "natural" is based on different cognitive and affective criteria and content domains. For example, by using measures of brain-wave patterns, heart rate, and other physiological variables, Ulrich (1981) and Ulrich et al. (1991) found that scenes of non-developed (natural) environments have a more positive influence on human emotional states and stress levels than do urban scenes. The implication of these studies is that humans apparently perceive and react differently to natural versus urban settings, and therein lies yet another way by which a social scientist may define "urban." Since perceptions are integral to people's motivations and actions, using a perceptually based definition of urban provides a key link in the cultural, political, physical, perceptual, and economic aspects that must be integrated in urban ecology. If the definition of "urban" incorporated perceptual variables known to be salient to a person's discrimination between urban and natural environments, then the definition may be useful to interdisciplinary approaches to urban ecology.

To be sure, these definitions and descriptions of "urban" in social science disciplines are neither all-encompassing nor necessarily representative of their respective disciplines, but they are a reflection of the varied factors found in social scientists' understanding of urban areas: these definitions ask us to reflect upon the cultural and socioeconomic as well as the physical setting in defining an urban ecosystem.

Integration of Social and Natural Science Definitions of "Urban" in Urban Ecology

In order to characterize the many physical, cultural, and socioeconomic characteristics of urban areas, a researcher must have certain social science skills and resources, which we do not expect all ecologists to possess. The best solution to this conundrum is collaboration between natural and social scientists, but even in circumstances where this is not possible it still remains within the means of ecologists to include quantified descriptions of their urban study sites. We can agree upon factors that constitute urban and agree to describe certain physical and socioeconomic and political aspects of a place as we seek to develop comparative urban studies. There are a number of potential attributes that could be used to quantify and define what urban means. So that there is comparability among independent research efforts, we propose that researchers describe the following attributes of their urban study sites:

Demography—a demographic description may work well at the large, whole-system scale. However, at a more detailed, within-city level, some demographic patterns may be misleading. For example, wealthier parts of cities may have relatively low human density but have a structure and function that is urban in other respects. The necessary variables to assess demography are:

- population density (e.g., by age, income level, etc.)
- economic characteristics (e.g., average housing value)
- governance type

Physical geography—based on one or several common attributes of urbanization, including:

- area (e.g., U.S. Bureau of the Census's Primary Metropolitan Statistical Area)
- growth pattern (see Ewing, 1997; Forbes, 1997)
- distance to other urban areas
- description of urban morphology (protocol in Weitz and Moore, 1998)
- study scale (grain and extent; Kotliar and Wiens, 1990)
- historical, current, and adjacent land-use types
- land-cover type
- age since conversion from indigenous habitat
- housing type and density
- road type and density
- traffic frequency

Ecological process rates—based on locating where rates of ecological processes change rapidly. For example, it is already well established that fossil fuel combustion increases nitrogen and sulfur deposition across large areas within and adjacent to industrial areas, which leads to increases in net nitrogen availability, nitrification, and nitrate leaching (Aber, 1993). Rapid assays of such changes could be used to delimit the urban system.

Energy—the feature that distinguishes an urban ecosystem unambiguously from its surroundings is the level of energy use, with the amount of energy consumed per unit area per year in an urban environment being at least an order of magnitude greater than in other ecosystems (Odum, 1997). For example, data from Brussels (Belgium) have shown that in addition to the annual natural radiation inputs of 58×10^{12} kcal/yr, there is an additional energy use of 32×10^{12} kcal/yr from fossil fuel combustion, more than half the natural total (Sukkopp, 1990). A distinguishing characteristic of urban ecosystems is thus the high levels of energy use per year per area, largely derived from the combustion of fossil fuels, that is required to construct and maintain the urban infrastructure. Thus, although urban areas cover a relatively small percentage of the Earth's surface, they represent "hot spots" of energy use. Although in practice it may not always be easy to obtain detailed estimates of annual energy use, the advantages of using energy to define urban from other ecosystems are several. First, annual energy use could be used to place urban ecosystems in context with non-urban but human-dominated systems (e.g., agroecosystems). Second, a measure of annual energy use provides a means of defining urban systems independently of culture and uses units (e.g., kilocalories/km^2/yr) that have international currency. Third, since there is a monetary cost involved in producing energy for human use, energy units can be translated into local monetary units and hence serve as a means of linking ecological and socioeconomic studies.

Our approach focuses on urban properties that can be measured directly and relatively quickly and easily. Researchers should not be limited by this list and should add other important attributes as they see fit. In addition to these socioeconomic data, researchers should maintain an extensive and on-going pictoral inventory of their study site. This will aid in keeping track of changes in rural-to-urban conversion, urbanization, and allow the opportunity to test human perceptual and conceptual responses to changes in the landscape under study (Palmer and Lankhorst, 1998). Some of the data listed above are categorical (e.g., type of urban morphology, some land-use characteristics, growth pattern, and governance type) and are therefore amenable to a checkbox-style datasheet. Some of the data are relatively easy to obtain. For example, U.S. demographic data are widely available at university and local libraries as well as over the Internet (http://www.census.gov). Land use and growth information are generally available from a local planning office. These attributes can then be used to characterize three fundamental aspects of urbanization which determine when "rural" becomes "urban": timing, dynamics, and boundaries (Pond and Yeates, 1994).

There are some precedents in the existing ecological literature where social factors were incorporated into ecological studies, such as human population density, traffic volume, and land use (e.g., Medley *et al*., 1995; McDonnell *et al*., 1997; Wear and Bolstad, 1998). However, these studies were the exception rather than the norm. There were numerous cases where an ecological study would have been strengthened had it quantified the urban area in detail and included an explicit definition of "urban." For example, Faeth and Kane (1978) research examined how the number of flies and beetles in parks in Cincinnati, Ohio (USA), varied with park size. They found that species richness and abundance were positively related to park size, but there was some variation in these responses. Faeth and Kane did not quantify the parks individually, nor did they quantify the setting surrounding each park. It is possible that the parks' physical and socioeconomic surroundings affected the species present, but without inclusion of urban variables relating to park context, this remains uncertain.

Because an urban environment is both a physical and a social entity in its creation, functioning, and future, an in-depth understanding of what constitutes an "urban" ecosystem requires integration of natural and social sciences (Stearns and Montag, 1974; Pickett and McDonnell, 1993; Pickett *et al*., 1997). Each discipline would be strengthened if it were to include variables usually attributed to the other. We thus recommend that social scientists include a more ecological description of "urban" in their work, because information on the ecological status of cities may be very informative to various social aspects of urban life (e.g., quality of life, or purchasing or mobility decisions made by people of different socioeconomic levels). For example, information about the occurrence and abundance of scorpions (a household pest with a potentially lethal sting) in Phoenix, Arizona (USA), may be useful to homeowners by informing them of scorpion-prone areas and to urban planners in deciding where future development in Phoenix should occur so as to minimize human-scorpion encounters (McIntyre, 1999). The field of urban ecology would thus greatly benefit from reciprocity. To illustrate this kinds of understanding we may gain from interdisciplinary, reciprocal urban research, we outline two case studies below.

Case Studies of Integrative Urban Ecology

Vegetation Structure vs. Socioeconomic Index in Baltimore, Maryland (USA)

The work of Grove and Burch (1997) highlights some features of how an integrated approach to the study of urban ecosystems can bring about an understanding not possible with a single disciplinary approach. Grove and Burch (1997) research explored the reciprocal relationship between social differentiation (land use and residential stratification) and vegetation structure. Their study focused on how spatial patterns of urban development and land-use dynamics have influenced cycles and fluxes of critical resources (e.g., energy, materials, nutrients, genetic and nongenetic information, population, labor, and capital). A particular focus of their research was how differential access to and control over critical resources affect the structure and function of urban ecosystems. The overall study area was the Gwynns Falls watershed in Baltimore, MD (USA), a naturally defined urban-rural catchment area, combining the ecosystem/watershed approach of Bormann and Likens (1979) with social ecology theory and a community forestry perspective (e.g., Cernea, 1991). The degree of urbanization was defined in terms of land use and population density. The research differed from the usual studies of environmental equity in that it focused on the impacts of a diffuse set of drivers (e.g., private markets, government agencies, and local residents) over an entire watershed, rather than being restricted to a limited set of factors and environmental conditions usually related to point sources of pollution (e.g., Bullard, 1990).

Grove and Burch developed vegetative and socioeconomic indices for the city of Baltimore to explore the linkages between socio-cultural and biophysical patterns and processes on a spatially explicit basis. Characterization of patches within the watershed was achieved using a limited number of representative indicators. The socio-cultural indicators were obtained from the U.S. Bureau of the Census (at the block group level) and included socioeconomic (based on % professional/managerial workers, household earnings, and % with college degree), household (% married, % one-family households, % owner-occupied dwellings), and ethnicity (% non-white and foreign-born) indicators, which were used to derive a "social area index." Vegetation in the area was simultaneously described based on whether the ground surface was impervious or not (i.e., had vegetation cover) and whether it had a tree canopy or not. One of Grove and Burch's main findings was that after accounting for variations in population density, there is a positive relationship between the likelihood of a community to contain areas with trees and grass and its level of income and education. This study used a continuous definition of "urban" that was based on human population density, socio-cultural variables such as household income and education level, and the presence of anthropogenic landscape alterations (such as the presence of pavement). Without this multifaceted definition, Grove and Burch would have been unable to uncover the linkage between the structure of people's surroundings and their socioeconomic status.

Urban Water Balance for a Portion of Vancouver, British Columbia (Canada)

The hydrology of urban ecosystems differs significantly from many other ecosystems in that there is a piped water supply, organized water disposal, and changes in infiltration due to the increased coverage of impervious surfaces and water addition due to irrigation. Grimmond and Oke (1986) investigated the relationships among various elements of the hydrological cycle and urban conditions. Daily, monthly and annual water balance components for a catchment in Vancouver, BC (Canada), were compared to those from a rural area to elucidate the effects of urban development. The study site was the Oakridge catchment, a residential area in Vancouver. Observations in the catchment included daily totals of water used by the households, precipitation amounts, weekly soil moisture measurements, and estimates of soil water storage capacity. An overall water balance (i.e., inputs versus outputs versus changes in storage) was calculated for an entire year, as well as changes in these terms on a seasonal basis. The system consisted of two main components, the internal system (inputs of piped water to homes and output of waste water via sewers) and the external system (the "people-modified" hydrologic system, which included modifications of surface cover such as removal of natural vegetation, paving over of soil, artificial drainage networks, garden irrigation, swimming pools, street cleaning, storage ponds, reservoirs, and changes in precipitation and evapotranspiration brought about by the effects of urban modification on climate).

The results revealed the importance of understanding the roles of human behavior and perceptions in the overall water balance of the catchment. In the colder half of the year, total human water use was constant and variations in the water balance could be attributed to basic hydrologic parameters. During the summer months, however, the pattern of water use was much more variable and was related to prevailing weather conditions, the role of irrigation in yards, and a feedback mechanism between these two factors. This variability was primarily due to the residents perceiving a need to supplement the precipitation during periods of increased temperatures and/or solar radiation. There was a strong statistical relationship between human water use and weather—66 to 72% of water use during this period was explained by air temperature alone; the addition of the number of days since precipitation, soil moisture, and net radiation improved this to 85%. The authors concluded that the processes involved were more than a physical cause-and-effect system; rather, the system involved human decision-making and action (to water or not to water the yard) indirectly linked to weather patterns via a perception and assessment of the need for water. Interpretation of the results revealed the role of irrigation (especially yard watering), prevailing weather conditions (particularly evapotranspiration) and the complex feedback system between human behavior and biophysical controls. This urban study thus linked human population and behavior to resource use.

Concluding Remarks

Most of the ecological papers we reviewed were simply traditional plant or animal ecological studies conducted in urban settings, with humans considered to be agents of disturbance. However, urban ecology implicitly recognizes the role that humans play in developing unique ecosystems (Parlange, 1998), because urbanization is both an ecological and a social phenomenon. Therefore, integrating both social and natural sciences in the study of urban ecosystems is crucial, making urban ecology a truly interdisciplinary field (Walbridge, 1997). One difficulty in this integration lies in the fact that the term "urban" is used differently by social and natural scientists. Social scientists use the term to refer to areas with high human population density, whereas ecologists use the term more broadly to refer to areas under human influence.

As research into urban ecosystems expands, the need for an unambiguous, quantitative definition of urban becomes more apparent. Such a definition poses a problem, however. Although the development of a predictive (*a priori*) way of defining urban environments may be more desirable than the current descriptive (*a posteriori*) mode, it is probably not feasible for two reasons. Foremost of these reasons is the fact that the description of urban depends on the research question being asked. A certain urban variable may exist at an inappropriate scale for some questions. For example, a variable such as average building height may exist at too coarse a scale to be useful in answering a question like "how do the population dynamics of soil nematodes differ in areas with paved roads versus unpaved roads?" even though it may be quite appropriate for a question such as "do monarch butterflies avoid areas with tall buildings during migration?". Secondly, the boundary between the urban environment and the surrounding landscape is often not a clear one, but rather a gradient from an increasingly disturbed natural or agricultural system to increasingly dense suburban to urban core.

Recognizing these difficulties, we recommend that at least a working definition of the "urban environment" be included in each study, explicitly including baseline information on demography, physical geography, socioeconomic, and cultural factors that can potentially explain existing urban structure and predict trajectories of urban growth. A description of the urban environment should be as quantified as much as possible to facilitate comparisons among studies and areas, as would be needed for repeating a study in a different location or at a different time. We have suggested an interdisciplinary approach whereby ecologists borrow heavily from social sciences to construct a definition of their research setting. Likewise, social scientists would benefit from this same sort of consistency in defining their study systems.

Despite the clear need for the focus of ecological research to be turned towards human-dominated systems and particularly urban ecosystems (McDonnell and Pickett, 1990), there is relatively little research activity in this area. There is thus an unprecedented opportunity for research in urban ecosystems, especially considering that urbanization is increasing in both scope and magnitude (Botkin and Beveridge, 1997). Without a more quantitative definition of "urban," however, progress in urban ecology will be made slowly.

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Appendix 1. Ecological Research Papers that were Reviewed

Blair, 1996
Blair and Launer, 1997
Blanco and Velasco, 1996
Botkin and Beveridge, 1997
Cicero 1996
Clark and Samways, 1997
Czechowski, 1982
Davis, 1978
DeGraaf and Wentworth, 1981
Dreistadt *et al.*, 1990
Dulisz and Nowakowski, 1996
Edgar and Kershaw, 1994
Ehler and Frankie, 1979a,b
Emlen, 1974
Erskine, 1992
Erz, 1964
Faeth and Kane, 1978
Frankie and Ehler, 1978
Friesen *et al.*, 1995
Goszczynski *et al.*, 1993
Guthrie, 1974
Hadidian *et al.*, 1997

Hohtola, 1978
Hooper *et al*., 1975
Huhtalo and Jarvinen, 1977
Jim, 1998
Jokimaki and Suhonen, 1993
Jones and Clark, 1987
Kemp and Spotila, 1997
Klausnitzer and Richter, 1983
Kozlov, 1996
Lancaster and Rees, 1979
Majzlan and Holecova, 1993
McDonnell and Pickett, 1990
McGeoch and Chown, 1997
McIntyre, 1999
Medley *et al*., 1995
Mirabella *et al*., 1996
Natuhara and Imai, 1996
Natuhara *et al*., 1994
Nowakowski, 1986
Nuorteva, 1971
Pouyat *et al*., 1994
Rebele, 1994
Rosenberg *et al*., 1987
Ruszczyk and de Araujo, 1992
Ruszczyk *et al*., 1987
Sewell and Catterall, 1998
Speight *et al*., 1998
Sustek, 1987, 1992, 1993
Tischler, 1973
Trojan, 1981
Vincent and Frankie, 1985
Walcott, 1974
Wear and Bolstad, 1998
Weber, 1975
Woolfenden and Rohwer, 1969a,b
Zapparoli, 1997

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Section II
Conceptual Foundations of Urban Ecology

In this section, we seek to display the main roots of American and European Urban Ecology, as well as some of the most recent thinking springing at least indirectly from those roots. In the US "Urban Ecology", and some of its concepts, was introduced by a collection of biologists and social scientists at the University of Chicago in 1925, in the seminal document, "The City". Burgess' chapter "The Growth of the City" from that document is presented here because it is an early and influential work, strictly focusing on social processes (aspects) such as social organization of mobility. The European roots of Urban Ecology are reviewed by Sukopp (2002). In contrast to the anthropocentric American view, the European school had a clear emphasis on interactions among the non-human biotic and abiotic components. Burgess and Sukopp thus introduce the history of Urban Ecology of the New and the Old Worlds. They set a foundation for readers to embrace modern Urban Ecology by reading the much more recent works by Pickett et al. (2001), Grimm et al. (2000), and Alberti et al. (2003). These three papers discuss the interdisciplinary synthesis that is modern Urban Ecology. The choice was difficult – several other similar papers (e.g., Collins et al. 2000; Bradshaw 2003) could have been included, but would largely serve to reiterate the historical and conceptual themes introduced and reviewed by these five papers.

Several major themes can be seen within *Conceptual Foundations*.

First, Urban Ecology has gained in importance in the last few decades because urbanization is clearly recognized as the dominant demographic trend worldwide. We are now finally coming to grips with its full impacts on human society, global energy flow, nutrient cycling, and land cover transformation.

Second, Urban Ecology has evolved quite dramatically. Urban Ecology began as a discipline that sought to understand cities as "super-organisms". Burgess (1925) shows the extent to which this "Chicago School" saw cities as organisms or communities that could be characterized by their growth, metabolism, succession, and mobility. In this early stage Urban Ecology clearly focused on the human city dwellers. The field then changed largely to a more reductionist, natural science approach in which scientists (including Sukopp and his colleagues) worked within the city to understand the ecological details of the biology of urban plants and animals (e.g., Sukopp & Wittig 1998). Here human beings are mostly excluded and their activities are seen as one driving force determining the urban environment. Today, Urban Ecology continues to evolve, and increasingly looks at large landscapes that include both urban core areas and the surrounding gradients of urbanization, seeking to determine how urban ecosystems function. Modern Urban Ecology synthesizes the early American and European concepts by including both human and non-human city dwellers as driving forces of the system.

In the papers by Sukopp, and Pickett et al., we see characterizations of the physical and ecological aspects of urban environments. Much of this sort of ecology, which Pickett et al. (2001) and Grimm et al. (2000) call "ecology *in* the city," was pioneered by Sukopp and other Europeans. They watched the establishment and growth of new urban ecosystems after the devastation of World War II, and, in some cases they were simply unable to easily get beyond the city because of unfortunate political realities. For Sukopp, this reality was the Berlin Wall, which kept him focused on ecology within Berlin.

A transition to an *urban ecology of the city* is apparent in Pickett et al. (2001), Grimm et al. (2000), and Alberti et al. (2003). These authors develop conceptual models that link human actions with ecological function at scales that include landscapes and global ecosystems. They argue that understanding the ecology of organisms in the city, often at the population or community level, is necessary, but by no means sufficient to an understanding of the ecology of cities. That understanding requires a larger view, because of how thoroughly cities affect energy and material flows far beyond their political and geographic boundaries. New concepts about the way ecosystems work are integrated into these latest conceptual models - all of them incorporate the modern understanding that ecological systems are not in fact exhibiting one stable equilibrium. There may be several stable states, like in shallow lakes, that are highly dynamic, responding to a variety of disturbances and internal patch dynamics, and in them hierarchy and scale affect ecological patterns and processes.

Third, Urban Ecology, while interdisciplinary from the start, has increasingly sought to fully integrate our understanding of ecological systems with those of the embedded human systems. All modern conceptual models emphasize the interdependence of humans and ecological processes. The urban ecosystem is clearly a product of natural and social processes. The integrated understanding is being sought, today, in several significantly different ways. Pickett et al.'s (2001) "Human Ecosystem Framework" generally links ecosystems, human resources, and human social systems. Grimm et al. (2000) and Alberti et al. (2003) expand this into greater detail by explicitly linking drivers, patterns, processes, and impacts in urban ecosystems. However, Grimm et al. (2000) emphasize a cycle centered on land use and ecological patterns and processes, both of which affect both ecological and human systems - while Alberti et al. (2003) do not segregate human and ecological aspects of urban ecosystems. Alberti et al. (2003) seem to be the first to explicitly put human and non-human aspects of urban ecosystems into the same set of conceptual compartments, and their drivers, patterns, processes, and impacts are illustrated by both "human" and more "traditional ecological" examples.

Fourth, to understand urban ecosystems absolutely requires a blending of natural and social science. The drivers, patterns, processes, and impacts that are core constructs in modern conceptual models cannot be fully understood without the simultaneous use of the tools of the biologist, geographer, demographer, economist, political scientist, psychologist, urban planner, and others.

Fifth, Urban Ecology continues to evolve. Modeling, development of theory, and empirical studies will all make important individual, piecemeal contributions. But to succeed as a science that improves the human condition, educational approaches to Urban Ecology will need to evolve quickly. This is made evident by Alberti et al. (2003), who suggest that future urban ecologists will need not multidisciplinary but rather "TRANS-disciplinary" training. Transdisciplinarity requires a genuine merging of natural and social sciences to apply their integrated combined knowledge to real world issues.

Sixth, there is a need for academic-practitioner dialog with active, rapid feedback between academic researchers and the world of actual practitioners. Urban Ecology is actually practiced daily, around the world, on large spatial scales, with immense long-reaching consequences. It is practiced largely by urban planners, landscape architects, zoning commissions, and other governmental and regulatory bodies who must make decisions in the here-and-now. Their needs cannot wait for academic consensus, and there is a striking lack of two-way communication as to what information is really needed from academia by practitioners. Urban ecological education, properly designed, can clearly provide a transdisciplinary experience as planners and architects work with biologists and social scientists to understand the ecology of the city.

These realizations lead to a broad modern definition of urban ecology:

Urban Ecology is an integrative sub-discipline of ecology. It focuses on urban-dominated ecosystems: these include cities, suburbs, exurbs, villages connected to cities by transportation or utilities, and hinterlands managed or affected by the energy and material from the urban core and suburbs.

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The Growth of the City: An Introduction to a Research Project

Ernest W. Burgess

Abstract The aggregation of urban population has been described by Bücher and Weber. A sociological study of the growth of the city, however, is concerned with the definition and description of processes, as those of (*a*) expansion, (*b*) metabolism, and (*c*) mobility. The typical tendency of urban growth is the expansion radially from its central business district by a series of concentric circles, as (*a*) the central business district, (*b*) a zone of deterioration, (*c*) a zone of workingmen's homes, (*d*) a residential area, and (*e*) a commuters' zone. Urban growth may be even more fundamentally stated as the resultant of processes of organization and disorganization, like the anabolic and katabolic processes of metabolism in the human body. The distribution of population into the natural areas of the city, the division of labor, the differentiation into social and cultural groupings, represent the normal manifestations of urban metabolism, as statistics of disease, crime, disorder, vice, insanity, and suicide are rough indexes of its abnormal expression. The state of metabolism of the city may, it is suggested, be measured by mobility, defined as a change of movement in response to a new stimulus or situation. Areas in the city of the greatest mobility are found to be also regions of juvenile delinquency, boys' gangs, crime, poverty, wife desertion, divorce, abandoned infants, etc. Suggested indexes of mobility are statistics of changes of movement and increase of contacts of city population, as in the increase per capita in the total annual rides on surface and elevated lines, number of automobiles, letters received, telephones, and land values. A cross-section of the city has been selected for the intensive study of urban growth in terms of expansion, metabolism, and mobility.

Keywords: Urban growth · succession · expansion · social organization · city metabolism · mobility

The outstanding fact of modern society is the growth of great cities. Nowhere else have the enormous changes which the machine industry has made in our social life registered themselves with such obviousness as in the cities. In the United States the transition from a rural to an urban civilization, though beginning later than in Europe, has taken place, if not more rapidly and completely, at any rate more logically in its most characteristic forms.

All the manifestations of modern life which are peculiarly urban—the skyscraper, the subway, the department store, the daily newspaper, and social work—are characteristically American. The more subtle changes in our social life, which in their cruder manifestations are termed "social problems," problems that alarm and bewilder us, as divorce, delinquency, and social unrest, are to be found in their most acute forms in our largest American cities. The profound and "subversive" forces which have wrought these changes are measured in the physical growth and expansion of cities. That is the significance of the comparative statistics of Weber, Bücher, and other students.

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J.M. Marzluff et al., *Urban Ecology,*

These statistical studies, although dealing mainly with the effects of urban growth, brought out into clear relief certain distinctive characteristics of urban as compared with rural populations. The larger proportion of women to men in the cities than in the open country, the greater percentage of youth and middle-aged, the higher ratio of the foreign-born, the increased heterogeneity of occupation, increase with the growth of the city, and profoundly alter its social structure. These variations in the composition of population are indicative of all the changes going on in the social organization of the community. In fact, these changes are a part of the growth of the city, and suggest the nature of the processes of growth.

The only aspect of growth adequately described by Bücher and Weber was the rather obvious process of the *aggregation* of urban population. Almost as overt a process, that of *expansion*, has been investigated from a different and very practical point of view by groups interested in city planning, zoning, and regional surveys. Even more significant than the increasing density of urban population is its correlative tendency to overflow, and so to extend over wider areas, and to incorporate these areas into a larger communal life. This paper therefore will treat first of the expansion of the city, and then of the less known processes of urban metabolism and mobility, which are closely related to expansion.

Expansion as Physical Growth

The expansion of the city from the standpoint of the city plan, zoning, and regional surveys is thought of almost wholly in terms of its physical growth. Traction studies have dealt with the development of transportation in its relation to the distribution of population throughout the city. The surveys made by the Bell Telephone Company and other public utilities have attempted to forecast the direction and the rate of growth of the city in order to anticipate the future demands for the extension of their services. In the city plan the location of parks and boulevards, the widening of traffic streets, the provision for a civic center, are all in the interest of the future control of the physical development of the city.

This expansion in area of our largest cities is now being brought forcibly to our attention by the Plan for the Study of New York and Its Environs, and by the formation of the Chicago Regional Planning Association, which extends the metropolitan district of the city to a radius of 50 miles, embracing 4,000 square miles of territory. Both are attempting to measure expansion in order to deal with the changes that accompany city growth. In England, where more than one-half of the inhabitants live in cities having a population of 100,000 and over, the lively appreciation of the bearing of urban expansion on social organization is thus expressed by C. B. Fawcett:

> One of the most important and striking developments in the growth of the urban populations of the more advanced peoples of the world during the last few decades has been the appearance of a number of vast urban aggregates, or conurbations, far larger and more numerous than the great cities of any preceding age. These have usually been formed by the simultaneous expansion of a number of neighboring towns, which have grown out towards each other until they have reached a practical coalescence in one continuous urban area. Each such conurbation still has within it many nuclei of denser town growth, most of which represent the central areas of the various towns from which it has grown, and these nuclear patches are connected by the less densely urbanized areas which began as suburbs of these towns. The latter are still usually rather less continuously occupied by buildings, and often have many open spaces.
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> These great aggregates of town dwellers are a new feature in the distribution of man over the earth. At the present day there are from thirty to forty of them, each containing more than a million people, whereas only a hundred years ago there were, outside the great centers of population on the waterways of China, not more than two or three. Such aggregations of people are phenomena of great geographical and social importance; they give rise to new problems in the organization of the life and well-being of their inhabitants and in their varied

activities. Few of them have yet developed a social consciousness at all proportionate to their magnitude, or fully realized themselves as definite groupings of people with many common interests, emotions and thoughts.[1]

In Europe and America the tendency of the great city to expand has been recognized in the term "the metropolitan area of the city," which far overruns its political limits, and in the case of New York and Chicago, even state lines. The metropolitan area may be taken to include urban territory that is physically contiguous, but it is coming to be defined by that facility of transportation that enables a business man to live in a suburb of Chicago and to work in the loop, and his wife to shop at Marshall Field's and attend grand opera in the Auditorium.

Expansion as a Process

No study of expansion as a process has yet been made, although the materials for such a study and intimations of different aspects of the process are contained in city planning, zoning, and regional surveys. The typical processes of the expansion of the city can best be illustrated, perhaps, by a series of concentric circles, which may be numbered to designate both the successive zones of urban extension and the types of areas differentiated in the process of expansion.

This chart represents an ideal construction of the tendencies of any town or city to expand radially from its central business district—on the map "The Loop" (I). Encircling the downtown area there is normally an area in transition, which is being invaded by business and light manufacture (II). A third area (III) is inhabited by the workers in industries who have escaped from the area of deterioration (II) but who desire to live within easy access of their work. Beyond this zone is the "residential area" (IV) of high-class apartment buildings or of exclusive "restricted" districts of single family dwellings. Still farther, out beyond the city limits, is the commuters' zone—suburban areas, or satellite cities—within a thirty- to sixty-minute ride of the central business district.

This chart brings out clearly the main fact of expansion, namely, the tendency of each inner zone to extend its area by the invasion of the next outer zone. This aspect of expansion may be called *succession*, a process which has been studied in detail in plant ecology. If this chart is applied to Chicago, all four of these zones were in its early history included in the circumference of the inner zone, the present business district. The present boundaries of the area of deterioration were not many years ago those of the zone now inhabited by independent wage-earners, and within the memories of thousands of Chicagoans contained the residences of the "best families." It hardly needs to be added that neither Chicago nor any other city fits perfectly into this ideal scheme. Complications are introduced by the lake front, the Chicago River, railroad lines, historical factors in the location of industry, the relative degree of the resistance of communities to invasion, etc.

Besides extension and succession, the general process of expansion in urban growth involves the antagonistic and yet complementary processes of concentration and decentralization. In all cities there is the natural tendency for local and outside transportation to converge in the central business district. In the downtown section of every large city we expect to find the department stores, the skyscraper office buildings, the railroad stations, the great hotels, the theaters, the art museum, and the city hall. Quite naturally, almost inevitably, the economic, cultural, and political life centers here. The relation of centralization to the other processes of city life may be roughly gauged by the fact that over half a million people daily enter and leave Chicago's "loop." More recently sub-business centers have grown up in outlying zones. These "satellite loops" do not, it seems, represent the "hoped for" revival of the neighborhood, but rather a telescoping of several local communities into

[1] "British Conurbations in 1921," *Sociological Review*, XIV (April, 1922), 111–12.

a larger economic unity. The Chicago of yesterday, an agglomeration of country towns and immigrant colonies, is undergoing a process of reorganization into a centralized decentralized system of local communities coalescing into sub-business areas visibly or invisibly dominated by the central business district. The actual processes of what may be called centralized decentralization are now being studied in the development of the chain store, which is only one illustration of the change in the basis of the urban organization.[2]

Expansion, as we have seen, deals with the physical growth of the city, and with the extension of the technical services that have made city life not only livable, but comfortable, even luxurious. Certain of these basic necessities of urban life are possible only through a tremendous development of communal existence. Three millions of people in Chicago are dependent upon one unified water system, one giant gas company, and one huge electric light plant. Yet, like most of the other aspects of our communal urban life, this economic co-operation is an example of co-operation without a shred of what the "spirit of co-operation" is commonly thought to signify. The great public utilities are a part of the mechanization of life in great cities, and have little or no other meaning for social organization.

Yet the processes of expansion, and especially the rate of expansion, may be studied not only in the physical growth and business development, but also in the consequent changes in the social organization and in personality types. How far is the growth of the city, in its physical and technical aspects, matched by a natural but adequate readjustment in the social organization? What, for a city, is a normal rate of expansion, a rate of expansion with which controlled changes in the social organization might successfully keep pace?

Social Organization and Disorganization as Processes of Metabolism

These questions may best be answered, perhaps, by thinking of urban growth as a resultant of organization and disorganization analogous to the anabolic and katabolic processes of metabolism in the body. In what way are individuals incorporated into the life of a city? By what process does a person become an organic part of his society? The natural process of acquiring culture is by birth. A person is born into a family already adjusted to a social environment—in this case the modern city. The natural rate of increase of population most favorable for assimilation may then be taken as the excess of the birth-rate over the death-rate, but is this the normal rate of city growth? Certainly, modern cities have increased and are increasing in population at a far higher rate. However, the natural rate of growth may be used to measure the disturbances of metabolism caused by any excessive increase, as those which followed the great influx of southern negroes into northern cities since the war. In a similar way all cities show deviations in composition by age and sex from a standard population such as that of Sweden, unaffected in recent years by any great emigration or immigration. Here again, marked variations, as any great excess of males over females, or of females over males, or in the proportion of children, or of grown men or women, are symptomatic of abnormalities in social metabolism.

Normally the processes of disorganization and organization may be thought of as in reciprocal relationship to each other, and as co-operating in a moving equilibrium of social order toward an end vaguely or definitely regarded as progressive. So far as disorganization points, to reorganization and makes for more efficient adjustment, disorganization must be conceived not as pathological, but as normal. Disorganization as preliminary to reorganization of attitudes and conduct is almost invariably the lot of the newcomer to the city, and the discarding of the habitual, and often of what has been to him the moral, is not infrequently accompanied by sharp mental conflict and sense of

[2] See E. H. Shideler, *The Retail Business Organization as an Index of Community Organization* (in preparation).

personal loss. Oftener, perhaps, the change gives sooner or later a feeling of emancipation and an urge toward new goals.

In the expansion of the city a process of distribution takes place which sifts and sorts and relocates individuals and groups by residence and occupation. The resulting differentiation of the cosmopolitan American city into areas is typically all from one pattern, with only interesting minor modifications. Within the central business district or on an adjoining street is the "Main Stem" of "Hobohemia," the teeming Rialto of the homeless migratory man of the Middle West.[3] In the zone of deterioration encircling the central business section are always to be found the so-called "slums" and "bad lands," with their submerged regions of poverty, degradation, and disease, and their underworlds of crime and vice. Within a deteriorating area are rooming-house districts, the purgatory of "lost souls." Near by is the Latin Quarter, where creative and rebellious spirits resort. The slums are also crowded to overflowing with immigrant colonies—the Ghetto, Little Sicily, Greektown, Chinatown—fascinatingly combining Old World heritages and American adaptations. Wedging out from here is the Black Belt, with its free and disorderly life. The area of deterioration, while essentially one of decay, of stationary or declining population, is also one of regeneration, as witness the mission, the settlement, the artists' colony, radical centers—all obsessed with the vision of a new and better world.

The next zone is also inhabited predominatingly by factory and shopworkers, but skilled and thrifty. This is an area of second immigrant settlement, generally of the second generation. It is the region of escape from the slum, the "Deutschland" of the aspiring Ghetto family. For "Deutschland" (literally Germany) is the name given, half in envy, half in derision, to that region beyond the Ghetto where his successful neighbors appear to be imitating German Jewish standards of living. But the inhabitant of this area in turn looks to the "Promised Land" beyond, to its residential hotels, its apartment-house region, its "satellite loops," and its "bright light" areas.

This differentiation into natural economic and cultural groupings gives form and character to the city. For segregation offers the group, and thereby the individuals who compose the group, a place and a rôle in the total organization of city life. Segregation limits development in certain directions, but releases it in others. These areas tend to accentuate certain traits, to attract and develop their kind of individuals, and so to become further differentiated.

The division of labor in the city likewise illustrates disorganization, reorganization, and increasing differentiation. The immigrant from rural communities in Europe and America seldom brings with him economic skill of any great value in our industrial, commercial, or professional life. Yet interesting occupational selection has taken place by nationality, explainable more by racial temperament or circumstance than by Old World economic background, as Irish policemen, Greek ice-cream parlors, Chinese laundries, negro porters, Belgian janitors, etc.

The facts that in Chicago 1,000,000 (996,589) individuals gainfully employed reported 509 occupations, and that over 1,000 men and women in *Who's Who* gave 116 different vocations, give some notion of how in the city the minute differentiation of occupation "analyzes and sifts the population, separating and classifying the diverse elements."[4] These figures also afford some intimation of the complexity and complication of the modern industrial mechanism and the intricate segregation and isolation of divergent economic groups. Interrelated with this economic division of labor is a corresponding division into social classes and into cultural and recreational groups. From this multiplicity of groups, with their different patterns of life, the person finds his congenial social world, and, what is not feasible in the narrow confines of a village, may move and live in widely separated and, perchance, conflicting worlds. Personal disorganization may be but the failure to harmonize the canons of conduct of two divergent groups.

[3] For a study of this cultural area of city life see Nels Anderson, *The Hobo*, Chicago, 1923.

[4] Weber, *The Growth of Cities*, p. 442.

If the phenomena of expansion and metabolism indicate that a moderate degree of disorganization may and does facilitate social organization, they indicate as well that rapid urban expansion is accompanied by excessive increases in disease, crime, disorder, vice, insanity, and suicide, rough indexes of social disorganization. But what are the indexes of the causes rather than of the effects of the disordered social metabolism of the city? The excess of the actual over the natural increase of population has already been suggested as a criterion. The significance of this increase consists in the immigration into a metropolitan city like New York and Chicago of tens of thousands of persons annually. Their invasion of the city has the effect of a tidal wave inundating first the immigrant colonies, the ports of first entry, dislodging thousands of inhabitants who overflow into the next zone, and so on and on until the momentum of the wave has spent its force on the last urban zone. The whole effect is to speed up expansion, to speed up industry, to speed up the "junking" process in the area of deterioration (II). These internal movements of the population become the more significant for study. What movement is going on in the city, and how may this movement be measured? It is easier, of course, to classify movement within the city than to measure it. There is the movement from residence to residence, change of occupation, labor turnover, movement to and from work, movement for recreation and adventure. This leads to the question: What is the significant aspect of movement for the study of the changes in city life? The answer to this question leads directly to the important distinction between movement and mobility.

Mobility as the Pulse of the Community

Movement, per se, is not an evidence of change or of growth. In fact, movement may be a fixed and unchanging order of motion, designed to control a constant situation, as in routine movement. Movement that is significant for growth implies a change of movement in response to a new stimulus or situation. Change of movement of this type is called *mobility*. Movement of the nature of routine finds its typical expression in work. Change of movement, or mobility, is characteristically expressed in adventure. The great city, with its "bright lights," its emporiums of novelties and bargains, its palaces of amusement, its underworld of vice and crime, its risks of life and property from accident, robbery, and homicide, has become the region of the most intense degree of adventure and danger, excitement and thrill.

Mobility, it is evident, involves change, new experience, stimulation. Stimulation induces a response of the person to those objects in his environment which afford expression for his wishes. For the person, as for the physical organism, stimulation is essential to growth. Response to stimulation is wholesome so long as it is a correlated *integral* reaction of the entire personality. When the reaction is *segmental*, that is, detached from, and uncontrolled by, the organization of personality, it tends to become disorganizing or pathological. That is why stimulation for the sake of stimulation, as in the restless pursuit of pleasure, partakes of the nature of vice.

The mobility of city life, with its increase in the number and intensity of stimulations, tends inevitably to confuse and to demoralize the person. For an essential element in the mores and in personal morality is consistency, consistency of the type that is natural in the social control of the primary group. Where mobility is the greatest, and where in consequence primary controls break down completely, as in the zone of deterioration in the modern city, there develop areas of demoralization, of promiscuity, and of vice.

In our studies of the city it is found that areas of mobility are also the regions in which are found juvenile delinquency, boys' gangs, crime, poverty, wife desertion, divorce, abandoned infants, vice.

These concrete situations show why mobility is perhaps the best index of the state of metabolism of the city. Mobility may be thought of in more than a fanciful sense, as the "pulse of the community." Like the pulse of the human body, it is a process which reflects and is indicative of all the changes that are taking place in the community, and which is susceptible of analysis into elements which may be stated numerically.

The elements entering into mobility may be classified under two main heads: (1) the state of mutability of the person, and (2) the number and kind of contacts or stimulations in his environment. The mutability of city populations varies with sex and age composition, the degree of detachment of the person from the family and from other groups. All these factors may be expressed numerically. The new stimulations to which a population responds can be measured in terms of change of movement or of increasing contacts. Statistics on the movement of urban population may only measure routine, but an increase at a higher ratio than the increase of population measures mobility. In 1860 the horse-car lines of New York City carried about 50,000,000 passengers; in 1890 the trolleycars (and a few surviving horse-cars) transported about 500,000,000; in 1921, the elevated, subway, surface, and electric and steam suburban lines carried a total of more than 2,500,000,000 passengers.[5] In Chicago the total annual rides per capita on the surface and elevated lines were 164 in 1890, 215 in 1900, 320 in 1910, and 338 in 1921. In addition, the rides per capita on steam and electric suburban lines almost doubled between 1916 (23) and 1921 (41), and the increasing use of the automobile must not be overlooked.[6] For example, the number of automobiles in Illinois increased from 131,140 in 1915 to 833,920 in 1923.[7]

Mobility may be measured not only by these changes of movement, but also by increase of contacts. While the increase of population of Chicago in 1912–22 was less than 25 per cent (23.6 per cent), the increase of letters delivered to Chicagoans was double that (49.6 per cent) (from 693,084,196 to 1,038,007,854).[8] In 1912 New York had 8.8 telephones, in 1922, 16.9 per 100 inhabitants. Boston had, in 1912, 10.1 telephones, ten years later, 19.5 telephones per 100 inhabitants. In the same decade, the figures for Chicago increased from 12.3 to 21.6 per 100 population.[9] But increase of the use of the telephone is probably more significant than increase in the number of telephones. The number of telephone calls in Chicago increased from 606,131,928 in 1914, to 944,010,586 in 1922,[10] an increase of 55.7 per cent, while the population increased only 13.4 per cent.

Land values, since they reflect movement, afford one of the most sensitive indexes of mobility. The highest land values in Chicago are at the point of greatest mobility in the city, at the corner of State and Madison streets, in the loop. A traffic count showed that at the rush period 31,000 people an hour, or 210,000 men and women in sixteen and one-half hours passed the southwest corner. For over ten years land values in the loop have been stationary, but in the same time they have doubled, quadrupled, and even sextupled in the strategic corners of the "satellite loops,"[11] an accurate index of the changes which have occurred. Our investigations so far seem to indicate that variations in land values, especially where correlated with differences in rents, offer perhaps the best single measure of mobility, and so of all the changes taking place in the expansion and growth of the city.

In general outline, I have attempted to present the point of view and methods of investigation which the department of sociology is employing in its studies in the growth of the city, namely, to describe urban expansion in terms of extension, succession, and concentration; to determine how

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[5] Adapted from W. B. Monro, *Municipal Government and Administration*, II, 377.

[6] *Report of the Chicago Subway and Traction Commission*, p. 81, and the *Report on a Physical Plan for a Unified Transportation System*, p. 391.

[7] Data compiled by automobile industries.

[8] Statistics of mailing division, Chicago Post-office.

[9] Determined from *Census Estimates for Intercensual Years*.

[10] From statistics furnished by Mr. R. Johnson, traffic supervisor, Illinois Bell Telephone Company.

[11] From 1912–23, land values per front foot increased in Bridgeport from $600 to $1,250; in Division-Ashland-Milwaukee district from $2,000 to $4,500; in Back of the Yards from $1,000 to $3,000; in Englewood from $2,500 to $8,000; in Wilson Avenue from $1,000 to $6,000; but decreased in the Loop from $20,000 to $16,500.

expansion disturbs metabolism when disorganization is in excess of organization; and, finally, to define mobility and to propose it as a measure both of expansion and metabolism, susceptible to precise quantitative formulation, so that it may be regarded almost literally as the pulse of the community. In a way, this statement might serve as an introduction to any one of five or six research projects under way in the department.[12] The project, however, in which I am directly engaged, is an attempt to apply these methods of investigation to a cross-section of the city—to put this area, as it were, under the microscope, and so to study in more detail and with greater control and precision the processes which have been described here in the large. For this purpose the West Side Jewish community has been selected. This community includes the so-called "Ghetto," or area of first settlement, and Lawndale, the so-called "Deutschland," or area of second settlement. This area has certain obvious advantages for this study, from the standpoint of expansion, metabolism, and mobility. It exemplifies the tendency to expansion radially from the business center of the city. It is now relatively a homogeneous cultural group. Lawndale is itself an area in flux, with the tide of migrants still flowing in from the Ghetto and a constant egress to more desirable regions of the residential zone. In this area, too, it is also possible to study how the expected outcome of this high rate of mobility in social and personal disorganization is counteracted in large measure by the efficient communal organization of the Jewish community.

[12] Anderson, Nels, *The Slum: An Area of Deterioration in the Growth of the City*; Mowrer, Ernest R., *Family Disorganization in Chicago*; Reckless, Walter C., *The Natural History of Vice Areas in Chicago*; Shideler, E. H., *The Retail Business Organization as an Index of Business Organization*; Thrasher, F. M., *One Thousand Boys' Gangs in Chicago; a Study of Their Organization and Habitat*; Zorbaugh, H. W., *The Lower North Side; a Study in Community Organization*.

On the Early History of Urban Ecology in Europe

Herbert Sukopp

Abstract Early investigations on the ecology of cities were in the tradition of natural history and focused on single biotopes. Of special interest were the plants and animals introduced into new areas directly or indirectly by man. In Central Europe, studies of anthropogenic plant migrations and cultural history were combined in a specific way, the so called Thellungian paradigm. The succession of vegetation on ruins after the bombing during the Second World War was studied in many cities. Ecological studies on whole cities started in the 1970s with investigations on energy flow and nutrient cycling. Today the term urban ecology is used in two different ways: in developing programs for sustainable cities, and in investigation of living organisms in relation to their environment in towns and cities.

Keywords: urban flora and vegetation · human impact · ecological studies · Thellungian paradigm

Introduction

Urban ecology is the investigation of living organisms in relation to their environment in towns and cities, as in ecological studies of forests or the sea. The ecological approach considers a city as an ecosystem, characterized by its history, its structure and function, including both biotic and abiotic components, and the cycling and conversion of energy and materials. Cities also have their own spatial organization and distinctive patterns of change through time, which result in patterns of species behaviour, populations dynamics and the formation of communities, each of which is specific to the urban environment. In policy and planning the term urban ecology is synonymous with "sustainable cities".

Early investigations looked for particular biotopes, for the nature in a city, not for the nature of a city as a whole. The initial studies were on castles and ruins or gardens and parks. Investigations on the peculiarities of urban flora and vegetation revealed a high species diversity and a dynamic development of vegetation. Ecosystem studies started in the 1970's.

The remarks presented here are mainly about studies on the urban flora and vegetation in Central and Western Europe.

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The innovative work of Slavomil Hejný inspired many urban ecologists. I use this opportunity to express my personal appreciation of his work.

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Wild Flora of Castles and Ruins

In all cities "ruderal" biocoenoses exist. "Ruderata" (from Lat. rudus: rubble, ruins) is the name for a specific habitat (Buxbaum 1721, Linnaeus 1751). The definition connects ruderal plants with their habitat: they grow in places strongly disturbed by man, but not cultivated, e.g., on rubble. Classical "ruderata" are ruins and waste lands, walls and pavements. Plant remains from Roman "castra" and their surroundings have been studied; these localities often prove to be the nuclei of modern cities (Knörzer 1970). Studies include Celtic "oppida" (e.g. Küster 1992), Slavic cities (Opravil 1969, 1987, 1990, Wasylikowa 1978, Trzcinska-Tacik & Wasylikowa 1982, Wasylikowa et al. 1991) and Viking settlements (Behre 1983).

Castles are an ecological model of the changes in the environment and organisms that occur in settlements. Due to favourable microclimatic conditions and ruderal soils, plants and animals from warmer regions are spreading. In Berlin 60% of all non native plants come from such regions: this is true of archaeophytes as well as neophytes (Scholz 1960). The differences between the "Burgbergflora" and surrounding flora and vegetation were studied by botanists in the 19th century (Chatin 1861, Kirschleger 1862, Krause 1896). Around castles the number of species of wild flowering plants can be twice that in similar areas in the vicinity (Lohmeyer 1984).

The flora of ruins and walls has been studied for centuries, e.g., the flora of the Colosseum of Rome (Panarolis 1643, Sebastiani 1815, Deakin 1855, Celesti Grapow et al. 2001), walls in Palestine (Hasselquist 1762, Weinstein & Karschon 1976, 1977) and in Algeria (Jourdan 1866, 1872), and the walls of the churches of Poitiers (Richard 1888); for subsequent studies see Brandes (1992). Observations on the development of vegetation on the ruins of the Conseil d'Etat in Paris, destroyed during the last days of the Commune uprising, were published by Flammarion (1881) and Vallot (1884). Plant growth on walls is enriched when stones 'alien' to the locality of the town and old types of mortar are used. Modern Portland cement is harder and more resistant to weathering (Segal 1969). Wall plants have interested botanists for centuries (Fitter 1945). Gerard (1597) in his "Herball" mentioned several plants on the walls in London: *Parietaria judaica* "groweth neere to old walls in the moist corners of churches and stone buildings" (Woodell 1979).

Cultivated Flora in Gardens and Parks

The existence of gardens worldwide is recorded in manuscripts, paintings and drawings from the time of Byzantium time to the present (e.g. Beuchert 1983, Willerding 1984). Ornamental plants, like lilies, roses and chrysanthemum have been cultivated for a long time. The first survey of the plants in German gardens was carried out by Conrad Gesner (in Cordus 1561). Introduced plants dominated over native ones. The history of introductions of ornamental plants is described by Kraus (1894), Wein (1914, 1943), Goeze (1916), Kowarik (1992) and Groeningen (1996). Plants introduced intentionally for horticultural use were planted first in monastery and peasant gardens and, nowadays in urban areas from which they have spread into the surrounding landscapes. The effect of escaped ornamentals on the flora around cities is documented by Kosmale (1981), Kunick (1991), Adolphi (1995) and Maurer (2002).

Botanical gardens are recorded from e.g. Pergamon, Athens, Byzantium and Toledo. The oldest European botanical gardens are those at Pisa 1543, Padua 1545 and Firenze 1545 (Chiarugi 1953), and north of the Alps those at Leiden 1577 and Leipzig 1580. Physic gardens were established to produce medicinal herbs, e.g. the Chelsea Physic Garden, which was established in London in 1673 by the Society of Apothecaries to grow and study the plants of their trade.

The well known drawings by Merian show gardens near cities. In Thünen's "Der isolierte Staat" (1826) is presented a theory of the relations between an ideal city and its surroundings in which intensively used gardens are situated close to the city.

Archaeological and palaeo-ethnobotanical studies have shown that fruit-trees were cultivated in early neolithic times. Under the Romans the number and performance of fruit-trees increased in Central and Western Europe. Vegetable cultivation was well developed in areas under Roman influence and improved during the Middle Ages. There is documentary evidence of the use of spice plants in the Roman era and the Middle Ages. Results of countrywide palaeo-ethnobotanical surveys given in Zeist et al. (1991) show differences in the species of cultivated plants in cities and villages.

In antiquity trees were planted around places of worship, during the Middle Ages in the squares in front of churches or town halls, and in Central Europe the trees were mainly *Tilia* species (Hennebo 1978). Since the 19th century trees have been an important element of urban planning (Stübben 1890). Around 1900 all big cities had trees planted along the streets.

Green roofs and roof gardens are known from ancient times (Babylon, Herculaneum) and many parts of the world (Japan: rice straw, Scandinavia with sod or turf). Roofs with self-established vegetation are described by Bornkamm (1961). The roof gardens in Berlin (Darius & Drepper 1985) became a well known feature of the city by the end of the 19th century and inspired architects.

During the medieval plague epidemics cemeteries were for the first time situated outside city walls. This was for hygienic reasons and because of the lack of space inside the city walls. Prior to 1900 publications on the flora of cemeteries only list ornamental plants (Unger 1870, Murr 1901) and spontaneously growing plants were only recorded over the last 25 years. A survey of the ecological investigations of cemeteries is given by Graf (1986).

Species Diversity in Urban Habitats

The first recorded botanical ramble in the London region was on Hampstead Heath (Johnson 1629). Together with another report (Johnson 1632) it is the first list (72 resp. 97 species recorded) for a particular area with heath, bog and ruderal flora (compare Fitter 1945). The flora of Paris was repeatedly studied in the 17th and 18th century (Cornut 1635, Tournefort 1698, Vaillant 1727). Many famous botanists collected plants there (Jolinon 1997): among others, four members of the family Jussieu, Linnaeus, Rousseau, Buffon, Willdenow, Kunth, Bonpland and A. v. Humboldt.

The first printed urban floras were not restricted to a particular area "intra muros", but included areas distant from the centre, e.g. Jungermann's flora of Altdorf and Gieÿen (Jungermann 1615, 1623), Ray's flora of Cambridge (Ray 1660). Willdenow (1787) in his "Florae Berolinensis Prodromus" mentioned only 8 species "in urbe ipsa" (among a total of 822 species). Later there were publications with titles like "Intramuralornis" (Paquet 1874, Schalow 1877) and "Intramuralflora" (Vallot 1884). The number of publications on urban ecology – even on well known groups like birds and mammals – increased only slowly during the 19th century, but accelerated after 1900 and grew rapidly after 1950 (Jolinon 1997 for Paris).

Early publications refer to migration of plants (Anom. 1782; see Brandes 2001), which later became introduction and naturalization of non native plants or invasions. In early studies of plant geography (Willdenow 1792, Chapter "Geschichte der Gewächse" in his "Grundriÿ der Kräuterkunde") there are no references to urban conditions. It was Schouw (1823) in the first textbook on plant geography who introduced the term "plantae urbanae" for plants living near cities and villages, e.g. *Onopordum acanthium, Xanthium strumarium*. He added: "In most cases foreign origin is the cause why these plants are located only near cities and villages". In particular, he named the plants growing on walls, ruins, roofs and rubble, and weeds of gardens. Chamisso (1827) described the conditions and effect of man on the flora and fauna of settlements: "Wherever man settles, the face of nature is changed. His domesticated animals and plants follow him; the woods become sparse; and animals shy away; his plants and seeds spread themselves around his habitation; rats, mice and insects move in under his roof; many kinds of swallow, finch, lark and partridge seek his care and enjoy, as guests, the fruits of his labor. In his gardens and fields a number of plants grow as weeds among the crops he has planted. They mix freely with the crops and share their fate. And where he no longer claims the entire area his tenants estrange themselves from him and even the wild, where he has not set foot, changes its form."

This quotation is taken from an instructional work "Botany for the non botanists", which Chamisso wrote for the Culture Ministry. The subtitle of this book, written in the tradition of natural history, was "A survey of the most useful and harmful plants, whether wild or cultivated, which occur in Northern Germany, including views on botany and the plant kingdom". In scientific terms this is the introduction of non native species, changes in biotopes, synanthropy, hemerochory, apophyty and agriophyty. Introduction and naturalization of non native plants, the so called adventive plants, was first studied by Watson (1859) and de Candolle (1855).

The concepts and terminology of the Swiss botanist Thellung (1912, 1918/19) have influenced the Central European approach up to the present day. A number of similar earlier and contemporary attempts proved less successful (de Candolle 1855, Rikli 1903, Linkola 1916).

Thellung's achievement was not in the development of a new approach, but in producing a systematic summary of the basic concepts and methods, which were used, with some variations, in the "adventive floristics" of his time. He provided an exact definition of the terms according to prevailing usage (1918/19), in particular that of Rikli (1903). Thellung discussed and defined terms like native, introduced, aliens, casuals etc. in French, German and English and created a scientific (Greek) terminology. One is tempted to speak of a paradigm (Trepl 1990). However, it is descriptive and non theoretical, and not clearly defined. Scheuermann (1948) wrote that Thellung "opened up research in the field of alien plants". He combined natural science and cultural history in a specific paradigm, later criticized by natural scientists.

There are several summaries of the development of the concepts and terminology (Holub & Jirásek 1967, Kornas 1968, Schroeder 1969, Sukopp & Scholz 1997, Richardson et al. 2000), one of the aims of which was to standardize terminology. Thellung's terms are still used but all have been simplified (Schroeder 1969). A comparison of the Central European terminology with that used in studies on plant invasions is given by Pyšek (1995b).

The studies of the "Working Group on Synanthropic Plants" at the Botanical Institute of the Czechoslovak Academy of Sciences in Průhonice included plants on arable land, in particular so called "quarantine" weeds (Hejný et al. 1973, Jehlík 1998; see Pyšek 2001), as well as urban flora. Since Laus (1908) the vegetation of specific habitats has been studied, e.g. railway sites (Jehlík 1986), cemeteries (Pyšek 1988), road verges (Klimeš 1987) and factories (Pyšek & Pyšek 1988c). The distribution of plant communities in particular habitats was studied by Pyšek 1978 (Plzeň), Tlusták 1990 (Olomouc) and Pyšek & Rydlo (1984) in villages. The occurence of individual species in habitats was considered in detail in the papers by Pyšek & Pyšek (1988a for the city of Plzeň, 1988b for West Bohemian villages).

Historically, bombed sites have been important in the development of studies on urban flora. Less than 3 years after the blitz Salisbury (1943) described the plants that colonized the ruined houses in London. In many cities the war damage and its effects gave rise to studies of rubble flora and fauna (Erkamo 1943, Balke 1944, Lousley 1944, Scholz 1960, Pfeiffer 1960). Rubble offers warmer and drier conditions than natural habitats and is a suitable habitat for plants and animals from warmer regions of the world. Many plants that were previously rare became permanent members of the urban flora in war-damaged European cities.

Dispersal of organisms and vegetation development on rubble was studied extensively on rubble sites which differ in their environment from previously studied ruderal sites. As Pfeiffer (1957) wrote: "The recolonization of rubble, created in many cities due to the activity of bombers in the last war, has unintentionally become a tremendous natural experiment, which with respect to its size, must be compared to the colonization of new habitats created by volcanic activity."

A similar situation occurred many years earlier following the Great Fire of London in 1666, when Ray recorded a population explosion of *Sisymbrium irio*, which was subsequently named the London rocket. Soon it became established in towns over a wide area but is now extremely rare and one of the few urban species to be included in an early edition of the Red Data book for vascular plants (Perring & Farrell 1977); later it was removed because it is a non-native species (Wiggington 1999). After a century's absence it reappeared in London at the end of the second world war. Gilbert (1989) concluded that a number of today's common ruderals will show a similar decline in the future.

Plants spread during war – mainly with horse-fodder – were called polemochors (Gaudefroy & Mouillefarine 1871, Mannerkorpi 1944, Luther 1948). Thellung (1917) gave the name stratiobotany (polemobotany) to research on the development of new plant formations with characteristic flora and on changes in cultivation that result from war (see also Kupffer 1922, Pettersson 1944).

Studies on the dynamic character of vegetation on rubble resulted in the first peak in urban ecology studies. Every big city has its own naturalized plants (Gilbert 1992). Unlike floristic studies, ecological research focused on dispersal strategies of particular species, succession under various site conditions, and the formation of new plant communities. However, all these investigations were carried out on rubble, not on a city as a whole.

The City as a Whole

Present urban biota and communites can be seen as a result of historic development. In Central Europe the re-establishment of forest vegetation after the last ice-age was incompleted when man began to influence the vegetation by disturbances on a local scale. Large scale disturbance, however, began with clear-cutting of extensive areas for agriculture and 2000 years ago with the development of cities.

The environment of cities is very different from that prevailing outside their limits. Nevertheless, in many cities fields and gardens have been present for many centuries. Today, however, cities usually consist of a mixture of densely settled areas in the historic center, remnants of agro-ecosystems ("encapsulated countryside") and even near-natural areas in urban forests, parks, and nature reserves.

Open spaces in cities are modifications of older habitats. The similarity between former and present habitat conditions decreases with time and along a gradient from the periphery to the center. Historic periods, like the pre-industrial, industrial and post-industrial, created specific site conditions and favoured different plant and animal communities in urban areas. Currently, parts of cities of different ages have different plant species and communities (Saarisalo-Taubert 1963, Aey 1990).

Early investigations revealed that even man-made sites have characteristic combinations of organisms. More exact analysis revealed a considerable variety of sites, organisms and communities in London (Fitter 1946, Gill & Bonnett 1973), Paris (Jovet 1954), New York (Kieran 1959, Rublowsky 1967), Vienna (Kühnelt 1955, Schweiger 1962), Polish cities (Falinski 1971), Saarbrücken (Müller 1972), Brussels (Duvigneaud 1974), Berlin (Kunick 1974), Birmingham (Teagle 1978) etc. The Man and the Biosphere Project 11 of UNESCO resulted in the first attempts at complex studies, e.g. Hong Kong (Boyden et al. 1981) or Tokyo (Miyawaki et al. 1975, Numata 1977). These studies deal with questions of health, human welfare and the connection between culture and nature.

Cities were compared to organisms, with the parks and gardens as the "green lungs" (Francé 1920 for Munich, Peters 1954 for Stuttgart). Pfalz (1910) described wild and planted habitats. Ecosystem studies of cities started in the 1970's with investigations of energy flow and nutrient cycling. Duvigneaud (1974) applied the methods of forest and lake analysis used in the International Biological Program (IBP) to analyse a big city. Taking Bruxelles as an example, the input and output of matter and energy were calculated, treating the city like a black-box. The inner differentiation of a city was looked along the lines of the Berlin model (Kunick 1974): densely built up zone, partly built up central areas, inner suburbs, outer suburbs.

Ecosystem studies used the methods of ecological assessments and “Produktlinienanalysen” (Newcombe et al. 1978, Maier et al. 1996, Baccini & Baader 1997, Simon & Fritsche 1998). Biogeochemical budgets, ecological footprints (Rees 1996) and summaries of citywide species richness are characteristic of this approach.

Flora and Fauna

Investigations of the flora of whole cities, rather than single habitats, include lists of species and their systematic, biological, ecological and geographic analyses. The term flora first referred to garden plants. The “Flora Danica” (Pauli 1648) and the “Flora Marchica” (Elsholz 1663) were the first books to include native and cultivated plants (Wein 1932). The higher number of species in cities compared to the surrounding countryside was first pointed out by Walters (1970) and Haeupler (1974). As a quantitative assessment, frequency (percentage of squares in which a species is recorded) was used by Kunick (1974). Numbers of species in cities were summarized by Falinski (1971), Pyšek (1989, 1993), Brandes & Zacharias (1990), Klotz (1990). Complete lists of urban floras are available for the following Czech cities: Prague (Špryňar & Münzbergová 1998), Brno (Grüll 1979), Plzeň (Pyšek & Pyšek 1988a), and Most (Pyšek & Hejný 2003). For a summary of the methods to study the flora and vegetation in urban areas see Pyšek (1995a).

In cities for which the history of the flora is known, the changes and their causes have been studied (Table 1). The changes reflect the economic and cultural history. The development of the ruderal flora “essentially runs parallel to the size and the intensity of trade and industry; it is a direct standard of the technical culture” (Naegeli & Thellung 1905: 226). The spatial structure of urban floras can be mapped on a grid of regular spatial units; Jackowiak (1998) reviews the published maps and atlases of city floras. The first atlases of a city’s flora were for London (Burton 1983) and Duisburg

Table 1 Historical analysis of the development of urban floras

Author	City/region	Period
Strumpf 1969, 1992	Altenburg	1768, 1889–92, 1938, 1968
Scholz 1960	Berlin	1787, 1884, 1959
Brandes 1984, Hellwig 1990	Braunschweig	1650, 1830, 1876, 1990
Godefroid 2001	Brussels	1940–1971, 1991–1994
Zimmermann-Pawlowsky	Euskirchen	1910, 1983, 1985
Schwarz 1967	Gdansk	1825, 1866, 1941, 1965
Klotz 1984	Halle/S. a. Halle-Neustadt	1848, 1983
Linkola 1916	Karelia (56 settlements)	1600, 1850–1880, 1915
Klotz & Il’minskich 1988	Kazan	1900, 1983
Trzcinska-Tacik 1979	Krakow	1809, 1920, 1977
Gutte 1989	Leipzig	1830, 1846, 1867,
Gusev 1968	Leningrad	ca. 1760, 1960 (only ruderal flora)
Fijalkowski 1994	Lublin	1787, 1848, 1917, 1944, 1993
Thellung 1912	Montpellier	ruderal flora since 1570
Gödde 1982	Münster	1947, 1974, 1981
Michalak 1970	Opole	1882, 1904, 1946, 1965
Jackowiak 1990	Poznan	1850, 1896
Sudnik-Wojcikowska 1987	Warszawa	1824, 1914,
Griese 1999	Wolfsburg	1985, 1998
Hetzel & Ullmann 1981	Würzburg	1947, 1980
Landolt 1991, 1992, 2001	Zürich	Middle Ages, 1839, 1905, 1990

(Düll & Kutzelnigg 1992). Based on 6000 vegetation relevés made in West Berlin, the hemeroby concept (Jalas 1955) was used to express the response of each species of the urban flora to the complex measure of human influence (Kowarik 1990). Human impact is made up of many factors, some of which (stress, disturbance) cannot be directly measured (Sudnik-Wojcikowska 1988). That city floras have specific features was shown by Gutte (1969) for Leipzig, Chemnitz and Dresden, based on the distribution of thermophilous plants and communities.

Unger (1852) cites the ecological characteristics of ruderal plants: specifically those of the families of *Urticaceae, Amarantaceae, Polygonaceae, Solanaceae* etc., are growing near human settlements because of the high levels of nitrogen in the soil. Here the term ruderal is congruent with nitrophilous according to the definition of Warming (1902). "Fertilizer is known to replace heat" (Thellung 1914) is similar formulation for plants that need of heat and whose roots penetrate nutrient rich soils further than on nutrient poor soils in climatically unfavourable habitats (Hügin 1992).

The current distribution of plants in urban areas is determined by land use. Wittig et al. (1985) distinguished urbanophilous, urbanoneutral and urbanophobous species (see also Korsch 1999). The distribution of plants in urban areas of different ages has been investigated. Saarisalo-Taubert (1963) showed that the distribution of the "Begleitflora alter Siedlung" (flora accompanying old settlements) is determined by favourable edaphic and microclimatic conditions. They can be natives as well as introduced plants (Aey 1990). The spatial structure of the urban flora and fauna is subject to temporal changes due to dispersal (Sudnik-Wojcikowska 1987a, b) and retreat of species (Linkola 1933).

With the domestication of plants (Körber-Grohne 1987, Zohary & Hopf 1993, Hondelmann 2002) and animals (Benecke 1994) and the storage of food, some wild animals entered into a specific relationship with man (Povolny 1963, Kenward & Allison 1994) and many of them became cosmopolitan. For a long time, cities and cultural landscapes were seen as biologically impoverished. At first Hesse (1924) stated: "The garden and park landscape interspersing and surrounding cities and villages is rich in (animal) species according to the diverse plant world and the varied general character of this formation."

The intramural fauna dates from the beginning of cities in the Near East 7 to 10,000 years ago, and reached Central and Western Europe 1,000 to 1,300 years ago (Davis 1987, Reichstein 1987). Pests of stored foods occurred in ancient Egypt (2,900 BC) and in Europe since the 16th century (Stein 1986).

The literature on the animal life in cities is summarized by Klausnitzer (1993), Gilbert (1989), Erz & Klausnitzer (1998), Luniak & Pisarski (1994). The species richness of urban gardens is reported by Owen (1991).

Vegetation

The term "urban vegetation" includes all types of spontaneously occurring and cultivated vegetation in cities (Sukopp & Werner 1983). Recording of urban vegetation started in Berlin (Scholz 1956); a survey of the following investigations is given by Wittig (2002). Complete vegetation surveys of Czech cities are published for Brno (Grüll 1979), Plzeň (Pyšek 1978, Pyšek & Pyšek 1988c), Most (Pyšek & Hejný 2003), Sušice (Pyšek 1972), Chomutov (Pyšek 1975), Prague (Kopecký 1980–1984, 1986, 1990), Brno (Grüll 1981), Bechyně (Hadač 1982), Liberec (Višňák 1986), and Olomouc (Tlusták 1990). A survey of urban vegetation research in East-Central and East European countries is given by Mucina (1980). Pyšek & Pyšek (1991) demonstrated important differences between the ruderal vegetation of cities and villages. Numbers of plant communities in 39 European towns and 85 Czech villages were summarized by Pyšek (1993).

Although the considerable spatial and temporal variability in urban vegetation makes vegetation classification difficult, the deductive method is most promising (Kopecký & Hejný 1978).

Apart from vegetation mapping a semi-quantitative representation of vegetation is possible using the method of unit areas (Pyšek 1975, Pyšek & Pyšek 1987).

Phytosociological studies of ruderal vegetation need an ecological basis (Sukopp 1971, 1973, Kopecký 1980–1984). The effects of city soils and air, e.g. stress due to deicing salt, heavy metals, SO_2 and other harmful substances, have been studied (Antonovics et al. 1971, Bornkamm 1990, Darius 1996, Rebele 1996). Bornkamm et al. (1982) refer to international investigations and ecosystem studies on suburban forests (Faensen-Thiebes et al. 1991), which gave a new impetus to general ecology (Cornelius et al. 1999).

Urban flora and vegetation is poorly integrated into urban biocoenoses. They are non-equilibrium systems in which stochastic processes are more important than deterministic ones. Succession in urban biocoenoses, relative to that in non-urban ones, is subject to strong and extremely variable anthropogenic influences, which are strongly linked to site history. Hence, these communities do not experience directional succession but are dominated by chance and unpredictable events; there are no climax conditions. The initial species composition is important in their future development.

A major reason for the (relative) unpredictable nature of succession in urban ecosystems is the high frequency with which they are invaded by aliens species; the biogeographical spectrum of species in cities is very different from that of the surrounding countryside. The reason for this may be (a) the ease with which these biocoenoses are invaded, and (b) the favourable conditions for dispersal (introduction, transportation). Disturbances generally favour invaders and urban ecosystems are subject to disturbances. Towns are subject to invasion by alien species and the number is unpredictable (Trepl 1994).

Urban biocoenoses are an extreme example of communities produced by successive invasions and not by co-evolutionary development. In principle, the historic uniqueness of urban ecosystems, i.e. their combination of environmental factors and organisms, differentiate them from most natural ones, even those subject to strong disturbance.

Environmental Conditions

Abiotic factors are an important component of the ecology of plants and animals (Humboldt 1807). The earliest studies showed that urban climate differed between cities and the surrounding countryside (Howard 1833). The heat requirements of ruderal plants are summarized in Sudnik-Wojcikowska (1998) and Hügin (1999). Urban heat islands are an important determinant of distribution of urban flora in Central European cities, whereas in Southern European cities it is ancient walls and ruins (Celesti Grapow et al. 2001).

Emission research ("Rauchschadenforschung") started with studies on the relation between the sulphur content of coal and damage by SO_2 (Stöckhardt 1850). Measures taken to reduce the damage caused by emissions are reported in Strabon in 7 BC (Meineke 1969) and in the Corpus Iuris Civilis (533/534 AD). Grindon (1859) and Nylander (1866) were the first to recognize the correlation between increasing air pollution and the retreat of epiphytic lichens. Sernander (1926) subdivided cities into zones according to the presence of certain lichens indicating different degrees of air pollution. Bioclimatology began in 1929 with studies on radiation and cooling (Kuttler 1993). Kratzer (1937, 1956) established another branch of urban ecology with his Berlin thesis on the urban climate. Regular recording of the first flowering of cherries started in Japan as early as 812; in England phenological investigations started in 1736 (Margary 1926, Sparks & Corey 1995).

Specific characteristics of soils in urban and industrial areas rarely received attention, except that of soil near smelting plants (Senft 1857). Archaeologists used high phosphate levels in soils as an indicator of settlements of hunters, fishermen and cattle breeders (phosphate mapping after Arrhenius 1931). For a long time, substrates in cities were regarded by pedologists as heterogenous and too young to develop into soils. Pedological studies in cities were not started before the 1970s and include Perth (Andrews 1971), Berlin (Runge 1975, Grenzius & Blume 1983), Washington (Smith 1976), and Halle (Billwitz & Breuste 1980). Recommendations for soil mapping in cities were published by Blume et al. (1989) and Burghardt et al. (1997). The International Working Group "Soils of urban, industrial, traffic and mining areas" was founded in 1998 (Burghardt & Kneib 2001).

Most biological studies of urban areas involve human health and welfare, e.g. control of disease, plants for medicines, air and water pollution, disposal of solid waste, treatment of contaminated soil. Thurnwald (1904) analysed the effects of city climate and professional life on physiological and psychic behaviour. Classical synopses are the books of Hellpach (1939) and Rudder & Linke (1940).

Most cities are located near rivers, the waters of which are changed by the increased run-off following urbanization of the catchment area and the resultant increase in impervious surface cover, chanellization, pollution and decline in richness of biotic communities (Paul & Meyer 2001). Methods of supplying drinking water and treating waste water were devised early in the establishment of urban agglomerations. Kolkwitz & Marsson (1902) devised biological indication of water quality, based on plant and animal indicator species and their capability to exist under different saprobic conditions.

Kolkwitz (1909, (1914) was the first to successfully rehabilitate a lake. He investigated the mass growth of algae in the Lietzensee, a lake in Berlin, and came to the conclusion that the factor regulating algal production was the almost continual supply of nutrients from the mud at the bottom of the lake. He successfully reduced the supply of nutrients by flushing out the nutrient-rich mud from the bottom of the lake with nutrient-poor water.

Urban ecology developed methodologically out of landscape ecology by intensively studying settlements (Sukopp 1990, Sukopp & Wittig 1998), which are regarded as ideal landscapes for such studies (Leser 1991). In geography landscape ecology is the study of the economy of nature (Naturhaushalt), i.e. the ecological aspects of mainly cultural and harmonic landscape. In pre-industrial times this concept also applied to cities, the direct and immediate expression of the natural-ecological conditions of an area. Modern cities are characterized by their worldwide connections and are uncoupled (disassociated) from the local surroundings with the consequence that the concept of a cultural landscape is only partly relevant. In ecology, on the contrary, landscape ecology is the investigation of several adjacent ecosystems; this concept can be applied to cities. In urban geography (Petermann 1903, Hard 1985, Lichtenberger 1998) the term urban landscape was developed around 1920 for a type of cultural landscape characterized by settlements. A major contribution to landscape ecology was the book "Design with nature" (McHarg 1969).

In urban history, ecological aspects increasingly played a role influenced by regional studies (Hauptmeyer 1987). Archaeological and archival studies of modern city centres made ecological analysis possible. Palaeoecology is especially suitable for revealing lifestyle, land use and demography (for Germany e.g. Meckseper 1985, Herrmann 1989). Palaeo-ethnobotany was developed by Unger (1851, 1852) and Heer (1865, 1883) and is reviewed by Willerding (1987). Urban archaeology of city centres lead, together with early floras lists, to today's urban botany (Willerding 1986, Hellwig 1990, Landolt 1991, 2001). The cultivated plants and accompanying wild flora in gardens, fields and meadows in Pompei, which was destroyed in 79 AD, are documented in ikonographical, literal, archaeological, ethnobotanical and palynological records (Jashemski 1979). So another basis of urban ecology is urban archaeology.

First urban ecology syntheses were published by Weidner (1939), Rudder & Linke (1940) and Peters (1954), authors whose expertise was in biology, geography or medicine. Summaries are to be found in Gilbert (1989), Sukopp (1990), Wittig (1991, 2002), Klausnitzer (1993), Sukopp & Wittig (1998), Breuste et al. (1998), Friedrichs & Hollaender (1999), Kavtaradze & Fridman (2000) and Picket et al. (2001).

As a separate discipline urban ecology was established in the early 1970s with systematic studies of climate, soil, water and organisms. With growing interest in nature conservation in cities (e.g. Barker 1997) programmes focused on the mapping of biotopes appeared (Schulte et al. 1993).

Hejný (1971) distinguished 68 habitat types in Prague. A total of 223 cities in Germany (all cities and many medium-sized towns) and 2000 villages and small towns have been biotope mapped (Schulte & Sukopp 2000). Although widely applied mainly in Germany such studies are mostly lacking application in other countries.

The term urban ecology was introduced by the Chicago school of social ecology within sociology (Park et al. 1925). In ecology, the term was formally defined in the 1970s, whereas the content had existed for centuries. Internationally, the institutionalization of urban ecology came with UNESCO's intergovernmental Man and the Biosphere (MAB) Programme in 1971.

Comparison with Non European Approaches

In many countries urban ecology research is less concerned with nature conservation, but more with sociological investigations under the heading of "human ecology". These investigations are conducted by sociologists and psychologists. In Japan relatively natural environments have been created by constructing of "native" forests with native trees, integrating research on the potential natural vegetation and traditional Japanese methods of creating "chinju-no mori" (shrine and temple forests). The introduction of "environmental forests" into various places – such as factories and power stations on reclaimed coastal land, schools, parks, streets, airports and harbours – has been increasingly successful. Restoration of natural environments by creating native forests has been carried out at such sites. By combining the traditional method of planting native trees in towns and villages with modern phytosociological and ecological diagnosis (map of present-day vegetation, report on the site conditions), and prediction (map of potential natural vegetation and habitats), new forests have been created in more than 120 locations throughout Japan (Miyawaki et al. 1987).

In Third World countries, more attention is given to agriculture or forestry in urban areas as well as water pollution, control of disease, and waste disposal in an attempt to improve the situation of the inhabitants (International Experts Meeting 1984).

When comparing European studies on urban nature conservation with contributions from North America, it is striking that the protection of game animals has almost no significance in the European programmes, where nature conservation is taken to refer to vertebrates that are not hunted, invertebrates and plants.

The widespread investigation and development of "urban forests" (McBride & Jacobs 1976, Grey & Deneke 1975, Rowntree 1986) in Anglo-American regions is not seen in Europe where the main concern is with the effect of man on forests near towns and cities. Inner-city areas are included only in Sweden and The Netherlands. For Düsseldorf Kürsten (1983) has studied roadside trees by using forestry methods and developed planning concepts.

Since 1970 there has been a gradual acceptance that nature conservation should include urban areas and their surroundings. Since the first quarter of this century pioneering naturalists and scientists, such as Jovet (1954), emphasized the importance of biological diversity in human-dominated systems. This has paved the way for cities to become an important component of nature conservation (Barker 1997).

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Urban Ecological Systems: Linking Terrestrial Ecological, Physical, and Socioeconomic Components of Metropolitan Areas

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Abstract Ecological studies of terrestrial urban systems have been approached along several kinds of contrasts: ecology in as opposed to ecology of cities; biogeochemical compared to organismal perspectives, land use planning versus biological, and disciplinary versus interdisciplinary. In order to point out how urban ecological studies are poised for significant integration, we review key aspects of these disparate literatures. We emphasize an open definition of urban systems that accounts for the exchanges of material and influence between cities and surrounding landscapes. Research on ecology in urban systems highlights the nature of the physical environment, including urban climate, hydrology, and soils. Biotic research has studied flora, fauna, and vegetation, including trophic effects of wildlife and pets. Unexpected interactions among soil chemistry, leaf litter quality, and exotic invertebrates exemplify the novel kinds of interactions that can occur in urban systems. Vegetation and faunal responses suggest that the configuration of spatial heterogeneity is especially important in urban systems. This insight parallels the concern in the literature on the ecological dimensions of land use planning. The contrasting approach of ecology of cities has used a strategy of biogeochemical budgets, ecological footprints, and summaries of citywide species richness. Contemporary ecosystem approaches have begun to integrate organismal, nutrient, and energetic approaches, and to show the need for understanding the social dimensions of urban ecology. Social structure and the social allocation of natural and institutional resources are subjects that are well understood within social sciences, and that can be readily accommodated in ecosystem models of metropolitan areas. Likewise, the sophisticated understanding of spatial dimensions of social differentiation has parallels with concepts and data on patch dynamics in ecology and sets the stage for comprehensive understanding of urban ecosystems. The linkages are captured in the human ecosystem framework.

Keywords: city · hierarchy theory · integration · patch dynamics · urban ecology

Introduction: Justification for Urban Ecological Studies

Urbanization is a dominant demographic trend and an important component of global land transformation. Slightly less than half of the world's population now resides in cities, but this is projected to rise to nearly 60% in the next 30 years (United Nations 1993). The developed nations have more urbanized populations; for example, close to 80% of the US population is urban. Urbanization has also resulted in a dramatic rise in the size of cities: over 300 cities have more than 10^6 inhabitants

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and 14 megacities exceed 10^7. The increasing population and spatial prominence of urban areas is reason enough to study them, but ecologists must also inform decision makers involved in regional planning and conservation. Proper management of cities will ensure that they are reasonable places to live in the future.

In addition to its global reach, urbanization has important effects in regional landscapes. For example, in industrialized nations, the conversion of land from wild and agricultural uses to urban and suburban occupancy is growing at a faster rate than the population in urban areas. Cities are no longer compact, isodiametric aggregations; rather, they sprawl in fractal or spider-like configurations (Makse et al. 1995). Consequently, urban areas increasingly abut and interdigitate with wild lands. Indeed, even for many rapidly growing metropolitan areas, the suburban zones are growing faster than other zones (Katz & Bradley 1999). The resulting new forms of urban development, including edge cities (Garreau 1991) and housing interspersed in forest, shrubland, and desert habitats, bring people possessing equity generated in urban systems, expressing urban habits, and drawing upon urban experiences, into daily contact with habitats formerly controlled by agriculturalists, foresters, and conservationists (Bradley 1995).

Urban habitats constitute an open frontier for ecological research. Ecologists have come to recognize that few ecosystems are totally devoid of direct or subtle human influence (McDonnell et al. 1993). Yet urban systems are relatively neglected as an end member with which to compare the role of humans in ecosystems. Notably, many classic geographic studies of cities, which offer valuable insights to ecologists, are based on outmoded ecological theory such as deterministic models of succession and assumptions of equilibrium dynamics of ecosystems. Hence classical ecological approaches and the geographic studies that have relied on them have not been as useful as they would be otherwise (Zimmerer 1994).

Although the ecology of urban areas has long elicited the academic attention of ecologists, physical and social scientists, and regional planners, there is much opportunity to extend and integrate knowledge of the metropolis using an ecological lens. The purpose of this paper is to review the status of ecological knowledge of the terrestrial components of urban areas and to present a framework for continued ecological research and integration with social and economic understanding. This paper complements the review of aquatic components of urban systems by Paul & Meyer (2001).

Definition and Roots of Urban Ecology

Urban ecosystems are those in which people live at high densities, or where the built infrastructure covers a large proportion of the land surface. The US Bureau of the Census defines urban areas as those in which the human population reaches or exceeds densities of 186 people per km^2. However, an ecological understanding of urban systems also must include less densely populated areas because of reciprocal flows and influences between densely and sparsely settled areas. Comparisons along gradients of urbanization can capture the full range of urban effects as well as the existence of thresholds. Therefore, in the broadest sense, urban ecosystems comprise suburban areas, exurbs, sparsely settled villages connected by commuting corridors or by utilities, and hinterlands directly managed or affected by the energy and material from the urban core and suburban lands.

The boundaries of urban ecosystems are often set by watersheds, airsheds, commuting radii, or convenience. In other words, boundaries of urban ecosystems are set in the same ways and for the same reasons as are the boundaries in any other ecosystem study. In the case of urban ecosystems, it is clear that many fluxes and interactions extend well beyond the urban boundaries defined by political, research, or biophysical reasons. Urban ecology, as an integrative subdiscipline of the science of ecology, focuses on urban systems as broadly conceived above. There is little to be gained from seeking distinctions between "urban" and abutting "wild" lands, as a comprehensive, spatially extensive, systems approach is most valuable for science (Pickett et al. 1997) and management (Rowntree 1995).

There are two distinct meanings of urban ecology in the literature (Sukopp 1998). One is a scientific definition, and the other emerges from urban planning. In ecology, the term urban ecology refers to studies of the distribution and abundance of organisms in and around cities, and on the biogeochemical budgets of urban areas. In planning, urban ecology has focused on designing the environmental amenities of cities for people, and on reducing environmental impacts of urban regions (Deelstra 1998). The planning perspective is normative and claims ecological justification for specific planning approaches and goals. We review key aspects of these complementary approaches and then frame a social-ecological approach to integrate these two approaches.

Biogeophysical Approaches

There are two aspects to the biogeophysical approach to urban ecological studies. One, the pioneering and most common approach, examines ecological structure and function of habitats or organisms within cities. This approach is called ecology in cities. The second, more recent and still emerging approach, examines entire cities or metropolitan areas from an ecological perspective. The second approach is labeled ecology of cities (Grimm et al. 2000). Although the differences in the prepositions in the phrases identifying the two contrasting approaches may appear subtle, the understanding achieved by identifying them as poles between which urban ecological studies sort out, is crucial to understanding the history of urban ecology, and the integration it is now poised to make. We review literature that has taken these two contrasting approaches in turn.

Ecology in the City

The study of ecology in the city has focused on the physical environment, soils, plants and vegetation, and animals and wildlife. These studies are the foundation for understanding urban ecosystems. The literature in this area has taken a case study approach, and unifying themes are still to emerge. We highlight key examples from among the many cases.

Urban Physical Environment

The urban heat island constitutes climate modification directly related to urban land cover and human energy use (Oke 1995). The urban heat island describes the difference between urban and rural temperatures. Such differences often are negligible in the daytime but develop rapidly after sunset, peaking 2–3 h later. Ambient air temperatures may reach maxima of 5–10° C warmer than hinterlands (Zipperer et al. 1997). For example, New York City is, on average, 2–3° C warmer than any other location along a 130-km transect into surrounding rural areas (McDonnell et al. 1993). The duration and magnitude of the temperature differential depend on the spatial heterogeneity of the urban landscape. As the percentage of artificial or human-made surfaces increases, the temperature differential increases. Hence, the urban core is warmer than neighboring residential areas, which are warmer than neighboring farmlands or forests. The differences also change seasonally. For example, cities in mid-lattitudes of the United States are typically 1–2° C warmer than the surroundings in winter, and 0.5–1.0° C warmer in summer (Botkin & Beveridge 1997).

The heat island effect varies by region, as seen in a comparison between Baltimore, Maryland, and Phoenix, Arizona (Brazel et al. 2000). During the summer, mean maximum temperatures in Baltimore were greater than in the rural landscape. Phoenix, on the other hand, became an oasis, with cooler temperatures than the surrounding desert. The cooling of Phoenix is due to the watering of mesic plantings in the city. In contrast, the mean minimum temperatures were warmer in both cities than in the respective neighboring rural landscape, although the differential in Phoenix was greater.

The heat island intensity also is related to city size and population density (Oke 1973, Brazel et al. 2000). For example, Baltimore's mean minimum temperature differential increased until the 1970s when the city experienced a decline in population. Since 1970, the mean minimal temperature differential has leveled off. Phoenix also showed an increase in mean minimum temperature differential with an increase in population. However, Phoenix is the second fastest growing metropolitan area in the United States, so the differential has continued to increase with population growth. In general, a nonlinear relationship exists between mean minimum temperature differential and population density (Brazel et al. 2000).

The differences in climate between city and countryside have biological implications. For example, as a result of climatic modification in temperate zone cities, leaf emergence and flowering times are earlier, and leaf drop is later than in the surrounding countryside (Sukopp 1998). Increased temperatures in and around cities enhance ozone formation, and increase the number of officially recognized pollution days and trace gas emissions (Sukopp 1998). Ozone concentrations tend to be highest in and around urban areas. As urban areas have expanded through processes of suburban sprawl, the spatial influence of urbanization has increased. Within regions of ozone pollution, agricultural crops may be adversely affected and yields decrease 5–10% (Chameides et al. 1994). Crop type and stage of development, and the degree, spatial extent, and duration of ozone exposure may all influence the decrease in production (Chameides et al. 1994).

Precipitation is enhanced in and downwind of cities as a result of the higher concentrations of particulate condensation nuclei in urban atmospheres. Precipitation can be up to 5–10% higher in cities, which can experience greater cloudiness and fog (Botkin & Beveridge 1997). The probability of precipitation increases toward the end of the work week and on weekends due to a buildup of particulates resulting from manufacturing and transportation (Collins et al. 2000).

Urban hydrology is drastically modified compared to agricultural and wild lands. This topic is covered more fully in a companion review (Paul & Meyer 2001). Relativizing a water budget to 100 units of precipitation, and comparing urban to nonurban areas, evapotranspiration decreases from a value of 40% to 25%, surface runoff increases from 10% to 30%, and groundwater decreases from 50% to 32% (Hough 1995). Forty-three percent of precipitation exits the urban area via storm sewers, with 13% of that having first fallen on buildings. The role of impervious surfaces is crucial to the functioning of urban watersheds (Dow & DeWalle 2000). The hydrology in urban areas can be further modified by ecological structures. For example, reduced tree cover in urban areas increases the rate of runoff and decreases the time lag between initiation of storms and initiation of runoff (Hough 1995). The increased runoff in urban areas changes the morphology of urban streams, which become deeply incised in their floodplains. Remnant riparian vegetation may suffer as a result of isolation from the water table.

Urban Soils

Soils in urban landscapes retain and supply nutrients, serve as a growth medium and substrate for soil fauna and flora, and absorb and store water. Soils also intercept contaminants such as pesticides and other toxics generated through human activities (Pouyat & McDonnell 1991). However, in urban settings soils are modified by human activity and, consequently, are functionally altered (Effland & Pouyat 1997). In addition, completely new substrates are created by deposition of debris, soil, and rock in urban sites. Such new substrates are called made land.

As land is converted to urban use, both direct and indirect factors can affect the functioning of soils. Direct effects include physical disturbances, burial of soil by fill material, coverage by impervious surfaces, and additions of chemicals and water (e.g., fertilization and irrigation). Direct effects often lead to highly modified substrates in which soil development then proceeds (Effland &

Pouyat 1997). Indirect effects change the abiotic and biotic environment, which in turn can influence soil development and ecological processes in intact soils. Indirect effects include the urban heat island (Oke 1995), soil hydrophobicity (White & McDonnell 1988), introductions of nonnative plant and animal species (Airola & Buchholz 1984, Steinberg et al. 1997), and atmospheric deposition of pollutants (Lovett et al. 2000). Moreover, toxic, sublethal, or stress effects of the urban environment on soil decomposers and primary producers can significantly affect the quality of organic matter and subsequent soil processes (Pouyat et al. 1997).

The results from a transect along an urban-rural land-use gradient in the New York City metropolitan area show the influence of urban environments on intact forest soils (McDonnell et al. 1997). Along this transect, human population density, percent impervious surface, and automobile traffic volume were significantly higher at the urban than the rural end of the gradient (Medley et al. 1995). Soil chemical and physical properties, soil organism abundances, and C and N processes were investigated along this gradient to assess the sometimes complex and sometimes contradictory interactions between urban soil chemistry, local leaf litter quality, and exotic species.

Soil chemistry significantly correlated with measures of urbanization. Higher concentrations of heavy metals (Pb, Cu, Ni), organic matter, salts, and soil acidity were found in the surface 10 cm of forest soils at the urban end of the transect (Pouyat et al. 1995). The most probable factor is metropolitan-wide atmospheric deposition.

Litter decomposition studies in both the field and the laboratory determined that urban-derived oak litter decomposed more slowly than rural-derived oak litter under constant conditions (Carreiro et al. 1999), with lignin concentration explaining 50% of the variation. Moreover, a site effect was measured for reciprocally transplanted litter, as decomposition was faster in the urban than in the rural sites, regardless of litter origin. This result is surprising because the lower litter fungal biomass and microinvertebrate abundances found in urban stands compared to rural (Pouyat et al. 1994) would lead to an expectation of slower litter turnover rates in urban forests. However, abundant earthworms (Steinberg et al. 1997) and higher soil temperatures in the urban stands may compensate for lower litter quality, lower fungal biomass, and lower microinvertebrate abundances in the urban stands (R.V. Pouyat & M.M. Carreiro, submitted for publication). Such compensation likely explains the faster decomposition rates in the urban stands.

Litter quality, site environment, and soil organism differences between the urban and rural sites also affected C and N pools and processes in the soil. Urban litter was found to be of lower quality (higher C:N ratio) than rural litter. Typically, poor quality litter either decreases the rate at which labile C mineralizes N or increases the amount of organic matter transferred to recalcitrant pools, or both. Hence, urban litter is expected to be more recalcitrant than rural litter. Indeed, measurements of soil C pools along the urban-rural transect suggest that recal-citrant C pools are higher and passive pools lower in urban forest soils relative to rural forest soils (Groffman et al. 1995). Consequently, N mineralization rates were expected to be lower in urban stands. However, the opposite was found. Net potential N-mineralization rates in the A-horizon were higher in the urban stands than in the rural stands (Pouyat et al. 1997). Soil cores taken from urban areas accumulated more NH_4^+ and NO_3^- than soil from rural areas. In addition, from a reciprocal transplant experiment, soil cores incubated at urban sites accumulated more inorganic N than cores incubated at the rural sites, regardless of where the cores originated. These net N mineralization rates contradict both the litter decomposition results from the transplant experiment discussed above and expectations derived from measurements of soil C pools.

In contrast, when net N-mineralization rates were measured for mixed O, A, and B horizon material (15 cm depth), the total inorganic N pool accumulated was higher in rural than in urban sites, although less NO_3^- was accumulated in rural samples (Goldman et al. 1995). The mechanisms behind these results have yet to be elucidated, though it has been hypothesized that methane consumption rates and the biomass of methanotrophs are important regulators in overall soil C and N dynamics (Goldman et al. 1995).

Intense soil modifications resulting from urbanization may potentially alter soil C and N dynamics. To assess the potential effect on soil organic C, data from "made" soils (1 m depth) from five different cities, and surface (0–15 cm) soils from several land use types in Baltimore were analyzed (R.V. Pouyat, P.M. Groffman, I. Yesilonis & L. Hernandez, submitted for publication). Soil pedons from the five cities showed the highest soil organic C densities in loamy fill (28.5 kg m^{-2}) with the lowest in clean fill and old dredge materials (1.4 and 6.9 kg m^{-2}, respectively). Soil organic C for residential areas (15.5 ± 1.2 kg m^{-2}) was consistent across cities. A comparison of land-use types showed that low density residential and institutional land had 44% and 38% higher organic C densities than commercial land, respectively. Therefore, made soils, with their physical disturbances and inputs of various materials by humans, can greatly alter the amount of C stored in urban systems.

The complex patterns of C and N dynamics that have emerged from the studies reviewed above indicate interactions between key soil and organism processes in urban environments. Simple predictions based on trends in pollution, stress, or exotic species alone are inadequate to understand the complex feedbacks between these three governing factors of urban soil dynamics. Studies of soil C and N dynamics in unmanaged urban forests, highly disturbed soils, and surface soils of various urban land-use types all show that urbanization can directly and indirectly affect soil C pools and N-transformation rates. Our review also suggests that soil C storage in urban ecosystems is highly variable. How generalizable these results are across cities located in similar and dissimilar life zones needs to be investigated. In addition, more data are needed on highly disturbed soils, such as landfill, managed lawns, and covered soils to make regional and global estimates of soil C storage and N-transformation rates in urban ecosystems. Specific uncertainties include the quality of the C inputs governed by the input of litter from exotic plant species and by stress effects on native species litter, the fate of soil C in covered soils, measurements of soil C densities at depths greater than 1 m, particularly in made soils, and the effects of specific management inputs on N-transformation rates.

Vegetation and Flora in Cities

The assessment of vegetation in urban landscapes has a long history. For example, in Europe, studies by De Rudder & Linke (1940) documented the flora and fauna of cities during the early decades of the twentieth century. After World War II, Salisbury (1943) examined vegetation dynamics of bomb-sites in cities. At the same time, ecologists in the United States focused on describing flora in areas of cities minimally altered by humans, such as parks and cemeteries. One of the first comprehensive studies of urban vegetation and environments was conducted by Schmid (1975) in Chicago.

The structure and composition of vegetation has been one of the foci of ecological studies in cities, and these studies have documented large effects of urbanization on forest structure. Urban stands tend to have lower stem densities, unless those stands are old-growth remnants in large parks or former estates (Lawrence 1995). In the Chicago region, street trees and residential trees tend to be larger than those in forest preserves, natural areas, and wild lands (McPherson et al. 1997), although individual trees in urban sites are often stressed, especially on or near streets (Ballach et al. 1998). Street trees in Chicago account for 24% of the total Leaf Area Index (LAI) and 43.7% of the LAI in residential areas (Nowak 1994). In mesic forest regions of the United States, tree cover of cities is approximately 31%, compared to the nearly continuous forest cover the areas would have supported before settlement. Brooks & Rowntree (1984) quantified the forest cover in counties classified as nonmetropolitan, peripheral to a central city, and encompassing a central city. They found that the proportion of total land area in forest decreased from nonmetropolitan to central city counties, with the steepest reductions between nonmetropolitan and peripheral counties. In contrast, for prairie-savanna and desert regions, tree cover in cities is greater than in pre-urban conditions (Nowak et al. 1996).

There are also local effects on urban vegetation. For example, forest patches adjacent to residential areas have increased edge openness (Moran 1984), and their margins have retreated because of recreational use, especially by children, and damage to regeneration (Bagnall 1979). Such regeneration failure is frequent in urban and suburban stands, owing to reduced natural disturbance or gap formation, substitution for natural disturbances of unfavorable anthropogenic disturbances such as frequent ground fires, trampling, and competition with exotics (Guilden et al. 1990).

The composition of urban and suburban forests differs from that of wild and rural stands in several ways. Species richness has increased in urban forests as a whole, but this is because of the increased presence of exotics (Zipperer et al. 1997). Urban areas show a preponderance of trees of wetland or floodplain provenance, owing to the lower oxygen tensions shared by wetlands and impervious urban soils (Spirn 1984). Even when the tree composition remains similar between urban and rural forests, the herbaceous flora of urban forests is likely to differ between the two types of forest (Wittig 1998). Such compositional trends reflect the context and configuration of the forest stands in urban areas. For example, vascular plant diversity increases with the area of the stand (Iida & Nakashizuka 1995, Hobbs 1988). Furthermore, in some urban stands, the adjacent land use affects species composition. For example, the interior of forests in residential areas often has more exotics than forests abutting either roads or agricultural zones (Moran 1984).

The role of exotic species has received particular attention in urban studies. The percentage of the flora represented by native species decreased from urban fringes to center city (Kowarik 1990). Along the New York City urban-to-rural gradient, the number of exotics in the seedling and sapling size classes of woody species was greater in urban and suburban oak-dominated stands (Rudnicky & McDonnell 1989). Rapoport (1993) found the number of noncultivated species decreased from fringe toward urban centers in several Latin American cities for various reasons exemplified in different cities. In Mexico City, there was a linear decrease in the number of species from 30–80 ha^{-1} encountered in suburbs to 3–10 ha^{-1} encountered in the city center. The social context also influenced species richness. At a given housing density, more affluent neighborhoods had more exotic species than less affluent ones in Bariloche, Argentina. In Villa Alicura, Argentina, exotic species increased with increasing local site alteration by humans. Near homes there were 74% exotics, while there were 48% along river banks. Although exotics were present along all roads, the number decreased on roads less frequently used. No exotic species were found outside the town. Pathways in rural recreation areas (Rapoport 1993) and in urban parks (Drayton & Primack 1996) have enhanced the presence of exotics. In an urban park in Boston, of the plant species present in 1894, 155 were absent by 1993, amounting to a decrease from 84% to 74% native flora. Sixty-four species were new. In addition to trails, Drayton & Primack (1996) blamed fire and trampling for the change in exotics.

Structural and compositional changes are not the only dynamics in urban biota. Plants and animals in cities have evolved in response to the local conditions in cities (Sukopp 1998, Bradshaw & McNeilly 1981). The famous population genetic differentiation in copper and zinc tolerance in creeping bent grass (*Agrostis stolonifera*) and roadside lead tolerance of ribwort plantain (*Plantago lanceolata*) in urban sites is an example (Wu & Antonovics 1975a, b). Industrial melanism and other forms of melanism are also urban evolutionary responses (Bishop & Cook 1980).

A major feature of urban vegetation is the spatial heterogeneity in urban landscapes created by the array of building densities and types, different land uses, and different social contexts. To characterize this heterogeneity, plant habitats have been subjected to various classification systems. Forest Stearns (Stearns 1971), one of the first American ecologists to call for research in urban landscapes, identified three major vegetation types—ruderal, managed, and residual. In addition to assessing spatial heterogeneity, classification systems recognize the importance of characterizing natural habitats in urban landscapes. Rogers & Rowntree (1988) developed a system to classify vegetation using life forms. This process was used to assess natural resources in New York City (Sisinni & Emmerich 1995). Based on site histories, Zipperer et al. (1997) classified tree-covered habitats as planted, reforested, or remnant. These approaches allow for spatially explicit comparisons between vegetation and other variables of interest for a single point in time.

The vegetation and floristic studies of nature in cities share key characteristics. They are largely descriptive, but illustrate spatial heterogeneity as a source of diversity, and suggest a functional role for landscape structure (Rebele 1994). Although it is legitimate to view the city as an open and dynamic ecosystem, few of the plant ecological studies document successional processes (Matlack 1997) or expose the functional relationships of the vegetation (Mucina 1990).

Animals and Wildlife

Douglas (1983), Gilbert (1989), VanDruff et al. (1994), and Nilon & Pais (1997) have summarized the literature on the animal life of cities. They reveal a long history of research on the fauna of European and North American cities. Although much of the research has been descriptive, several studies focused on processes that are also major foci of research in ecology as a whole. We review animal studies starting with coarse scale or gradient comparisons, followed by patch-oriented studies, including some that incorporate socioeconomic aspects, then show how wildlife biologists have contributed to urban ecology, and end with several examples of process studies involving animals.

Birds, mammals, and terrestrial invertebrates are the best studied taxonomic groups, with aquatic fauna, reptiles, and amphibians less studied (Luniak & Pisarski 1994). In addition, the fauna of green spaces are relatively well studied, with built-up and derelict areas and water bodies less well known. Andrzejewski et al. (1978) and Klausnitzer & Richter (1983) described how urban-to-rural gradients defined by human population density and building density impact the occurrence and abundance of various animal species. Among mammals, there is a shift to medium sized, generalist predators such as racoons and skunks in urban areas (Nilon & Pais 1997). The predator fauna contrasts between cities and nonurban environments. For example, ants were the most important predators in some urban areas although vertebrates were more significant in nonurban habitats (Wetterer 1997). Changes in fauna may influence vegetation. For example, invertebrate pest densities increased in urban trees compared to wild forest stands, reflecting the additional stress and damage to which urban trees are subjected (Nowak & McBride 1992).

Animal ecologists have recognized spatial heterogeneity in a variety of ways. Heterogeneity is typically expressed as patches or contrasting biotopes. The approach to spatial heterogeneity of urban fauna in Germany has been shaped by a national program of biotope mapping (Sukopp 1990, Werner 1999). The German biotope approach is based on phytosociologic-floristic and faunal characteristics of sites in cities (Sukopp & Weiler 1988). Biotope mapping can be used to document dynamics of urban fauna such as the changes in the bird species composition, and abundance of different types of biotopes (Witt 1996). Brady et al.'s (1979) general typology of urban ecosystems and Matthews et al.'s (1988) habitat classification scheme for metropolitan areas in New York State are examples of schemes that recognize how patterns of land-use history and resulting changes in land cover create ecologically distinct habitat patches.

Use of patch-oriented approaches is particularly well developed in studies of mammals. For example, land use and cover in 50-ha areas surrounding patches in Syracuse, New York, were the best predictors of small mammal species composition (VanDruff & Rowse 1986). Similarly, patch configuration in Warsaw, Poland, affected mammal populations. Urbanization blocked the dispersal of field mice and altered population structure and survivorship in isolated patches (Andrzejewski et al. 1978). Larger mouse populations were associated with increased percentages of built and paved areas, barriers to emigration, and decreases in patch size (Adamczewska-Andrzejewski et al. 1988). In addition, mammalian predators of seeds may be increased in urban patches compared to rural areas (Nilon & Pais 1997). The change in patch configuration resulting from suburban sprawl has led to the widespread increase in deer densities on urban fringes. Suburbanization has caused increased juxtaposition of forest habitats in which deer shelter with field or horticultural patches in which they feed (Alverson et al. 1988). The positive feedback of changing patch configuration on deer population density is magnified by reduced hunting and predator pressure associated with suburbanization. Greater deer densities have the potential to change both the animal community and vegetation (Bowers 1997).

Some biologists have looked to social causes of animal and plant distribution and abundance in cities. Studies of flora and fauna of metropolitan Liverpool, England, showed that changes in land use and technology have influenced habitat change since the industrial revolution (Greenwood 1999), and studies of fauna in suburban Warsaw included land-use information dating from the 1700s (Mackin-Rogalska et al. 1988). Key to the Warsaw study was the recognition that suburban areas have different socioeconomic processes than do either urban or rural areas, and that these processes influence land use, land-cover characteristics, and habitats for biota. Examples of the relationships of animal populations to land-cover types include those for birds. For a city as a whole, exotic generalists such as pigeons, starlings, and sparrows can constitute 80% of the bird community in the summer, and 95% in winter (Wetterer 1997). At finer scales, contrasting land-cover patches of residential areas, commercial sites, and parks supported different avian assemblages (Nilon & Pais 1997). Even differences in plant cover among residential neighborhoods affected bird community composition. Likewise, the land uses adjacent to urban green spaces strongly influence bird communities (Nilon & Pais 1997).

In Europe and North America, much of the research on the fauna of cities has been conducted by applied ecologists to support conservation and natural resource management. This work recognized that cities are areas worthy of study, and more importantly, that people and their activities create a unique context for natural resource management in and around cities (Waggoner & Ovington 1962, Noyes & Progulske 1974). In particular, the patch dynamics approach was foreshadowed by wildlife ecologists in cities. The activity of urban wildlife biologists is illustrated by the conferences of the National Institute for Urban Wildlife (Adams & Leedy 1987, 1991, Adams & VanDruff 1998). These conferences focused on (*a*) ecology of urban wildlife, (*b*) planning and design, (*c*) management issues and successes, and (*d*) public participation and education. The participatory approach to studying animals in cities involves urban residents in the process (Adams & Leedy 1987) and serves as a model for other ecological research in cities. Furthermore, the conferences focused on planning and management as activities that can change habitats at both citywide and local scales.

An additional stress on animal populations in cities is the direct or indirect effect of domestic pets. Building density is associated with an increase in the numbers of pets in an area (Nilon & Pais 1997). The amount of food energy available to free-ranging domestic cats depends on the affluence of the immediate neighborhood (Haspel & Calhoon 1991). Cats have a significant impact on bird populations in suburbs (Churcher & Lawton 1987).

An example of an ecological process familiar in wild and production landscapes that also appears in cities is animal succession. After the establishment of the new town of Columbia, Maryland, birds such as bobwhite and mourning dove, which are associated with agriculture, gave way to starlings and house sparrows, which had been absent before urbanization (Hough 1995). In an urban park near Dortmund, Germany, a 35-year study found an increase in generalist species richness and density at the expense of specialists. The species turnover rate of birds in the park between 1954 and 1997—42.1%—was higher than in a forest distant from the city over the same period (Bergen et al. 1998).

This overview of animals in urban areas has confirmed the diversity of exotic and native species in cities. These species, along with the planted and volunteer vegetation in cities, are in some cases an important amenity, and in others a significant health or economic load. They can serve to connect people with natural processes through educational activities. Although there is a great deal of descriptive knowledge about the biota of cities, there is the need to compare food web models for green and built parts of the city, to link these data with ecosystem function, and to quantify the relationships between infrastructural and human behavioral features of the metropolis (Flores et al. 1997). In addition, long-term studies are required. The case studies we report here are a foundation for generalizations concerning the structure and function of biota within cities.

Ecology of the City

The knowledge of nature in cities is a firm foundation for understanding ecological processes in metropolitan areas. Yet it is not sufficient (Flores et al. 1997). If scientists, planners, and decision makers are to understand how the social, economic, and ecological aspects of cities interact, the feedbacks and dynamics of the ecological linkages must be assessed. We therefore turn to a review of systems-oriented approaches to urban ecology. These represent a shift to the perspective of ecology of cities, as contrasted with the literature we have reviewed so far, which focused on ecology in cities.

The diverse spatial mosaics of metropolitan areas present a variety of ecological situations in which to examine ecological structure and dynamics. For example, several of the conditions in cities are analogous to major predictions of global climate change. Increased temperatures, altered rainfall patterns, and drying of soils anticipate trends projected for some wild lands. Examination of existing urban assemblages or experimentation with novel assemblages of native and exotic species may be useful for assessing the effects of climate change on biodiversity. The stranded riparian zones of urban sites, resulting from the downcutting of streams associated with impervious surfaces, can be used to examine altered environmental drivers of system function. Plant community regeneration and the response of ecosystems to soil nitrogen cycling under altered moisture and temperature conditions may be investigated in such areas. Examining succession in vacant lots may inform practical vegetation management and suggest strategies for changing land use as the density of humans and buildings decline in some city centers. Finally, patterns of adjacency of managed and wild patches in and around cities can be used to examine landscape function.

The ecology of the entire city as a system is represented by research relating species richness to the characteristics of cities. For instance, the number of plant species in urban areas correlates with the human population size. Species number increases with log number of human inhabitants, and that relationship is stronger than the correlation with city area (Klotz 1990). Small towns have from 530 to 560 species, while cities having 100,000 to 200,000 inhabitants have upwards of 1000 species (Sukopp 1998). The age of the city also affects the species richness; large, older cities have more plant species than large, younger cities (Sukopp 1998, Kowarik 1990). These plant assemblages are characteristic throughout Europe, with 15% of species shared among cities (Sukopp 1998).

Biogeophysical Budgets

One of the earliest modern ecological approaches to urban systems was the assessment of biogeochemical budgets of whole cities (Odum & Odum 1980). It is clear that urban areas are heterotrophic ecosystems that depend on the productivity from elsewhere (Collins et al. 2000). Cities in industrial countries may use between 100,000 and 300, 000 Kcal m^{-2} yr^{-1}, whereas natural ecosystems typically expend between 1,000 and 10, 000 Kcal m^{-2} yr^{-1} (Odum 1997). The energy budgets of cities are driven by fossil fuel subsidies, which contribute to the urban heat island effect.

How the green component of cities affects biogeochemical processing for the city as a whole has been examined. For example, in Chicago, trees have been estimated to sequester 5575 metric tons of air pollution, and 315,800 metric tons of C per year at an average rate of 17 metric tons ha^{-1} y^{-1} (McPherson et al. 1997). In the Mediterranean climate of Oakland, California, trees sequester 11 metric tons of C ha^{-1} y^{-1} (Nowak 1993). In contrast to urban forests, natural forests on average sequester 55 metric tons ha^{-1} y^{-1} (Zipperer et al. 1997). The capacity of trees to filter particulates from urban air is based on leaf size and surface roughness (Agrawal 1998).

Urban areas also concentrate materials from elsewhere. For example, the elevation of the surface in old cities is generally higher than the surrounding areas as a result of importing construction materials (Sukopp 1998). A carbon dioxide dome accumulates over cities in association with combustion of fossil fuels (Brazel et al. 2000). Anthropogenically produced forms of N are concentrated in (Lovett et al. 2000) and downwind of cities (Chamiedes et al. 1994). Nitrogen accumulated from human metabolism and fertilizers concentrates downstream of cities. Hence, population density of the watersheds is statistically correlated with N loading in the major rivers of the world (Caraco & Cole 1999).

A useful way to quantify the dependence of urban systems on ecosystems beyond their borders is the concept of the ecological footprint. The ecological footprint of an urban area indexes the amount of land required to produce the material and energetic resources required by, and to process the wastes generated by, a metropolis (Rees 1996). The city of Vancouver, Canada, requires 180 times more land to generate and process materials than the city actually occupies. The concept is highly metaphorical, because the actual networks from which any particular city draws resources, and the areas affected by its waste, may extend around the globe. An analysis of the growth of Chicago (Cronon 1991) showed that a network of resource acquisition extended throughout the western regions of the United States in the late nineteenth century. The metropolis in the postindustrial, information age in nations enjoying a high fossil fuel subsidy has different connections with the hinterland than did the industrially and agriculturally anchored Chicago of a century ago (Bradley 1995). Telecommuting, materially and energetically subsidized recreation, and the alteration of land values for urban uses in the countryside represent a footprint based on urban capital.

Ecosystem Pattern and Process

There are three ways in which contemporary studies of ecology of cities, or more properly entire metropolitan areas, are prepared to move beyond the classical ecological approaches and to support integration with social and physical sciences. 1. The contemporary assumptions of ecosystem function are more inclusive than the classical assumptions. 2. The net effects approach to ecosystem budgets has evolved to consider multiple processes and spatial heterogeneity. Finally, 3. the narrow theories that were used to bridge disciplines in the past have been broadened or replaced.

The first tool for integration is the contrast between contemporary and classical assumptions about ecosystem function. Classically, ecologists based studies of urban areas on the assumptions that ecosystems were materially closed and homeostatic systems. Such assumptions were a part of the ecosystem theory used by many early geographers (Zimmerer 1994). These assumptions have been replaced (Zimmerer 1994, Pickett et al. 1992). Consequently, there is a new theory of ecosystems that was not available to those who pioneered the budgetary approach to urban systems (e.g., Boyden et al. 1981). What remains is the basic concept of the ecosystem as a dynamic, connected, and open system (Likens 1992), which can serve the various disciplines (Rebele 1994) that need to be integrated to form a more comprehensive theory to support joint ecological, social, and physical study of urban systems.

Contemporary ecology propounds a systems view that builds on the rigorous budgetary approach to ecosystems (e.g., Jones & Lawton 1995). Of course ecologists necessarily continue to exploit the laws of conservation of matter and energy to generate budgets for ecosystem processes. However, many contemporary studies of ecosystem budgets do not treat systems as though they were black boxes. Rather, the structural details and richness of processes that take place within the boundaries of the system are a major concern of contemporary ecosystem analysis. Contemporary ecosystem ecology exposes the roles of specific species and interactions within communities, flows between patches, and the basis of contemporary processes in historical contingencies. These insights have not been fully exploited in urban ecological studies.

The third feature of contemporary ecology is the breadth of key theories that can be used in integration. Classical ecological theory provided social scientists with only a narrow structure for integration with ecology. The social scientists of the Chicago School, which was active in the early decades of the twentieth century, used ecologically motivated theories of succession and competition, for example. At the time, only the relatively general, deterministic, and equilibrium versions of those theories were available. Contemporary theories of interaction account for both positive and competitive effects, and predictions are based on the actual mechanisms for interaction rather than net effects. In contemporary succession theory, mechanisms other than facilitation are included, and the sequence of communities may not be linear or fixed. In addition, ecologists have come to recognize that those theories have hierarchical structure, which allows them to address different levels of generality and mechanistic detail depending on the scale of the study or the scope of the research questions (Pickett et al. 1994). Hence, social scientists and ecologists now can select the most appropriate levels of generality in a theoretical area for integration. No longer must integration rely on general theories of net effects of such processes as competition and succession.

Urban Ecology as a Planning Approach

Although the first volume of the journal *Ecology* contained a scientific paper devoted to the effect of weather on the spread of pneumonia in New York and Boston (Huntington 1920), the interactions of humans with the urban environment have been primarily the province of planners and landscape architects. For example, Central Park in New York City and other urban parks designed by Frederick Law Olmsted seem intuitively to link environmental properties to human well being in cities. In particular, Olmsted's design for the Boston Fens and Riverway shows ecological prescience in its sophisticated combination of wastewater management and recreational amenity (Spirn 1998). Ian McHarg's (1969) *Design with Nature* alerted planners and architects to the value of incorporating knowledge of ecological and natural features among the usual engineering, economic, and social criteria when developing a regional plan. In McHarg's approach, environmental risks and amenities of different types are mapped on separate layers. The composite map suggests where certain types of development should or should not occur. This approach presaged the technology of Geographic Information Systems (GIS), which has become an important tool to incorporate multiple criteria in planning (Schlutnik 1992, Grove 1997) such as those proposed by McHarg (1969). A more explicit ecological approach is that of Spirn (1984), who examined how natural processes are embedded in cities, and how the interaction between the built environment and natural processes affected economy, health, and human community. For instance, she showed how the forgotten environmental template of drainage networks continued to affect infrastructure and the social structure of a Philadelphia neighborhood.

The planning perspective of human ecology is especially strong in Europe (Sukopp 1998). Planning in Germany has been heavily influenced by a national program of biotope mapping that includes cities (Sukopp 1990, Werner 1999). This program includes descriptions of the flora and fauna of biotopes as a key to identifying types of habitats that are significant for 1. protecting natural resources, 2. quality of life, and 3. a sense of place and identity in the city (Werner 1999). In addition to identifying specific biotopes, researchers in Mainz have mapped the distribution of flora and fauna, natural phenomena, and recreational activities within the biotopes (Frey 1998). Similar research by the Polish Academy of Sciences has focused on urban and suburban areas. The research included studies of soils and abiotic ecosystem components, and research by social scientists in a mosaic of habitats with different degrees of development (Zimny 1990). Building upon the foundation of vegetation classification in cities, Brady et al. (1979) proposed a continuum of habitats from the natural to the highly artificial. Dorney (1977), using a similar approach, proposed an urban-rural continuum from a planning perspective and identified six representative land zones—central business district, old subdivisions, new subdivisions, urban construction zones, urban fringe, and rural. Each zone was characterized by three components or subsystems: cultural history, abiotic characteristics, and biotic features. We review the status and implication of such integrated classifications later.

Urban ecology manifest as city planning is contrasted with spatial planning (de Boer & Dijst 1998) in which primary motivations are the degree of segregation or aggregation of different economic and social functions, efficiency of transportation and delivery of utilities, and efficient filling of undeveloped space. Additional components of urban planning said to have ecological foundations include life cycle analysis of products, utility planning based on use rather than medium, efficiency of resource use, exploitation of green infrastructure, and requirements for monitoring of the results (Breuste et al. 1998).

Although the planning described above is ecologically motivated, and it relies on mapping to describe environmental amenities, it is rarely based on data concerning ecological function. It therefore relies on general ecological principles and assumptions, and on the success of prior case histories (Flores et al. 1997). The insights of urban ecology as planning are summarized in manuals and codified in zoning and planning practice. However, like other environmental practices, these insights may not be applicable in novel ecological circumstances. Given the changing forms of cities in both Europe and the United States, novel ecological circumstances may be in the offing.

An Integrated Framework for Urban Ecological Studies

We see three opportunities for improving the theory to understand urban systems. First, rather than modeling human systems and biogeophysical systems separately, understanding will be improved by using integrated frameworks that deal with social and biogeophysical processes on an equal footing (Groffman & Likens 1994). Second, knowing that the spatial structure of biogeochemical systems can be significant for their function (Pickett et al. 2000), we hypothesize that the spatial heterogeneity so obvious in urban systems also has ecological significance (Fig. 1). Third, insights from hierarchy theory can organize both the spatial models of urban systems and the structure of the integrated theory developed to comprehend them. We explore and combine these themes below.

Social Ecology and Social Differentiation

The study of social structures, and how those structures come to exist, are the key social phenomena to support integrated study of the ecology of urban systems. It is increasingly difficult to determine where biological ecology ends and social ecology begins (Golley 1993). Indeed, the distinction between the two has diminished through the convergence of related concepts, theories, and methods in the biological, behavioral, and social sciences. Social ecology is a life science focusing on the ecology of various social species such as ants, wolves, or orangutans. We may also study *Homo sapiens* as an individual social species or comparatively with the ecology of other social species. The subject matter of social ecology, like that of biological ecology, is stochastic, historic, and hierarchical (Grove & Burch 1997). In other words, living systems are not deterministic; they exhibit historical contingencies that cannot be predicted from physical laws alone (Botkin 1990, Pickett et al. 1994).

The underlying basis for this life science approach to the study of human ecological systems depends upon three points (Grove & Burch 1997):

Fig. 1 Patchiness in the Rognel Heights neighborhood of Baltimore, Maryland. The spatial heterogeneity of urban systems presents a rich substrate for integrating ecological, socioeconomic, and physical patterns using Geographic Information Systems. The patch mosaic discriminates patches based on the structures of vegetation and the built environment. The base image is a false color infrared orthorectified photo from October 1999. (M.L. Cadenasso, S.T.A Pickett & W.C. Zipperer, unpublished data.)

1. *Homo sapiens*, like all other species, are not exempt from physical, chemical, or biological processes. Biophysical and social characteristics of humans are shaped by evolution and, at the same time, shape the environment in which *Homo sapiens* live;
2. *Homo sapiens*, like some other species, exhibit social behavior and culture; and
3. Social and cultural traits are involved fundamentally in the adaptation of social species to environmental conditions.

Human ecology must reconcile social and biological facts to understand the behavior of *Homo sapiens* over time (Machlis et al. 1997). Such a biosocial approach to human ecological systems (Burch 1988, Field & Burch 1988, Machlis et al. 1997) stands in contrast to a more traditional geographic or social approach (see Hawley 1950, Catton 1994). This is not to say that social sciences such as psychology, geography, anthropology, sociology, economics, and political science are not important to social ecology. They are, because the most fundamental trait that distinguishes humans and their evolutionary history from other species—both social and nonsocial—is that human social development has enabled the species to escape local ecosystem limitations so that local ecosystems no longer regulate human population size, structure, or genetic diversity (Diamond 1997). Nowhere is this more apparent than in urban ecosystems.

One of the major tools for integration between social and biogeophysical sciences is in the phenomenon of social differentiation. All social species are characterized by patterns and processes of social differentiation (van den Berghe 1975). In the case of humans, social differentiation or social morphology has been a central focus of sociology since its inception (Grusky 1994). In particular, social scientists have used concepts of social identity (i.e., age, gender, class, caste, and clan) and social hierarchies to study how and why human societies become differentiated (Burch & DeLuca 1984, Machlis et al. 1997).

Social differentiation is important for human ecological systems because it affects the allocation of critical resources, including natural, socioeconomic, and cultural resources. In essence, social differentiation determines “who gets what, when, how and why” (Lenski 1966, Parker & Burch 1992). Being rarely equitable, this allocation of critical resources results in rank hierarchies. Unequal access to and control over critical resources is a consistent fact within and between households, communities, regions, nations, and societies (Machlis et al. 1997). Five types of sociocultural hierarchies are critical to patterns and processes of human ecological systems: wealth, power, status, knowledge, and territory (Burch & DeLuca 1984). Wealth is access to and control over material resources in the form of natural resources, capital, or credit. Power is the ability to alter others’ behavior through explicit or implicit coercion (Wrong 1988). The powerful have access to resources that are denied the powerless. One example is politicians who make land-use decisions or provide services for specific constituents at the expense of others. Status is access to honor and prestige and the relative position of an individual (or group) in an informal hierarchy of social worth (Lenski 1966). Status is distributed unequally, even within small communities, but high-status individuals may not necessarily have access to either wealth or power. For instance, a minister or an imam may be respected and influential in a community even though he or she is neither wealthy nor has the ability to coerce other people’s behavior. Knowledge is access to or control over specialized types of information, such as technical, scientific, and religious. Not everyone within a social system has equal access to all types of information. Knowledge often provides advantages in terms of access to and control over the critical resources and services of social institutions. Finally, territory is access to and control over critical resources through formal and informal property rights (Burch et al. 1972, Bromley 1991).

Social differentiation of human ecological systems has a spatial dimension characterized by patterns of territoriality and heterogeneity (Morrill 1974, Burch 1988). As Burch (1988) noted, “Intimate and distant social relations, high and low social classes, favored and despised ethnic, occupational, and caste groupings all have assigned and clearly regulated measures as to when and where those relations should and should not occur.” When ecosystem and landscape approaches are combined, the research changes from a question of “who gets what, when, how and why?” to a question of “who gets what, when, how, why and where?” and, subsequently, what are the reciprocal relationships between spatial patterns and sociocultural and biophysical patterns and processes of a given area (Grove 1997)?

Various processes of social differentiation occur at different scales and have corresponding spatial patterns and biophysical effects (Grove & Hohmann 1992). Based on existing social and ecological theory, examples include global and regional urban-rural hierarchies (Morrill 1974), the distribution of land uses within urban areas (Guest 1977), the stratification of communities within residential land uses (Logan & Molotch 1987), and the social differentiation of ownerships and households within communities (Burch & Grove 1993, Grove 1995).

Spatial Heterogeneity

A human landscape approach may be understood as the study of the reciprocal relationships between patterns of spatial heterogeneity and sociocultural and biophysical processes. Further, when human ecosystem and landscape approaches are combined, human ecosystem types are defined as homogeneous areas for a specified set of sociocultural and biophysical variables within a landscape. Analyses then focus on two primary issues: 1. the development and dynamics of spatial heterogeneity, and 2. the influences of spatial patterns on cycles and fluxes of critical ecosystem resources (e.g., energy, materials, nutrients, genetic and nongenetic information, population, labor, capital, organizations, beliefs, or myths). For instance, the development and dynamics of heterogeneity in a watershed spanning urban to rural conditions may influence and be influenced by sociocultural and biophysical processes. Patches within the watershed may function as either sources or sinks as well as to regulate flows and cycles of critical resources between other patches. The delineation and classification of these relatively homogeneous patches is based on a limited number of representative sociocultural and biophysical indicators (Burch & DeLuca 1984, Parker & Burch 1992), and the patches are studied as black boxes with fluxes and cycles of critical resources between areas (Zonneveld 1989). The spatial linkages between the social and ecological differentiation of the watershed and the relationship of the linkages to different types of allocation mechanisms at different scales are important for understanding the flows and cycles of critical resources within the watershed.

Hydrologists have recognized mosaics of spatial heterogeneity in the variable source area (VSA) approach. They examine how the abiotic attributes of different patches within a watershed–such as temperature and physical characteristics including topography, soil properties, water table depth, and antecedent soil moisture–contribute variable amounts of water and nutrients to streamflow, depending upon their spatial location in the watershed (Black 1991). This VSA approach can be integrated with a delineation of patches based upon the biotic attributes of the watershed, such as vegetation structure and species composition (Bormann & Likens 1979), and the social attributes of the watershed, such as indirect effects from land-use change and forest/vegetation management, and direct effects from inputs of fertilizers, pesticides, and toxins, to examine how the abiotic, biotic, and social attributes of different patches within a watershed contribute variable amounts of water and nutrients to streamflow, depending upon their spatial location in the watershed (Grove 1996). This integrated VSA approach combines nested hierarchies of land use and land cover, sociopolitical structures, and watershed heterogeneity (Fig. 2). GIS is a useful tool for analyzing nested hierarchies of spatial heterogeneity.

VSA approaches can be linked to additional social processes. For example, catchments can be examined via hydrological, land-use, and economic models (Costanza et al. 1990). The three components can be combined into an ecosystem model composed of grid cells within a catchment. The integrated model is built up from a basic model that has hydrologic and ecologic components, but no economic components. Therefore, in the basic model, human behavior causing land-use change must be considered as a factor external to the focal catchment. To construct truly integrated

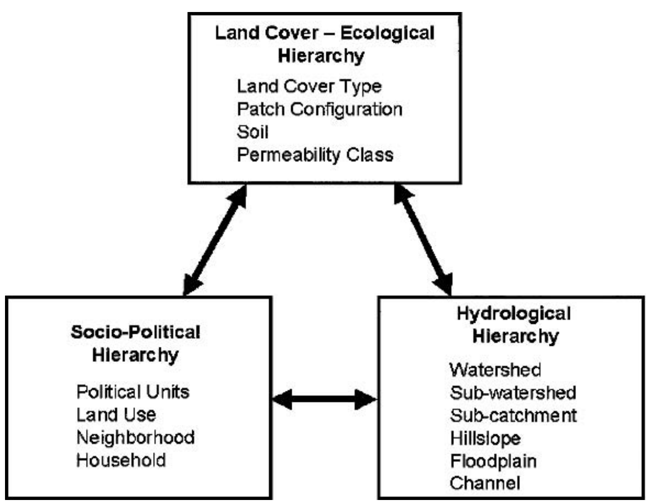

Fig. 2 Three interacting nested hierarchies of spatial heterogeneity representing key disciplinary perspectives used in modeling watershed function in urban areas. Land cover represents nested ecological structures that control interception and runoff. The sociopolitical hierarchy contains nested units of environmental decision making and resource use. The hydrological hierarchy indicates the nesting of spatially differentiated units connected by runoff and runon dynamics

ecological-economic models, major nutrient, water, productivity, and successional components of the basic model must be combined with land-use and economic valuations (Bockstael et al. 1994). The integrated model calculates land-use designation through a habitat-switching module that determines when, through natural succession or weather-driven ecological catastrophe (e.g., floods), the habitat shifts from one type to another. Hypothetical human-caused land-use changes can be imposed exogenously using the integrated model. Recognizing that the ecological effects of human activity are driven by the choices people make concerning stocks of natural capital, the economic modeling uses an understanding of how land-use decisions are made by individuals and how they are based on both the ecological and economic features of the landscape. Again, GIS serves as a tool for manipulating the spatial data on which the model depends, and for assessing the model output over space.

The Human Ecosystem Framework and Urban Ecological Systems

An integrated framework for analyzing urban systems as social, biological, and physical complexes now emerges. Social scientists have focused on interactions between humans and their environments since the self-conscious origins of their disciplines. However, the explicit incorporation of the ecosystem concept within the social sciences dates to Duncan's (1961, 1964) articles "From Social System to Ecosystem" and "Social Organization and the Ecosystem." Recently, the social sciences have focused increasingly on the ecosystem concept because it has been proposed and used as an organizing approach for natural resource policy and management (Rebele 1994).

The ecosystem concept and its application to humans is particularly important because of its utility as a framework for integrating the physical, biological, and social sciences. The ecosystem concept owes its origin to Tansley (1935), who noted that ecosystems can be of any size, as long as the concern is with the interaction of organisms and their environment in a specified area. Further, the boundaries of an ecosystem are drawn to answer a particular question. Thus, there is no set scale or way to bound an ecosystem. Rather, the choice of scale and boundary for defining any ecosystem depends upon the question asked and is the choice of the investigator. In addition, each investigator may place more or less emphasis on the chemical transformations and pools of materials drawn on or created by organisms; or on the flow, assimilation, and dissipation of biologically metabolizable energy; or on the role of individual species or groups of species on flows and stocks of energy and matter. The fact that there is so much choice in the scales and boundaries of ecosystems, and how to study and relate the processes within them, indicates the profound degree to which the ecosystem represents a research approach rather than a fixed scale or type of analysis.

Although the ecosystem concept is flexible enough to account for humans and their institutions (Tansley 1935, Rebele 1994), the application of an ecosystem approach to the study of human ecosystems requires additional analytical components. The analytical framework (Fig. 3) we use here (see Burch & DeLuca 1984, Machlis et al. 1997, Pickett et al. 1997) is not itself a theory. As Machlis et al. (1997: p. 23) noted,

"This human ecosystem model is neither an oversimplification nor caricature of the complexity underlying all types of human ecosystems in the world. Parts of the model are orthodox to specific disciplines and not new. Other portions of the model are less commonplace—myths as a cultural resource, justice as a critical institution. Yet we believe that this model is a reasonably coherent whole and a useful organizing concept for the study of human ecosystems as a life science."

Several elements are critical to the successful application of this framework. First, it is important to recognize that the primary drivers of human ecosystem dynamics are both biophysical and social. Second, there is no single determining driver of anthropogenic ecosystems. Third, the relative significance of drivers may vary over time. Fourth, components of this framework need to be examined

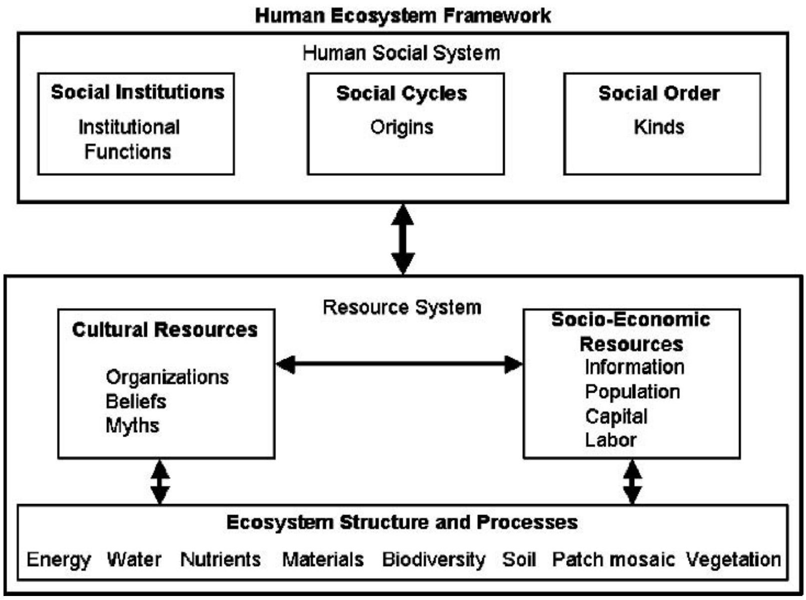

Fig. 3 A human ecosystem framework for integrating biogeophysical and social structures and processes. This conceptual structure shows the most general components of any ecosystem that includes or is affected by humans. The nesting of the boxes indicates the inclusion of specific structures or processes within more general phenomena. The social system contains social institutions which function for provision of government, administration of justice, delivery of health services, provision of sustenance, etc. Social order is determined by factors of individual and group identity, formal and informal norms of behavior, and hierarchies that determine allocation of resources. The resource system is founded on bioecological structures and processes that include the standard subjects of ecology texts. The resource system also includes cultural and socioeconomic resources, which interact with bioecological resources in determining the dynamics of the social system. Based loosely on the work of Machlis et al. (1997). Further ecological details added by Pickett et al. (1997)

simultaneously in relationship to each other (Machlis et al. 1997). Finally, researchers need to examine how dynamic biological and social allocation mechanisms such as ecological constraints, economic exchange, authority, tradition, and knowledge affect the distribution of critical resources including energy, materials, nutrients, population, genetic and nongenetic information, labor, capital, organizations, beliefs, and myths within any human ecosystem (Parker & Burch 1992).

Conclusions

Although there is a wealth of information on the terrestrial components of urban ecological sytstems, much of it is organized from the perspective of ecology in cities. This perspective stands in contrast to a more comprehensive perspective identified as the ecology of cities (Grimm et al. 2000). Studies of ecology in cities have exposed the environmental stresses, subsidies, and constraints that affect urban biota and have documented that the biotic components of metropolitan areas have considerable predictability. In addition, the capacity of certain organisms to adapt to urban environments results in characteristic assemblages.

An alternative approach to urban ecology exists in landscape architecture and planning. This professional practice is motivated by a desire to incorporate ecological principles, to make environmental amenities available to metropolitan residents, and to decrease the negative impacts of urban resource demand and waste on environments elsewhere. Although floristic and faunistic descriptions from urban sites are frequently used in design and planning, there are few data available on ecological functions in cities that can inform such practice. Furthermore, the rapidly changing spatial forms of urban growth and change, and the complex of environmental factors that interact in and around cities, make simple environmental extrapolations risky. Although most of the urban ecological research that has been motivated by planning is of the sort that can be labeled ecology in cities, the field often takes a more comprehensive approach that expresses the ideal of ecology of cities.

In basic ecological research the ecology of the city was first addressed by budgetary studies. Classically this approach has been informed by a biogeochemical perspective based on closed, homeostatic systems. Material and energy budgets of urban systems have been estimated under this rubric. Of course the budgetary approach works whether systems are closed or not, or externally regulated or not. However, assumptions about spatial uniformity and that social agents are external to the ecological processes are questionable.

The distinct ecological approaches classically applied to urban research, and the parallel planning approach, point to the need for integration of different disciplinary perspectives. We have presented a framework that uses middle level theories from ecology and social sciences to identify key factors that should govern the structure and function of biotic, abiotic, and socioeconomic processes in and around cities. This framework is identified as a human ecosystem model, in which both social and ecological processes are integral. Furthermore, the spatial heterogeneity in the biogeophysical and social components of urban systems can be portrayed as patch dynamics. Because patch dynamics can be addressed over many nested scales, structure-function relationships in the human ecosystem can be examined from household to region. The integrative tools and existing data prepare urban ecological studies to continue to benefit from the nature in cities approach, as well as to expoit the contemporary concerns of ecology with spatially heterogeneous, adaptive, self-organizing networks of entire metropolitan systems.

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Integrated Approaches to Long-Term Studies of Urban Ecological Systems*

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In 1935, Arthur Tansley wrote:

We cannot confine ourselves to the so-called "natural" entities and ignore the processes and expressions of vegetation now so abundantly provided by man. Such a course is not scientifically sound, because scientific analysis must penetrate beneath the forms of the "natural" entities, and it is not practically useful because ecology must be applied to conditions brought about by human activity. The "natural" entities and the anthropogenic derivates alike must be analyzed in terms of the most appropriate concepts we can find.
(Tansley 1935, p. 304)

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This quote captures the spirit of the new urban emphasis in the US Long-Term Ecological Research (LTER) network. We know now that Earth abounds with both subtle and pronounced evidence of the influence of people on natural ecosystems (Russell 1993, Turner and Meyer 1993). Arguably, cities are the most human dominated of all ecosystems. Recent calls for studies on "human-dominated ecosystems" (Vitousek et al. 1997) finally have been heeded, over 60 years after Tansley penned his warning, with the addition of two metropolises (Phoenix and Baltimore) to the LTER network.

In this article, we describe an emerging approach to understanding the ecology of urban areas by contrasting these two metropolises, and we present a call to action for ecologists to integrate their science with that of social scientists to achieve a more realistic and useful understanding of the natural world in general and its ecology in particular (Pickett and McDonnell 1993, Ehrlich 1997). We begin by framing a conceptual basis for the study of urban ecological systems: the rationale, contrasting approaches, and special considerations for including human interactions at different scales and in a spatial context. We then discuss the application of our conceptual approach by comparing site conditions and initial research results in Baltimore and Phoenix. We conclude with a summary and synthesis of implications for the integration of social and ecological sciences.

The Conceptual Basis for Studying Urban Ecological Systems

Why has the study of urban ecological systems attracted so much recent interest? The rationale for the study of human-dominated systems is three-pronged. First, humans dominate Earth's ecosystems (Groffman and Likens 1994, Botsford et al. 1997, Chapin et al. 1997, Matson et al. 1997, Noble and

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* Urban ecological systems present multiple challenges to ecologists—pervasive human impact and extreme heterogeneity of cities, and the need to integrate social and ecological approaches, concepts, and theory

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Dirzo 1997, Vitousek et al. 1997); therefore, humans must be integrated into models for a complete understanding of extant ecological systems. Second, development of these more realistic models for ecological systems will lead to greater success in finding solutions to environmental problems (Grossmann 1993). Third, although the study of ecological phenomena in urban environments is not a new area of science, the concept of city as ecosystem is relatively new for the field of ecology. Studying cities as ecosystems within new paradigms of ecosystem science (Pickett et al. 1992, Wu and Loucks 1995, Flores et al. 1997) will both raise the collective consciousness of ecologists regarding urban ecosystems and contribute to the further development of concepts that apply to all ecosystems.

Evidence that human influence on the environment has been pervasive for thousands of years has been accumulating from anthropological and archaeological research (Turner et al. 1990, Redman 1999). Major human impacts on the environment probably go back as far as 12,000–15,000 years ago, when Siberian hunters first entered the Americas. These hunters may have played an important role in the extinction of many species of large mammals (Morgan 1993, MacLeish 1994, Stringer and McKie 1996). The introduction of agriculture transformed human–environmental relations in virtually all parts of the world, leading to, in addition to the obvious benefits, localized but intense episodes of deforestation, soil erosion, disease, and regionwide degradation of vegetative cover long before the modern era (for the Mediterranean, Butzer 1996; for Central America, Rice 1996). Among the more severe human-induced environmental impacts are those associated with ancient urban societies, whose dense populations, rising rates of consumption, and agricultural intensification led to regional degradation so extreme that cities were abandoned and the productive potential of entire civilizations was undermined to the point of ruin. Archaeology has documented repeated examples of such impacts, including the salinization of southern Mesopotamia 4000 years ago (Redman 1992), valleywide erosion in ancient Greece (van Andel et al. 1990), and almost complete depopulation of large tracts of Guatemala (Rice 1996) and highland Mexico (O'Hara et al. 1993) from 1000 AD to 1400 AD. Clearly, human actions dramatically alter the functioning of ecosystems of which humans are a part, and, equally clearly, humans are a part of virtually all ecosystems and have been so for millennia. Nowhere has this human participation been more intense than in cities, suburbs, and exurbs and in the supporting hinterlands.

Today, urbanization is a dominant demographic trend and an important component of land-transformation processes worldwide. Slightly less than half of the world's population now resides in cities, but this proportion is projected to rise to 61% in the next 30 years (UN 1997a). The developed nations have more highly urbanized populations; for example, close to 80% of the US population is urban. However, projections for the twenty-first century indicate that the largest cities, and the largest growth in city size, will occur in developing nations. Between 1980 and 2030, the percentage of the urban population on the African continent will double from 27% to 54% (UN 1997a). Urbanization trends of the past century also show a dramatic rise in the size of cities: Over 300 cities have more than 1 million inhabitants, and 16 "megacities" have populations exceeding 10 million (UN 1997b). Urbanization interacts with global change in important ways. For example, although urban areas account for only 2% of Earth's land surface, they produce 78% of greenhouse gases, thus contributing to global climate change. Cities also play a central role in alteration of global biogeochemical cycles, changes in biodiversity due to habitat fragmentation and exotic species, and changes in land use and cover far beyond the city's boundaries (i.e., within the urban "footprint").

The growing impact of urban areas on the face of the earth is reason enough to study them. An even more compelling argument for understanding how cities work in an ecological sense is the fact that humans live in them and must depend on proper management to maintain an acceptable quality of life for the foreseeable future. Because human societies are an important part of urban ecological systems (and perhaps all ecosystems; McDonnell et al. 1993), ecologists now recognize that "most aspects of the structure and functioning of Earth's ecosystems cannot be understood without accounting for the strong, often dominant influence of humanity" (Vitousek et al. 1997, p. 494).

To understand human actions and influences on ecosystems, it is essential to use approaches developed in the social, behavioral, and economic sciences. Conceptual frameworks that explicitly include humans will be much more likely than those that exclude them to accurately inform environmental problem solving. The reasons for this contention are obvious: Human perception, choice, and action are often the phenomena that drive political, economic, or cultural decisions that lead to or respond to change in ecological systems.

What can the integration of social and ecological sciences as applied to human-dominated ecosystems bring to the field of ecology? One of the main goals of the LTER program is to understand the long-term dynamics of ecosystems. Although approaches, types of ecosystems, and disciplinary expertise often differ among the sites, conceptual similarities in many ways overshadow these differences. In their common commitment to understanding long-term ecological dynamics, for example, most LTERs recognize two classes of variables affecting ecosystems. The first and better-studied class of variables includes patterns and processes of ecosystems, which are constrained by "natural" factors such as geologic setting, climate and its variation, species pools, hydrologic processes, and other biological or geophysical factors. Underlying this first class of variables are the fundamental drivers of ecological systems: the flows of energy and information and the cycling of matter (by the flow of information we refer primarily to evolutionary origins and change; e.g., Reiners 1986). Understanding how these drivers, and the constraints they impose, interact with ecological patterns and processes to produce long-term dynamics has been a major goal of most LTER programs. The second class of variables are those directly associated with human activities, such as land-use change, introduction or domestication of species, consumption of resources, and production of wastes.

The simplified model shown in Fig. 1 defines the intellectual arena within which LTER ecologists typically work. The inclusion of both natural processes and human activities influencing long-term ecosystem dynamics is appropriate because even LTER sites in purportedly pristine areas are subject to human-caused disturbance and change. Patterns and processes of ecosystems, where the five core areas (primary production, populations, organic matter, nutrients, and disturbance) of the LTER program are centered, have been more completely specified than have social patterns and processes. It is clear, however, that several important interactions and feedbacks are missing from this approach. Because many of these missing features are the subject matter of the social sciences, it is through contributions in this area that an understanding of the world's ecosystems can be most enhanced (Ehrlich 1997). At the same time, these interactions and feedbacks should not be pursued in isolation within a self-contained, traditional social-scientific framework, because the human activities that influence ecosystem dynamics are reciprocally influenced both by biogeophysical driving forces and by ecosystem dynamics.

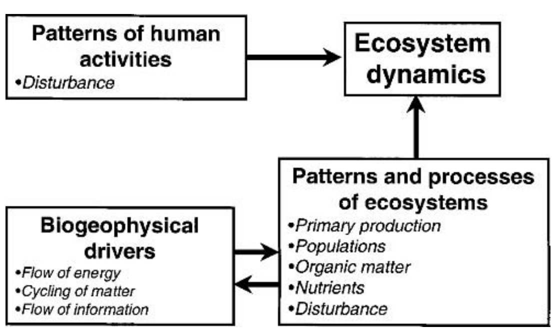

Fig. 1 Two classes of variables that affect ecosystems: patterns and processes of ecosystems and patterns of human activities. Ecologists have mostly studied the former, and have developed theory based on the fundamental biogeophysical drivers that determine ecosystem pattern and process. Items listed under "patterns and processes of ecosystems" are the core areas of long-term ecological research

An example of this reciprocal influence between human activity and ecosystem dynamics is provided by work at the North Temperate Lakes LTER site (Carpenter et al. 1999), one of two LTER programs that have received additional funding to add social science research and regionalization to their programs (the other augmented program is the Coweeta LTER). In Wisconsin, agricultural land uses are linked to eutrophication of the area's lakes, primarily through excess phosphorus inputs. Farmers' use of fertilizers can directly affect soil dynamics, and soil conditions can, in turn, affect a farmer's decision to use fertilizers (see Carpenter et al. 1999). In addition, a wide set of socioeconomic drivers, such as the local or regional economy, can influence human interaction with the natural landscape. In the example of fertilizer use, the market potential of the crops, government subsidies, and even the practices of neighboring farmers can influence a farmer's decision to use fertilizers (see Carpenter et al. 1999). Finally, ecosystem dynamics themselves, as altered by impaired water quality caused by leaching of excess fertilizer-derived nutrients, can influence patterns of socioeconomic activity at a larger scale, including real estate values, industrial relocation, or recreation patterns. Examining these interactions both complicates and enhances the long-term study of an ecosystem.

Without understanding interactions and feedbacks between human and ecological systems, our view of ecosystem dynamics both at local and global scales will be limited—as will be our ability to apply these insights to public policy and land management. Acknowledging the central human component leads to an emphasis on new quantitative methods, new approaches to modeling, new ways to account for risk and value, the need to understand environmental justice, and the importance of working within a globally interacting network (e.g., Grossmann 1993). These added interactions and feedbacks traditionally have been studied by social scientists in isolation from life and earth scientists. Rarely, if ever, has a focused long-term study incorporated all the interactions implicit in Fig. 2. Such integration requires a research team that brings together scientists from the natural, social, and engineering sciences in a unified research endeavor.

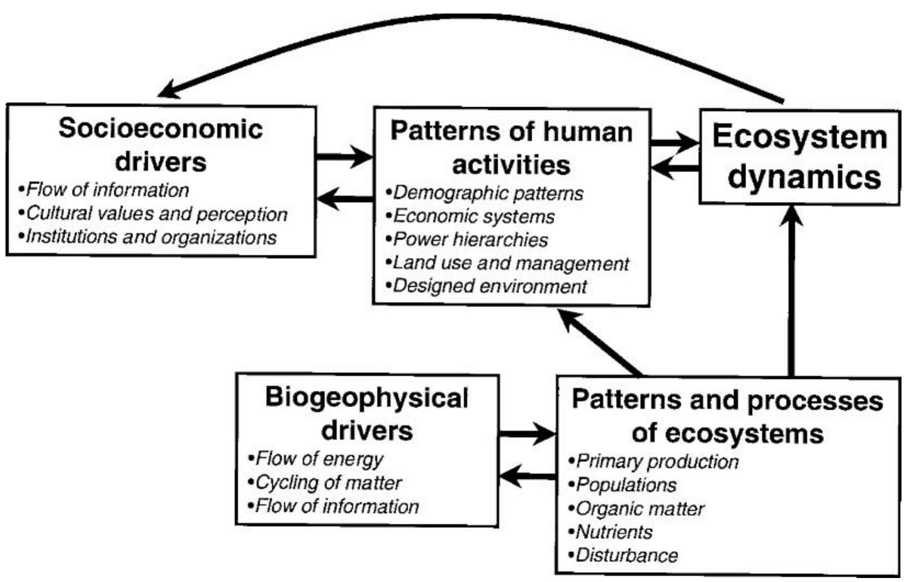

Fig. 2 A more comprehensive view of the drivers, interactions, and feedbacks affecting ecosystem dynamics that recognizes, in addition to biogeophysical drivers, socioeconomic drivers that determine patterns of human activity. Items listed under "patterns of human activities" are suggested core areas of long-term social science research

Ecology in and Ecology of Urban Ecological Systems

The simplest phrase describing the research being done in the Baltimore and Phoenix LTER studies is urban ecology. However, this term is fraught with misunderstanding because it is often applied to research that does not encompass the full suite of concepts we hope to develop. We prefer to distinguish between two different types of research that more accurately describe the projects we envision: ecology in cities and ecology of cities. There are abundant examples in the literature of ecology in cities (e.g., Sukopp and Werner 1982, Sukopp 1990). The basic questions addressed by such studies are, how do ecological patterns and processes differ in cities as compared with other environments? What is the effect of the city (i.e., a concentration of human population and activities) on the ecology of organisms inside and outside of its boundary and influence (e.g., McPherson et al. 1997)? Research topics exemplifying ecology in cities are distribution and abundance of animal and plant populations, air pollution and meteorology, patch-specific ecological pattern and processes, edge effects (because edges are especially pronounced in cities), and exotic–native species interactions. Tools for doing ecology in cities include before–after experiments, which allow the study of effects of rapid changes occurring in the urban environment, and the concept of urban–rural gradients (e.g., McDonnell et al. 1993), which can be a form of space-for-time substitution to detect impacts of urbanization on ecological processes. These examples help illustrate that ecology has been done in cities for a long time. We do not wish to claim that the urban LTERs are the inventors of urban ecological studies! But perhaps much of the uniqueness of the new urban LTER programs lies in their attempt to treat cities as ecosystems, that is, to study the ecology *of* cities.

The concept of ecology of cities has to do with how the aggregated parts sum, that is, how cities process energy or matter relative to their surroundings. Ecologists might take many different routes to understanding the ecology of cities: mass balances of nutrients for the entire system, patch dynamics (wherein all patch types, and the changes in and among them, are considered for the landscape as a whole), ecological effects of land-use change, whole-system metabolism, spatial distribution of resources and populations (e.g., a metapopulation approach), and estimation of the ecological footprint of a city (Rees and Wackernagel 1994, Folke et al. 1997). Tools for studying ecology of cities include the watershed approach, wherein measurement of inputs and outputs is simplified because the system is defined as the area of land drained by a particular stream (e.g., Bormann and Likens 1967, Likens and Bormann 1995); patch dynamics modeling; and monitoring and modeling of land-use change by incorporating remote sensing and GIS (geographic information systems) methodologies. Of course, each approach borrows from the other, and with increasing scale what is viewed as the ecology-of-cities approach may become ecology in an urban patch within the larger region of which the city is part. Our intent is not to advocate one approach over the other, but we do agree with McDonnell and Pickett (1990, p. 1231), who stated nearly a decade ago that "study of the [city] as an ecosystem . . .would be a radical expansion of ecology." Thus, both the Baltimore and Phoenix LTER programs are doing ecology in cities but are framing their work within the context of city as ecosystem.

An ecosystem is a piece of earth of any size that contains interacting biotic and abiotic elements and that interacts with its surroundings. By this definition, and with Tansley's purpose and definition of the term in mind—"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" (Tansley 1935, p. 299)—a city is most certainly an ecosystem. But few studies have treated cities as such (a notable exception is the study of Hong Kong by Boyden et al. 1981; see also Boyle et al. 1997). Ecosystems have definable structure and function. Structure refers to the component parts of the system: organismal (including human) populations, landscape patches, soils and geologic parent material, and local atmospheric and hydrologic systems. Ecosystem function is a general term referring to the suite of processes, such as primary production, ecosystem respiration, biogeochemical transformations, information transfer, and material transport, that occur within ecosystems and link the structural components. The function of a whole ecosystem or a part of an ecosystem can be thought of as an integrated measure of what that unit does in the context of its surroundings. For example, a component ecosystem in a region contributes stocks of resources or fluxes of materials to that larger region. More specifically, a city might contribute airborne particulates or nitrogenous or sulfurous gases to ecosystems located downwind of it.

In addition to the structure, function, and processes traditionally studied by ecologists in any ecosystem, urban systems also contain the dominant components of social institutions, culture and behavior, and the built environment. An ecology-in-cities study that incorporates these components might consider, for example, how irrigation practices or creation of hydrologic infrastructure (e.g., canals, pipes, or storm drains) influences the distribution of insects on household or neighborhood scales. An ecology-of-cities approach might include models of urban growth and spread that reflect economic and social drivers; for example, the tendency of people to want to live on hillslopes (behavior) or the market value of housing near transportation routes (economics). This type of study must necessarily involve the reconceptualization of human activities, not as disturbances to the ecosystem but as important drivers of and limitations to it (e.g., Padoch 1993). Traditional scholarship of urban systems in the human–social, ecological, geophysical, and civil infrastructural domains has been pursued in relative isolation, with each developing its own disciplines, methodologies, and evaluation tools (Borden 1993). Urban ecosystem studies can bring elements of these disparate approaches together, underscoring the interdependence of these phenomena.

Traditional biological or earth science–based approaches to ecosystem studies are insufficient for urban systems because of the interaction of social systems with biogeophysical systems (Borden 1993). Although some have argued that humans can be treated as just another animal population, albeit an important one (but see Padoch 1993), the suite of social drivers of urban ecosystems—information flow, culture, and institutions—are not easily modeled within a traditional population framework (Padoch 1993, Turner and Meyer 1993). Several modifications of this framework are necessary to successfully integrate human activity into an ecological model. The first is to acknowledge the primary importance of human decision-making in the dynamics of the urban ecosystem. This decision-making operates within a broad context of culture, information, and institutions. This modification puts an appropriate emphasis on the differential creation, flow, and control of information within the human ecosystem. Culture, the learned patterns of behavior for each particular society or group, and institutions, the formal structures that codify patterns of behavior, also are central components of decision-making and thus are key to understanding environmentally relevant decisions.

State variables of urban ecosystems must include more than measures of population size, species diversity, and energy flow. They must also include measures of state as perceived by humans, often referred to as "quality of life." Educational opportunities, cultural resources, recreation, wealth, aesthetics, and community health all are factors that may differ among cities, yet these variables have few parallels in traditional ecosystem studies.

Special Considerations for the Human Component of Urban Ecological Systems

Ecologists must ask whether theories developed for ecological systems in the presumed absence of human influence will be appropriate for systems, like cities, where human dominance is unquestionable We suspect that simple modification of ecological theory will prove unsatisfactory, because the modifications we have just discussed deal with aspects of human social systems that are far from simple. Although incorporation of existing social science models (Pickett et al. 1997) into ecological theory provides a starting point, development of a new integrative ecology that explicitly incorporates human decisions, culture, institutions, and economic systems will ultimately be needed (James Collins, Ann Kinzig, Nancy Grimm, William Fagan, Diane Hope, Jianguo Wu, and Elizabeth Borer, unpublished manuscript). At the same time, it is incumbent upon social scientists who hope for a realistic understanding of the urban system to consider biogeophysical feedbacks and interactions with the ecological system. The suite of social system components that we plan to include in our new models is listed in the box on page 576, although not all of these components may be relevant to any particular model or process.

Social system components

The following are examples of social system components to be incorporated into human–ecological models of urban ecosystems (from Pickett et al. 1997).

Social institutions
- Health
- Justice
- Faith
- Commerce
- Education
- Leisure
- Government

Social dynamics
- Physiological
- Individual
- Organizational
- Institutional
- Environmental

Social order
- Age
- Gender
- Class
- Norms
- Wealth
- Power
- Status
- Knowledge
- Territory

Social resources
- Economic (information, population, labor)
- Cultural (organizations, beliefs, myths)

Within the context of the LTER program, the subject matter for investigation started with a set of five core areas that are common to all LTER research: primary production, populations representing trophic structure, organic matter storage and dynamics, nutrient transport and dynamics, and disturbance. Given, as we have suggested (Figure 1), that three fundamental biogeophysical drivers of ecosystems are the flow of energy, the flow of information, and the cycling of materials, what are the comparable drivers and core areas of social systems? Social scientists working at the augmented (Coweeta and North Temperate Lakes) and urban LTER sites consider individuals to be guided in their activities by the knowledge, beliefs, values, and social resources shared with other members of their social system at different levels of organization (e.g., at family, community, state, and national levels). In this perspective, the three fundamental drivers of human elements in the ecosystem (Figure 2) are:

- Flow of information and knowledge
- Incorporation of culturally based values and perceptions
- Creation and maintenance of institutions and organizations

These drivers condition human activities and decisions (see also Turner and Meyer 1993). Although understanding the nature and interaction of these drivers must be the ultimate goal of most inquiries, actual investigations may more often be oriented toward measuring the patterns of human activity these processes create. Defining the following core topics—each characterized by its activities, structure, and historic trajectory—is key to a comprehensive approach to human ecosystem analysis (Figure 2):

- Demographic pattern
- Economic system
- Power hierarchy
- Land use and management
- Designed environment

Collectively, these core topics serve as guidelines of inquiry analogous to the five ecological core areas identified early in the LTER program's history. On the one hand, they reflect the processes operating within the system; on the other, they are a practical guide for field investigations. The entire LTER network (not just the urban and augmented sites) thus provides an excellent starting point to incorporate social science research that is relevant to, and integrated with, studies of ecosystem change.

A Conceptual Scheme for Understanding Urban Ecological Systems

We have now identified drivers and patterns of activity in both ecological and social systems that must collectively be considered for a full understanding of human-dominated ecosystems. A more specific conceptual scheme or model for how a study of urban ecological systems can be approached is represented by Figure 3. This scheme has been modified, through recent discussions, from schemes presented by the Baltimore and Phoenix LTER groups in grant proposals and various publications (e.g., Grimm 1997, Pickett et al. 1997); indeed, it is still evolving. The diagram includes a set of variables that are linked by interactions and feedbacks.

The two variables in the upper corners—coarse-scale environmental context and societal patterns and processes—can be viewed as constraints, which are the outcome of operation of the fundamental biogeophysical and societal drivers. The coarser-scale environmental context includes climate, geology, history, and biogeographical setting, and societal patterns and processes encompass the socioeconomic system, culture, demography, and social institutions.

The middle two variables are the focus of the new urban LTER research: land use and ecological patterns and processes associated with any given land use. Land use incorporates both the intent and the reality of how a given parcel within a metropolitan boundary is altered by human decisions, whereas ecological patterns and processes—energy flow; nutrient cycling; the hydrologic cycle; species distribution, abundance, and interactions; ecosystem and landscape structure; and disturbance—are those phenomena studied by all LTER teams. Land use and associated ecological conditions are viewed at a single point in time, using a hierarchical, patch mosaic perspective. Land-use change occurs because of development, urban renewal, changes in land management or ownership, or infrastructure development, among other causes.

The last two variables in the lower corners of Figure 3—changes in ecological conditions and changes in human perceptions and attitudes—result from the interaction between land-use change and ecological pattern and process. Changes in ecological conditions represent the next time step of ecological patterns and processes, and changes in human perceptions and attitudes represent the human reaction to either ecological pattern and process or changes therein, as expressed through the filter of human experience.

The interactions and feedbacks depicted in Figure 3 are intended to reflect temporal dynamics to some extent. As an example, land-use decisions are based on the environmental and politico–socioeconomic context, the land use is then perceived as good or bad, and this feedback can lead to

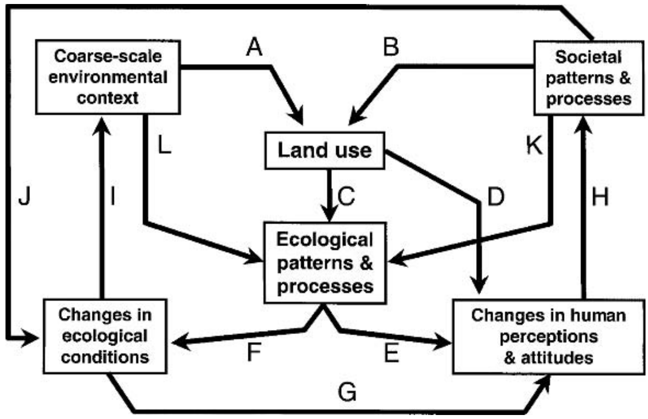

Fig. 3 Conceptual scheme for integrating ecological and social systems in urban environments. Variables are in boxes; interactions and feedbacks are arrows: A, environmental context sets the range of possibilities for land use–land cover; B, societal decisions and human behavior (incorporating their suite of determinants) are the direct drivers of land-use change; C, the pattern of land use (whatever the driver) determines ecological patterns and processes; D, humans perceive and react to land-use change (independent of any ecological effects); E, humans also perceive and react to ecological patterns and processes; F, in this interaction, ecological processes as affected by land-use change result in a change in ecological conditions; G, such changes in ecological conditions may result in changes in attitudes (even if human perception previously ignored ecological pattern and process), and changed ecological conditions are perceived as good or bad by humans; H, changes in perception and attitude feed back to the societal system (patterns and processes of society) to influence decision-making, and this part of the cycle begins anew; I, in some cases, changed ecological conditions can alter the coarse-scale environmental context (example: urban heat island), resulting in a feedback that is relatively independent of human response. J, K: When a societal response to changed ecological conditions is deemed necessary, the society can act directly on the changed conditions (J) or on the underlying ecological patterns and processes producing the problem (K). Finally, the environmental context of course influences ecological patterns independent of land use (L)

different human decisions. This sequence of decisions and changes in land use does not incorporate new ecological information (i.e., the changes in ecological condition that result from the land-use change); such a situation is unstable and seems unlikely to lead to a sustainable urban environment. A sequence of interactions and feedbacks carried out when a change occurs or in response to an environmental problem, however, would incorporate either short-term solutions to those problems or adjustments in management decisions based on a solid ecological foundation.

To illustrate the sequence of changes and feedbacks to further change, consider an example from the Central Arizona–Phoenix LTER site: the establishment of an artificial lake in the once-dry Salt River bed (Figure 4). The initial land-use decision (establishment of the lake) was constrained on the physical–ecological side by the existence of an alluvial channel with no surface water flow (because of upstream impoundments) in a region of North America characterized by a high propensity for flash flooding (Baker 1977). Societal constraints included the economic cost of the project, the perceptions of political and economic benefit, available technology (collapsible dams, recirculating pumps), and the existence of human-created infrastructure (engineered channel, diversion of surface flow into canals). Given these constraints, and based on our best understanding of lake ecology, the ecological conditions associated with the lake when it was filled were likely to be high nutrients with concomitant high algal production, high rates of infiltration, and a high probability of floods. At the next time step, we expected such changes in ecological conditions as eutrophication, losses of water to the groundwater system, and establishment of a robust mosquito population. Preliminary monitoring confirms the predictions of high phytoplankton biomass and insect populations, and a summer flash flood resulted in a brief episode of high phosphorus loading (Amalfi 1999).

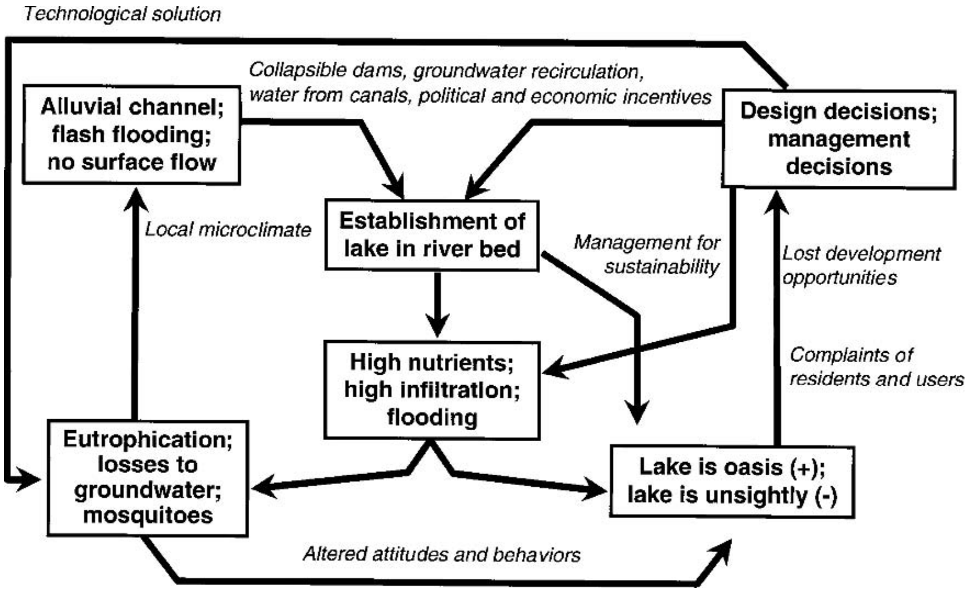

Fig. 4 Conceptual scheme adapted to show the interaction among physical, ecological, engineering, social, and management variables and drivers in the new Tempe Town Lake, Arizona

The predicted sequence of ecological conditions may result in a full range of reactions from the public. Feedbacks to the societal patterns and processes—in this case, the design and management decisions—could lead either to chemical additions to control algal blooms or to more ecologically based management practices, such as control of nutrient inputs. Indeed, complaints from the public about insect populations already have resulted in implementation of a chemical control program (Amalfi 1999), and feedbacks of ecological information and knowledge of the lake's dynamics have helped create and modify a lake management plan. Thus, societal responses can act directly on the changed conditions (arrow J in Fig. 3; e.g., addition of chemical control agents, the technological solution in Fig. 4) or indirectly, via an effect on the underlying ecological patterns and processes producing the problem (arrow K in Figure 3; e.g., diversion of upstream point-source nutrient inputs to the water supply, the arrow labeled management for sustainability in Fig. 4). We believe that this general scheme (Fig. 3) allows us to think about how social and ecological systems interact in urban areas. In addition to addressing fundamental ecological processes, this approach allows one to make predictions that are testable, and it also provides tools that citizens and decision-makers can use to create more ecologically sound policy.

A modeling framework for investigating cities

The conceptual insights of the broad scheme in Figure 3 illustrate in general how processes from the social realm, biological environment, civil infrastructure, and the larger climatic and geological context can be integrated with the specific characteristics of metropolitan Phoenix and metropolitan Baltimore. But the extreme patchiness of urban environments dictates attention to spatial detail, using novel approaches of landscape analysis and modeling. Although the two cities are quite different, integration is furthered by a common approach to spatial analyses: a hierarchical patch dynamics approach (Wu and Loucks 1995, Pickett et al. 1999). Hierarchical patch dynamics models start with ecological processes associated with fundamental units of landscapes at some specific scale, called patches. They then address ecological processes that are associated with the patches. The structure of the patches can be a major determinant of those processes. However, patch structure and arrangement also can change through time. Hence, models must account for such change; that is, they must be dynamic. Furthermore, because patches at a particular scale are often themselves composed of smaller patches, and can be aggregated into larger patches, the models must be hierarchical. By considering patch dynamics simultaneously at multiple scales, with an accompanying hierarchy of models, the complexity of urban systems is rendered more tractable and translation of information across scales is facilitated (Wu 1999).

Special Characteristics that Dictate a Novel Framework

The modern metropolis presents a strikingly heterogeneous pattern for study. For instance, the sharp contrasts between neighborhoods is a familiar characteristic of cities (Clay 1973). Within the span of a city block, which is on the order of 200 or fewer meters, an observer may cross several obvious boundaries. Different kinds of commercial use, shifts between owner-occupied and rental properties, and shifts in socioeconomic resources available to residents are but some of the many contrasts cities present. Such heterogeneity is not unique to dense, central urban districts. In fact, the contemporary suburb is zoned for even more discrete transitions than the traditional mixed use of older cities. Residential streets, feeder streets, commercial streets, strip malls, regional malls, and industrial parks are notable patches in the suburban landscape. Of course, the scale of transition in post–World War II suburbs tends to be coarser than that of older neighborhoods and districts because of the shift to dependence on the automobile. However, spatial patchiness in the social, economic, and infrastructural fabric of metropolitan areas remains their most obvious feature. Social scientists have long been aware of the functional significance of spatial heterogeneity and mixture of uses within urban areas (Jacobs 1962). The ecological significance of such socioeconomic heterogeneity is, however, an open question. Indeed, determining to what extent the well-recognized patchiness of urban areas has ecological dimensions and ecological implications is one of the main motivations of integrated, long-term ecological research in the metropolis.

Whatever its ecological significance, the conspicuous spatial heterogeneity in urban systems is an entry point for integration with social science. The existence of such clearly defined patches as neighborhoods and cityscapes, which combine infrastructural and natural features, is apparent to all researchers who must work together to generate the interdisciplinary synthesis for understanding cities as ecological systems. Hydrologists, ecologists, demographers, economists, engineers, and citizens all can and do recognize the spatial heterogeneity of cities. Neighborhood associations, watershed associations, census tracts, and similar groupings are institutional expressions of this common recognition. Of course, each discipline or constituency may see the boundaries or the most salient features of the patchwork somewhat differently. For example, the civil engineer and the urban recreationist will have different views of the boundaries of a watershed. The first may see a "sewershed"; the second, a visually unified landscape that is engaging on a morning jog. Therefore, new, multidimensional classifications of the heterogeneity of the metropolis are required. Among the principal dimensions of such classifications, however, will be factors that control the flow of materials, energy, and information through and within the metropolis. An emphasis on these kinds of variables suggests that a spatially explicit, ecosystem perspective can emerge from the heterogeneity of the metropolis.

The interdisciplinary integration required to understand the ecological significance of spatial pattern in urban ecosystems must account for several important features of humans and their institutions. Although these features are implied in the social drivers and phenomena we introduced earlier (Figure 2, see box page 129), these features add complexity to ecological studies of the city and, therefore, cannot be ignored. The spatial heterogeneity of urban systems is established by formal institutions, such as zoning regulations, and maintained by other formal institutions, such as public works and the courts. However, less formal institutions, such as families and community associations, also contribute to the spatial structure and its function. Humans, as individuals and groups, are self-aware, capable of learning quickly, and engaged in extensive networks of rapid communication. These features of the human components of urban systems mean that the feedback among the biological, human, infrastructural, and the larger physical contexts can be strong and, in many cases, rapid. This is one reason that education has been incorporated into the structure of urban LTER programs. We hypothesize that learning about the heterogeneity and function of an urban area can be a tool that citizens and institutions can demonstrably use to improve their neighborhoods, city, and region through management, planning, and policy (Grove and Burch 1997).

Patch Dynamics and Hierarchical Approaches

Spatial heterogeneity was independently chosen as a starting point by both the Central Arizona–Phoenix and Baltimore Ecosystem Study LTER teams. In addition to the advantages already laid out, the patch dynamics approach also brings the advantage of hierarchical nesting and aggregation. Such a flexible hierarchical approach is important because the structures, and consequently the processes, that govern the function of the metropolis as an ecological system occur at a variety of scales. For example, just because some social processes occur at the scale of neighborhoods of, say, 15 square blocks, does not mean that the most important ecological impacts occur at the same scale. Similarly, the concentration of resources or wastes in particular patches can be due to decisions made at great distances from the point of concentration. Therefore, patches may be scaled up or down for different functional analyses, and the configuration of patches that are relevant to specific paths along which resources and information flow can be assessed.

As an example, Baltimore can be described in terms of the five-county metropolitan area (Figure 5). Within it are three principal watersheds that extend from the rural hinterlands to the central city. Each of these watersheds can be divided into subcatchments. The 17,000-ha Gwynns Falls catchment contains 16 smaller watersheds that have been used for discussion of management and restoration activities. Still smaller units can be used for mechanistic studies of ecosystem and socioeconomic processes. The fact that different nestings are possible within the larger units means that the scales at which important interactions occur can be captured, and the promulgation of their effects to different scales, whether coarser or finer, can be determined (see Grove and Burch 1997).

A similar nested set of units is being identified in the Central Arizona–Phoenix study area (Fig. 5). At the largest scale, three patches exist along an east–west gradient from primarily commercial–industrial–residential, to primarily agricultural, to primarily desert land use-land cover. The political boundaries of the 24 different municipalities in the Phoenix metropolitan area form another set of smaller patches, and heterogeneity of land use-land cover is evident within these smaller units. Interestingly, patch size, regularity, and connectivity differ within the three broadest patches (Matthew Luck and Jianguo Wu, Arizona State University, personal communication).

The patch dynamics approach focuses explicitly not only on the spatial pattern of heterogeneity at a given time but also on how and why the pattern changes through time and on how that pattern affects ecological and social processes. Because cities are both expanding and changing within their boundaries, the dynamic aspect of this approach is crucial to complete understanding of urban ecological systems. Even within a coarse resolution land-cover type, change occurs over time in the resources available for management of biotic structure and maintenance of civil infrastructure and in the requirements and interests of humans in specific patches. The explosive changes in patch structure at the urban fringe of the rapidly expanding Phoenix metropolis is one example of this phenomenon: formerly desert patches are converted to residential housing develop-

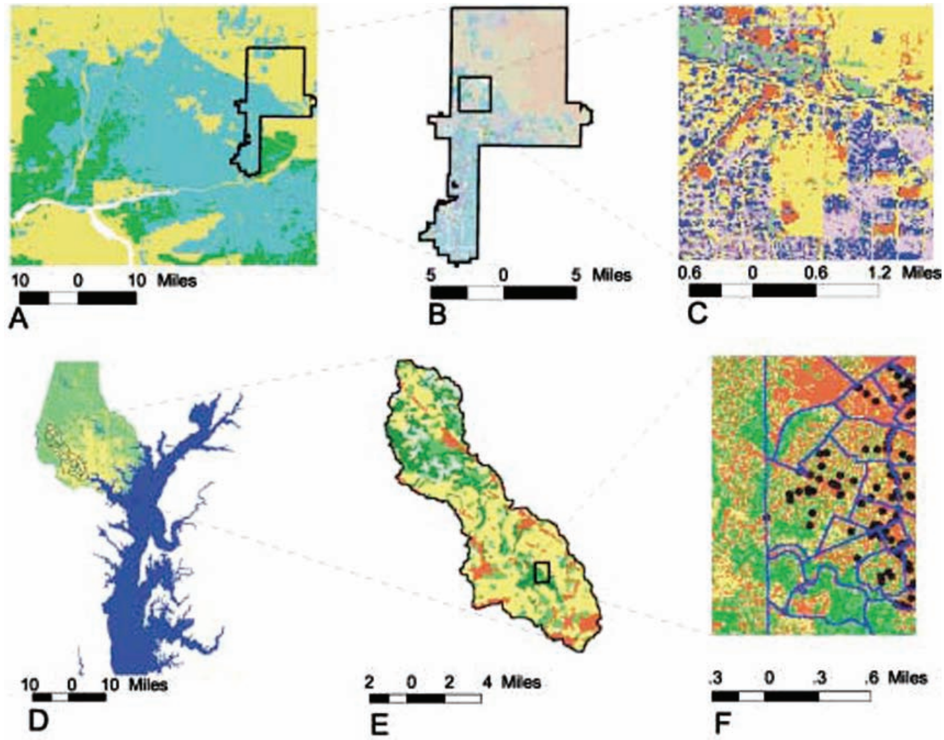

Fig. 5 Examples of hierarchically nested patch structure at three scales in the Central Arizona–Phoenix (CAP; upper panels) and Baltimore Ecosystem Study (BES; lower panels) regions. At the broadest scale (A, D), patches in the CAP study area include desert (mustard), agriculture (green), and urban (blue); for the BES, patches are rural (green), urban (yellow), and aquatic (blue). (B) The municipality of Scottsdale, AZ, showing major areas of urban–residential development (blue, lower portion) and undeveloped open lands (tan, developable; brown, dedicated). (C) Enlargement of rectangle in B showing additional patch structure at a neighborhood scale (green, golf course/park; mustard, undeveloped desert; red, vacant; pink, xeric residential; purple, mesic residential; yellow, asphalt). (E) Gwynn's Falls watershed, MD, with residential (yellow), commercial/industrial (red), agricultural (light green), institutional (medium green), and forest (dark green) patch types. (F) Enlargement of rectangle in E showing additional patch structure at a neighborhood scale (dark green, pervious surface/ canopy cover; light green, pervious surface/no canopy cover; yellow, impervious surface/canopy cover; red, impervious surface/no canopy cover; blue, neighborhood boundaries; black circles, abandoned lots). Panel A courtesy of CAP Historic Land Use Project (caplter.asu.edu/research/contributions/HistoricLandUse_Color.pdf), B and C courtesy of CAP LTER/Geologic Remote Sensing Laboratory (elwood.la.asu.edu/grsl/), D, E, and F courtesy of USDA Forest Service and BES LTER

ments and, with this conversion, demand for amenities of urban life increases sharply. In Baltimore, patch dynamics are conspicuous both at the suburban fringe and in the ever-growing collection of vacant buildings and empty lots within the older, dense urban areas. In both the fringe and core patches, ecological processing of water, nutrients, and the provision of goods and services are not constant in time. The dynamics of patches in and around the city have implications for the ecological processes and status of areas well beyond the city and even beyond the present suburban and exurban areas. The search for open land, development opportunities, and changes in the economics of farming and production forestry all influence and are influenced by urban patch dynamics.

Application of Integration: Comparisons between Baltimore and Phoenix

Having laid out a coarse conceptual scheme for integrating social and ecological principles and a more specific modeling framework that deals with heterogeneity and dynamics of urban ecological systems, we turn next to a consideration of the distinctly different cities chosen for inclusion in the LTER network. In almost every characteristic, Phoenix and Baltimore are dramatically different (Table 1). In terms of the physical environments, Baltimore is an eastern seaboard metropolis straddling the Coastal Plain and Piedmont provinces, whereas Phoenix is an inland city situated in a broad alluvial basin at the confluence of two large desert rivers. Phoenix has a hot, dry climate (receiving less than 200 mm of precipitation annually), whereas Baltimore has a mesic, temperate zone climate (1090 mm of precipitation). Natural hydrological regimes are flashy (rapid rises and falls in flows) in Phoenix, whereas in Baltimore rainfall and runoff are more uniform throughout the year. The transport of water and materials by surface runoff is thus highly episodic in Phoenix, where baseflow exists only in manmade canals and as treated sewage effluent. Baltimore's hydrology and material transport, in contrast, can be studied using standard watershed approaches, although flashiness associated with urbanization dictates a focus on storm flows even there.

Ecological contrasts between Baltimore and Phoenix also are striking: deciduous forest versus desert, high (temperate forest) versus low (desert) biotic diversity, and different disturbances that

Table 1 Contrasting characteristics of Baltimore and Phoenix, two metropolises that are recent additions to the US Long-Term Ecological Research (LTER) network

Characteristic	Baltimore	Phoenix
Climate	Mesic, temperate	Arid, hot
Geographic–geomorphic	Land–sea margin	Alluvial basin
Topography	Coastal plain	Flat, outcrops and buttes, surrounding mountains
Hydrologic characteristics	Seasonal runoff systems	Flashy; episodic surface runoff
Natural vegetation	Eastern deciduous forest	Sonoran desert scrub
Native biotic diversity	High	Low–moderate
Invasibility–impact of exotic biota	Moderate	High
Primary limiting factor (ecological)	Seasonally variable	Water
Natural disturbances	Hurricane	Fire, flash flood
Succession rate	Moderate	Slow
Remnant patches	Forest	Desert
Prehistory	Low density, scattered	Large civilization 1000 years before present
History	Early seaport	Abandoned until 100 years before present
Age of city	300+ years	Less than 100 years
Rate of population growth	Moderate	Rapid
Urban growth mode	Spread and redevelopment	Spread
Urban form	Compact, with core and fringing suburbs	Extreme spread, coalescing cities, interior open space
Limits to expansion	None	Public and Indian land
Interior open space	Abandoned	Never developed or remnant
Economic base	Industrial	Hi-tech, tourism
Ecosystem boundaries of LTER study	Primary Statistical Metropolitan Area	County or regulated portion of watershed
Patch definition	Watershed; socioeconomic patches; ecological patches	Combination of patch age, position, neighbors, land use, land-use history

initiate succession (which probably occurs at very different rates). Water is undoubtedly the primary limiting factor to production in the desert region that Phoenix occupies, although nitrogen limitation also is prevalent in both terrestrial and aquatic ecosystems (Grimm and Fisher 1986). In Baltimore's eastern deciduous forest, limiting factors vary seasonally, but may include low temperatures and frost in winter and soil nutrient limitation and occasional late summer drought during the growing season.

In terms of societal organization, there are important differences as well. Modern Phoenix is a much younger city than Baltimore, having been established only after the Civil War and having experienced a meteoric rise in population just since World War II (and, interestingly, since the invention of air conditioning). Baltimore is more than 300 years old; it was established as a seaport after a long history of scattered habitation of the area (see Foresman et al. 1997 for a detailed analysis of land-use change in the region). The site of Baltimore did not support a large prehistoric urban settlement, but the Hohokam civilization in central Arizona included thousands of inhabitants until its demise (circa 1350 AD).

Because of climatic conditions, the Phoenix area was not actively settled until quite late in prehistory (circa 500 AD), compared with surrounding regions of the Southwest. The limitation was insufficient rainfall for agriculture; hence, any kind of substantial occupation would require knowledge of irrigation techniques and social organization to build and maintain the system. The Hohokam civilization was able to work cooperatively and establish a settlement system centered in the valley, which grew quickly. From prehistoric times to the present, the environmental challenges presented by an arid environment have required cooperative activity by human groups for urban centers in such environments to succeed. Individual farmsteads, such as those that characterized Baltimore's prehistory, would have been at serious risk in an arid environment. In contrast, prehistoric settlement of the Baltimore area succeeded based on small-scale agriculture and harvesting of coastal resources by small social groups. However, for Baltimore to move beyond its small-scale roots, as it has done in historic times, cooperative action was required, this time to build and maintain the harbor and to establish global trading connections. The summons to cooperative action was both environmental (an accessible harbor) and socioeconomic (a market for long-distance exchange of goods). The establishment of a plantation culture both relied on and contributed to global trade because of the perceived need for slave labor.

Finally, present-day contrasts in population growth and characteristics of urban growth and form reflect the distinction between an older city that has undergone economic restructuring (loss of manufacturing jobs and more service jobs) and a newer city with an economic base that has always been primarily in the service sector, with, recently, high-tech manufacturing. The Phoenix metropolitan area is one of the fastest growing in the nation (exceeded only by Las Vegas), with a population that is projected to double within approximately 25 years (Hall et al. 1998) and spreading development that consumes up to 4 square miles of desert or agricultural land per 1000 residents added to the population (Gober et al. 1998). The population of Baltimore city has declined 23% between 1950 and 1990, but growth is high at the county level, reflecting flight from urban to suburban and rural communities (Rusk 1996).

Despite these marked dissimilarities, we see an opportunity for convergence in the realm of urban ecological study. For both projects, the central objective is to understand how land-use change over the long term influences ecological pattern and process, and how this suite of changes feeds back through the social system to drive further change (e.g., Figure 3). Furthermore, although some of the key questions posed for the two systems are necessarily different, there are some common questions and a common approach has been adopted for answering many of them. The guiding principle of the LTER program—that research should, whenever possible, enable comparison between different ecosystems and across sites—dictates attention to ensuring comparability of approach. Some of the research efforts of the Baltimore and Phoenix studies reveal how social and ecological research can be integrated in ways that permit cross-comparison of results.

Baltimore Ecosystem Study: Integration of Physical, Biological, and Social Drivers of Watershed Dynamics

The Baltimore study uses watersheds as fundamental units in which to study the reciprocal interactions of the social, biophysical, and built environments. Over the past 300 years, human settlement and land management have substantially changed the character and productivity of the Chesapeake Bay, the largest and most productive estuarine system in the world (Brush 1994). Hydrologists, ecologists, and social scientists, together with public agencies, nonprofit organizations, and community groups, are working to understand how people at different scales (households, neighborhoods, and municipalities) directly and indirectly affect water quality in the watersheds of the Baltimore metropolitan region. Initial research has shown a significant relationship between concentration of political and economic power in the city and the different levels of public and private investment in green infrastructure among neighborhoods (green, open spaces and trees). Additional research is focusing on the direct ways in which households might affect water quality through irrigation and through the use of fertilizers and pesticides, as well as on how such land-management practices vary with household demographic and socioeconomic characteristics.

This research will provide important information on how people influence urban watersheds at different scales and will result in a hydrological–ecological–social watershed model that policymakers, planners, and managers can use to assess strategies for improving the water quality of the watersheds.

Central Arizona–Phoenix Study: Urban Fringe Dynamics

Phenomenal rates of urban growth and expansion in the Phoenix metropolitan area are being studied with the intention of defining a new framework through which to view the ecosystem (Gober et al. 1998). The doubling of population in each of the last two 30-year periods has led to a rapid spread of the Phoenix urban area into former farmlands and undisturbed desert landscapes. To monitor this growth, researchers are analyzing data on the exact locations of new residences for each year of the past decade. The data reveal that almost all new single-family residences are built along the periphery of the city and that urban sprawl acts as a "wave of advance" that spreads out from several nodes of urban development, as has been observed for other youthful cities (Whyte 1968). The speed of this wave and its geographic dimensions seem to respond to conditions of the local economy and characteristics of the landscape. In turn, the landscape and local ecosystem are transformed by this advance of new housing in several predictable stages—land-surface preparation (removal of vegetation, soil disturbance); infrastructure construction; scattered, "pioneer" housing developments; and fill-in of vacant land with housing. Behind the wave, neighborhoods age, leading to continued transformations in the nature of the human and biotic populations that inhabit those spaces. This analysis provides a locational tool that is more sensitive to the key processes that define the urban phenomenon than the normal grid map of the city.

Summary and Synthesis

If Vitousek et al. (1997) are correct that by excluding humans we cannot possibly understand ecosystems, and if it is critical that the social, behavioral, and economic sciences join in the endeavor to understand ecosystems, then it is essential that ecologists welcome the approaches and models of the social, behavioral, and economic sciences. We have argued that one way to do so is by focusing on five new core topic areas for social science. We have argued as well that the study of cities as ecological systems presents opportunities for theoretical advances in ecology, and that such advances cannot be accomplished without integration of the social sciences.

We believe that the expansion of social science activities within the LTER network will do much to facilitate construction of the concepts most appropriate for understanding change in the world, as will new initiatives in studies of humans as components of ecosystems. With this commitment, ecologists can begin to ask important questions about interdisciplinary research, agree about methodology and measurement, and successfully integrate social and ecological data across scales of time and space. At no time has there been a more compelling need for integration and such a wide diversity of researchers ready to begin.

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Integrating Humans into Ecology: Opportunities and Challenges for Studying Urban Ecosystems

Marina Alberti, John M. Marzluff, Eric Shulenberger, Gordon Bradley, Clare Ryan, and Craig Zumbrunnen

Abstract Our central paradigm for urban ecology is that cities are emergent phenomena of local-scale, dynamic interactions among socioeconomic and biophysical forces. These complex interactions give rise to a distinctive ecology and to distinctive ecological forcing functions. Separately, both the natural and the social sciences have adopted complex system theory to study emergent phenomena, but attempts to integrate the natural and social sciences to understand human-dominated systems remain reductionist—these disciplines generally study humans and ecological processes as separate phenomena. Here we argue that if the natural and social sciences remain within their separate domains, they cannot explain how human-dominated ecosystems emerge from interactions between humans and ecological processes. We propose an integrated framework to test formal hypotheses about how human-dominated ecosystems evolve from those interactions.

Keywords: ecology · human-dominated ecosystems · urban patterns · emergence · niche

For most of human history, the influence of human beings on biophysical processes, ecological systems, and evolutionary change has been relatively limited, as compared with the influence of "natural" (nonhuman) processes. Ecological and evolutionary change has generally been attributable to natural variation in energy and material flows and to natural selection by parasites, diseases, predators, and competitors. Today, however, humans affect Earth's ecosystems at extraordinary rates through conversion of land and resource consumption (Turner et al. 1991), alteration of habitats and species composition (McKinney 2002), disruption of hydrological processes (Arnold and Gibbons 1996), and modification of energy flow and nutrient cycles (Vitousek et al. 1997a, Grimm et al. 2000). Humans now use approximately 40% of global net primary production (Vitousek et al. 1986) and more than half of accessible freshwater runoff (Postel et al. 1996). At least half of the world's forests have disappeared as a result of human activity, and three-quarters of that total have disappeared since 1700 (Harrison and Pearce 2001). Human activities fix amounts of nitrogen and sulfur comparable to those fixed by all nonhuman causes (Graedel and Crutzen 1989). Humans have radically revamped Earth's carbon cycle (Prentice et al. 2001) and freed into the environment vast quantities of naturally occurring trace materials (e.g., cadmium, zinc, mercury, nickel, arsenic) and exotic new anthropogenic substances (e.g., polychlorinated biphenyls, chlorofluorocarbons) (Pacyna and Pacyna 2001).

Humans also influence evolutionary processes. Selection is more and more frequently directed by people, or at least by people interacting with other natural processes. For example, humans affect speciation by challenging bacteria with antibiotics, poisoning insects, rearranging and exchanging genes, creating and dispersing thousands of synthetic compounds, and selectively

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fishing (Palumbi 2001). By hunting, moving predators and competitors around the globe, and massively reconfiguring the planet's surface, humans have increased extinctions of other species to levels 1000 to 10,000 times higher than those resulting from nonhuman causes (Pimm et al. 1994, Vitousek et al. 1997b, Flannery 2001). The combined effect of changing speciation and extinction is rapid evolutionary change (Palumbi 2001).

Despite dominating Earth's ecosystems, humans remain conspicuously excluded as subjects of much ecological thinking and experimentation. Traditional ecological research investigates ecosystems in terms of biophysical, ecological, and evolutionary processes unaffected by human influences. During the last 100 years, formidable strides have been made in the scientific understanding of ecological systems (Likens 1998). Evolutionary theory and population genetics have made fundamental changes in the assumptions underlying ecological research. Ecological scholars no longer regard ecosystems as closed, self-regulating entities that "mature" to reach equilibria. Instead, they see such systems as multi-equilibria, open, dynamic, highly unpredictable, and subject to frequent disturbance (Pickett et al. 1992). In the newer non-equilibrium paradigm, succession has multiple causes, can follow multiple pathways, and is highly dependent on environmental and historical context. Ecosystems are driven by processes (rather than end points) and are often regulated by external forces (rather than internal mechanisms). The new ecological paradigm recognizes that humans are components of ecosystems (McDonnell and Picket 1993). Yet ecological scholars often fail to include humans in ecological science (Hixon et al. 2002, Reznick et al. 2002, Robles and Desharnais 2002).

Applied ecology has extensively challenged the assumptions of an ecological paradigm that assumes human-free systems, but ecology has not yet provided a new theoretical framework to fully integrate humans into ecosystem studies. Here we argue that humans must be explicitly incorporated into all aspects of ecological thought, because, by adding powerful selection forces at every spatial scale and at many temporal scales, humans are fundamentally changing the expression of the rules that govern life on Earth. To paraphrase Hutchinson (1965), humans are changing the ecological stage on which the evolutionary play is performed. To understand the new evolutionary play, ecological scholars must build a new stage with humans as a central plank.

Urban Ecology: Understanding Human-Dominated Ecosystems

Planet-scale changes induced by humans are most evident in and around the urbanizing landscape (Fig. 1). Urbanized areas cover only approximately 1% to 6% of Earth's surface, yet they have extraordinarily large ecological "footprints" and complex, powerful, and often indirect effects on ecosystems. Earth's urban population has increased more than 10-fold over the past century, from 224 million in 1900 to 2.9 billion in 1999 (Sadik 1999). According to the United Nations (Sadik 1999), all expected population growth from 2000 to 2030 (approximately 2 billion people) will be concentrated in urban areas. By 2030, more than 60% (4.9 billion) of the estimated world population (8.1 billion) will live in cities.

Ecological scholars studying urban areas have challenged ecological theory to explain the ecology in and of cities (Pickett et al. 2001). The urban long-term ecological research sites are now producing important empirical observations (Collins et al. 2000). Some have argued that important revisions to ecological theory are needed to include human activity (Collins et al. 2000, Grimm et al. 2000). To understand specific sets of interactions between humans and ecological processes that occur in urbanizing regions, we propose examining cities as emergent phenomena—phenomena that cannot be explained simply by studying the properties of their individual parts. Cities are both complex ecological entities, which have their own unique internal rules of behavior, growth, and evolution, and important global ecological forcing functions.

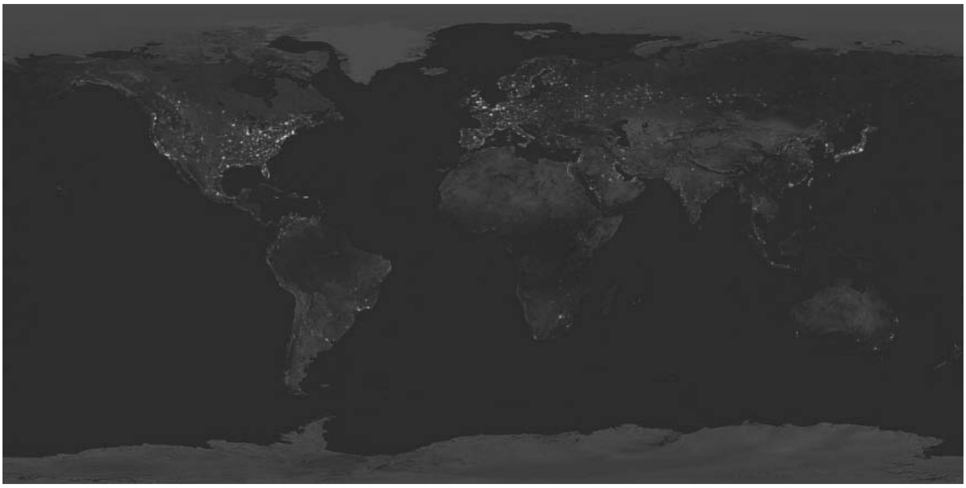

Fig. 1 The extraordinary impact of urbanization on Earth is now detectable from space. This composite of satellite images shows how Earth looks at night. Source: National Aeronautics and Space Administration

Cities as Emergent Phenomena

Ecology is a science of emergent phenomena: Populations have properties (birth and death rates) and behaviors (schooling in fishes, flocks of birds) not inherent in individuals. Like other ecosystems, cities are not the sum of their constituents; they are key examples of emergent phenomena, in which each component contributes to but does not control the form and behavior of the whole. Traffic congestion, air pollution, and urban sprawl emerge from local-scale interactions among variables such as topography, transportation infrastructure, individual mobility patterns, real estate markets, and social preferences. What makes urban regions different from many other ecosystems is that in these regions humans are a dominant component.

Cities evolve as the outcome of myriad interactions between the individual choices and actions of many human agents (e.g., households, businesses, developers, and governments) and biophysical agents such as local geomorphology, climate, and natural disturbance regimes. These choices produce different patterns of development (Fig. 2), land use (Fig. 3), and infrastructure density (Fig. 4). They affect ecosystem processes both directly (in and near the city) and remotely through land conversion, use of resources, and generation of emissions and waste. Those changes, in turn, affect human health and well-being (Alberti and Waddell 2000). We propose that *resilience* in cities—the

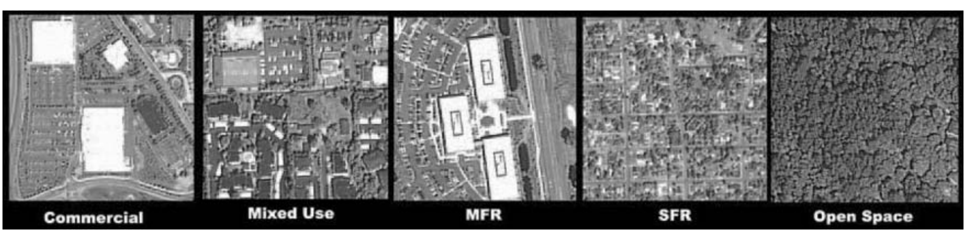

Fig. 2 Urban ecological studies need to explicitly represent the complex urban landscape patterns if they are to answer questions about strategies for achieving more sustainable urban forms. Urban development is characterized by different land-use types (industrial, commercial, mixed use, single-family residential [SFR], multifamily residential [MFR], and open space), which exhibit different land-cover composition and configuration. The Urban Ecology Team at the University of Washington is conducting a study that aims to shed some light on the impact of urban patterns on bird diversity and aquatic macroinvertebrates. Data source: IKONOS 2000

Fig. 3 Development patterns exhibit different degrees of residential density. Data source: IKONOS 2000

Fig. 4 Differing infrastructure densities imply varying degrees of land-cover change and fragmentation in urbanizing areas. Data source: IKONOS 2000

degree to which cities tolerate alteration before reorganizing around a new set of structures and processes (Holling 2001)—depends on the cities' ability to simultaneously maintain ecosystem and human functions.

Cities as Complex Ecological Entities

A diverse literature has begun to document some ecological characteristics of urban regions both in the United States (McDonnell and Picket 1993, Grimm et al. 2000) and in Europe (Sukopp and Werner 1982, Sukopp et al. 1995). Human-dominated landscapes have unique biophysical characteristics. Humans redistribute organisms and the fluxes of energy and materials. The effects are both obvious (e.g., pavement) and subtle (e.g., conversion of forest to agriculture and then to suburbs; acid rain), both immediate (e.g., dams drown river valleys) and long-term (e.g., new intercity highways direct and promote city growth on 20- to 100-year scales). Relative to non–human-dominated systems, urban ecosystems have low stability, different dynamics (complex and highly variable on all temporal and spatial scales), more nonnative species, different species composition (often simplified, always changed), and unique energetics (antientropic in the extreme). They have rich spatial and temporal heterogeneity—a complex mosaic of biological and physical patches in a matrix of infrastructure, human organizations, and social institutions (Machlis et al. 1997).

Human activities directly affect land cover, which controls biotic diversity, primary productivity, soil quality, runoff, and pollution. Urbanized areas also modify microclimates and air quality by altering the nature of the land surface and generating heat (Oke 1987). Urbanization's increase in impervious surface area affects both geomorphological and hydrological processes; it changes fluxes of water, nutrients, and sediment (Leopold 1968, Arnold and Gibbons 1996). Because ecological processes are tightly interrelated with the landscape, the mosaic of elements resulting from urbanization has important implications for ecosystem dynamics. The transformation of land cover favors organisms that are more capable of rapid colonization, better adapted to the new conditions, and more tolerant of people than are many endemic, sensitive, locally specialized organisms. As a result, urbanizing areas often have novel combinations of organisms living in unique communities. Mixes of native and nonnative species interact in complex, anthropogenically driven successions, but with human participation, they also equilibrate into communities stable over time. Diversity may peak at intermediate levels of urbanization, at which many native and nonnative species thrive, but it typically declines as urbanization intensifies (Blair 1996). Rearranging the pattern of land cover also changes the composition of communities; edge species, or those inhabiting interfaces among vegetation types and ecotones (such as white-tailed deer), typically increase, and interior species, or those rarely occurring within a few hundred meters of interfaces (such as northern spotted owls), decline (Marzluff 2001).

Cities as Global Ecological Forcing Functions

The importance of cities as drivers of economic development has been recognized for a long time (Jacobs 1961), but their role as a global ecological driving force is not yet fully appreciated (Rees 1992). Many ecological changes forced by cities on their immediate environments are obvious and extreme and have been extensively documented (McDonnell and Picket 1993). Although ecological impacts of urban development often seem to be local, urbanization also causes environmental changes at larger scales. Today's cities are sustained by a socioeconomic infrastructure that operates on global scales; the ecologically productive area required to support an urban area can be 100 to 300 times larger than the urban region (Rees and Wackernagel 1994). Scholars have drawn on the concept of carrying capacity to propose ways to measure a city's ecological footprint (Rees 1992, Rees and Wackernagel 1994) and appropriated ecosystem area (Folke et al. 1996). Rees and Wackernagel (1994) estimate the ecological footprint of Vancouver (British Columbia, Canada) at more than 200 times its geographic area; likewise, Folke et al. (1996) estimate that the appropriated ecosystem area required to supply renewable resources to 29 major cities in the Baltic Sea drainage basin is 200 times the total area of the cities.

The spatial organization of a city and its infrastructure affect the resources needed to support the city's human activities and thus the city's level of environmental pressure on the regional and global environment (Alberti and Susskind 1997). The land development needed to house the same number of people varies, depending on choices about location, density, and infrastructure. Whether an urban dweller chooses a private or public transportation system to commute between home and work, for example, depends on the availability of a public transportation system, which in turn depends on the political–economic feasibility of such a system, given the distribution of human activities. These choices have important ecological consequences globally and locally.

Challenges for Ecology

The greatest challenge for ecology in the coming decades is to fully and productively integrate the complexity and global scale of human activity into ecological research. How can ecological scholars best study the complex biotic and abiotic interactions within human-dominated ecosystems, the emergent ecology of these systems, and their ecological forcing functions? We challenge the assumption that a "human-free" ecosystem paradigm can be productively applied to human-dominated ecosystems. We argue that leaving humans out of the ecological equation leads to inadequate explanations of ecosystem processes on an increasingly human-dominated Earth.

Integrating humans into ecosystems will provide important opportunities for ecosystem science. Consider, for example, how the key ecological concept of the niche could benefit from explicit inclusion of humans. Hutchinson (1957) transformed and solidified the niche concept, changing it from a mere description of an organism's functional place in nature (Elton 1927) to a mathematically rigorous *n*-dimensional hypervolume that could be treated analytically (Fig. 5). He also emphasized a single dimension of the hypervolume, interspecific competition. Hutchinson's "realized niche" included only those places where an organism's physiological tolerances were not exceeded (its "fundamental" niche) and where its occurrence was not preempted by competitors (Fig. 5). Emphasizing competition in the niche concept distracted ecologists from investigating other potentially important community organizing forces, such as predation, resources variability, and human domination. A more complete understanding of ecological community assembly has begun to develop (Weiher and Keddy 1999), but it still lacks the inclusion of humans. We suggest that niche theory should distinguish realized from fundamental niches on the basis of human interaction (Fig. 5). Redefining the realized niche as an organism's hypervolume of occurrence in the presence of a gradient of human domination (Fig. 5) would quantify the myriad ways humans force population-level ecological functions that structure communities. Understanding the mechanisms of niche assembly in the presence of humans would allow ecologists to directly test the effects of competitors, preda-

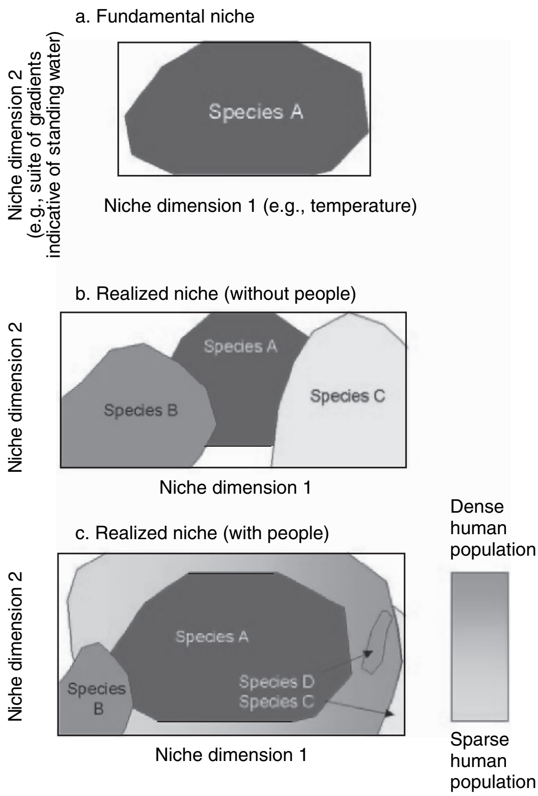

Fig. 5 The fundamental niche of a species (those areas on Earth where its physiological tolerance limits are not exceeded) is an *n*-dimensional hypervolume, where each environmental gradient relevant to a species is one dimension (Hutchinson 1957). (a) A two-dimensional view of two axes from the *n*-dimensional Hutchinsonian niche hypervolume for species A, which can exist in areas with moderate temperature and moderate relative humidity. We contend that theoretical ecologists have steered empiricists away from fully understanding how communities are assembled by emphasizing competition in the formalization of the niche concept. A better paradigm for understanding how biotic communities are structured is to document the effect of humans on species' realized niches. (b) The Hutchinsonian realized niche is that portion of the fundamental niche not preempted by competitors, shown here for three species in the absence of people. (c) In this model of a realized niche with human involvement, Species A (e.g., a human commensal, such as the Norway rat) expands to fill its fundamental niche in the presence of people. Species B has a restricted distribution because the human-subsidized species A outcompetes it. Species C is intolerant of humans and is confined to portions of its fundamental niche where people do not exist. Species D is imported by humans into the geographic niche space indicated in (c). As humans dominate more of Earth, the types of processes that assembled and structured this community (niche space) are fast becoming the rule rather than exception

tors, disease, and land-cover change on community organization, because these ecological processes are often manipulated by humans. The challenge for ecology is to define how humans differ in their effects on ecological processes and, through comparing these differences, to gain clearer insight into how nature works.

An integrated consideration of human interactions with food web complexity may shed light on another ecological contentious ecological principle: the influence of biological diversity on ecological stability. Human domination can increase food web complexity (e.g., by interspersing built and natural habitats; Blair 1996), but this does not necessarily increase ecological or anthropogenic stability (i.e., resilience). Uncoupling the connection between diversity and stability in human-dominated ecosystems highlights the importance of species identity, rather than simply species richness, to community stability. Investigating the changing relationship between diversity and stability along a gradient of human domination can clarify when diversity begets stability, when diversity simply means unnecessary redundancy of ecological roles, and when diversity leads to instability (e.g., diversity resulting from importation of invasive exotics).

Traditional ecological investigations of populations and communities could benefit from studying human-dominated ecosystems, as we suggest above. This has been shown, for example, by studies of the dynamics of nutrient cycling and energy flow that have begun to incorporate human domination (Vitousek et al. 1986, 1997a). These studies have enabled better prediction of ecosystem-level processes and have led to a greater appreciation of human influences on the planet.

A Conceptual Model for Urban Ecology

Ecologists are paying increasing attention to the relationship between urbanization and ecosystems (Collins et al. 2000, Grimm et al. 2000, Pickett et al. 2001), but few have directly addressed how human and ecological patterns emerge from the interactions between socioeconomic and biophysical processes. Current study of urban ecosystems uses such simplified representations of human–ecological interactions that their system dynamics cannot be fully appreciated and understood. For example, most ecological studies treat urban areas as homogeneous phenomena and combine all anthropogenic factors into one aggregated variable (e.g., pollution load, population density, total paved area); thus, they represent urbanization as unidimensional. This is unrealistic: Urbanization is multidimensional and highly variable across time and space. Socioeconomic studies, on the other hand, highly simplify and rarely discriminate among different and complex ecological and biophysical processes. This aggregate representation of human and ecological processes cannot explain human–environment interactions in human-dominated systems, nor can it allow ecological scholars to fully understand the complex dynamics of such systems, because many of these interactions occur at levels not represented in current integrated approaches (Pickett et al. 1994).

Ecologists and social scientists have studied emergent ecological and social phenomena, but they have not explored the landscape-level implications of interactions between social and ecological agents. In their separate domains, neither the natural nor the social sciences can explain how integrated human and ecological systems emerge and evolve, because human and ecological factors work simultaneously at various levels. Ecologists have studied self-organized patterns in social insect colonies composed of hundreds to millions of genetically similar individuals. These individuals interact locally, but collectively they produce large-scale colony dynamics that are not predictable from the individuals' characteristics. Urban planners, economists, and sociologists have described cities as self-organizing systems in which emergent bottom-up processes create distinct neighborhoods and unplanned demographic, socioeconomic, and physical clusters. The need to share local services and a customer base drives residents and businesses together, while competition for land, labor, and customers drives them apart. Because of these forces, initial random distributions in human-dominated landscapes rearrange spontaneously into a self-organized pattern with multiple diverse clusters (Krugman 1995).

To fully integrate humans into ecosystem science, we propose a new conceptual model that links human and biophysical drivers, patterns, processes, and effects (Fig. 6). Although several new models address the relationship between urbanization and ecosystem dynamics (Collins et al. 2000, Grimm et al. 2000, Pickett et al. 2001), they do not explicitly represent the interactions between human and biophysical patterns and processes, nor do they represent the feedbacks from these interactions. In our model, both biophysical and human agents drive the urban socioeconomic and biophysical patterns and processes that control ecosystem functions. Using this framework, ecological scholars can ask questions about how patterns of human and ecological responses emerge from the interactions between human and biophysical processes and how these patterns affect ecological resilience in urban ecosystems. This model can help test formal hypotheses about how human and ecological processes interact over time and space. It can also help establish (a) what forces drive patterns of urban development, (b) what the emerging patterns are for natural and developed land, (c) how these patterns influence ecosystem function and human behavior, and (d) how ecosystem and human processes operate as feedback mechanisms. Without a fully integrated framework, schol-

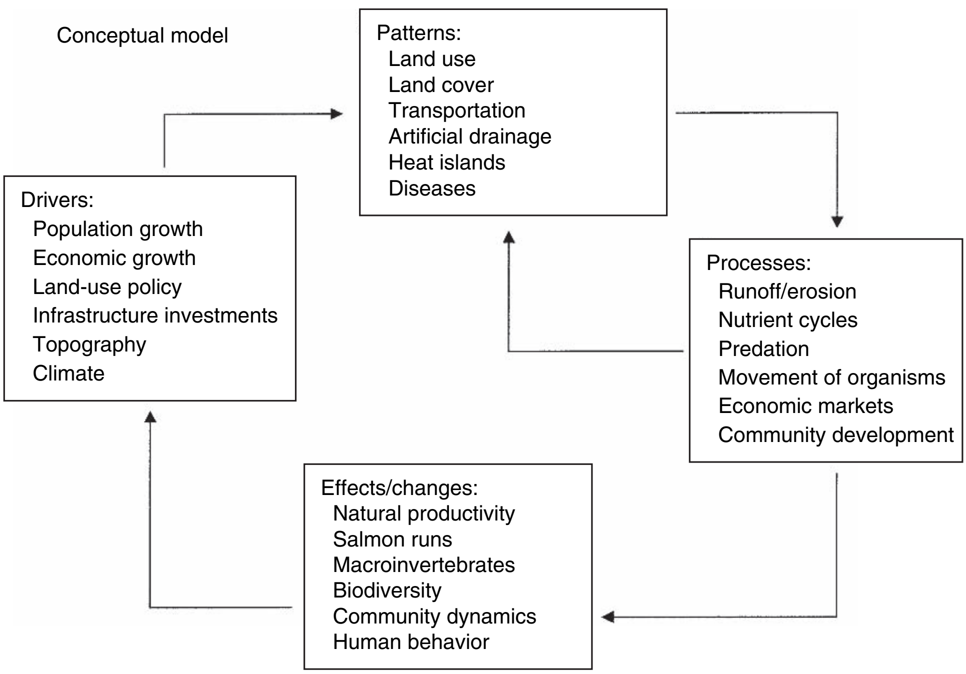

Fig. 6 An integrated model of humans and ecological processes to understand forces driving patterns of urban development, quantify resulting patterns of natural and developed land, determine how these patterns influence biophysical and human processes, and assess the resulting environmental changes and feedback on human and biophysical drivers. In this conceptual model, drivers are human and biophysical forces that produce change in human and biophysical patterns and processes. Patterns are spatial and temporal distributions of human or biophysical variables. Processes are the mechanisms by which human and biophysical variables interact and affect ecological conditions. Effects are the changes in human and ecological conditions that result from such interactions. In the diagram we provide some explicit examples of drivers, patterns, processes, and effects. For example, population growth in an area (driver) leads to increased pavement and buildings (patterns), leading to increased runoff and erosion (processes), causing lower water quality and decreased fish habitat (effects), which may lead to a new policy to regulate land use (driver). However, the same variable can fit into different boxes depending on the focus (issue, scale, and time frame). For example, erosion is a process, but it can also be seen as a pattern that influences other processes such as nutrient cycles or as an effect resulting from runoff

ars can neither test hypotheses about the systems' dynamics nor produce reliable predictions of ecosystem change under different human and ecological disturbance scenarios. Such knowledge is critical if managers and policymakers are to control and minimize the effects of human activities on ecosystems.

An Example: Urban Sprawl

Urban sprawl illustrates the complexity of interactions and feedback mechanisms between human decisions and ecological processes in urban ecosystems. Sprawl manifests as a rapid development of scattered (fragmented), low-density, built-up areas ("leapfrogging"; Ewing 1994). Between 1950 and 1990, US metropolitan areas grew from 538,720 square kilometers (km^2) (84 million people) to 1,515,150 km^2 (193 million people). Land development due to urbanization has grown 50% faster than population (Rusk 1999). Sprawl is driven by demographics (e.g., increases in numbers of households), socioeconomic trends (e.g., housing preferences, industrial restructuring), and biophysical factors (e.g., geomorphological patterns and processes) and is reinforced by infrastructure investment choices (e.g., development of highway systems; Ewing 1994). Sprawl is strongly encouraged by land and real estate markets (Ottensmann 1977) and is now a highly preferred urban living arrangement (Audirac et al. 1990).

The phenomenon of sprawl shows how considering only aggregated interactions between humans and ecological processes cannot help explain some important mechanisms that drive human-dominated ecosystems. Human decisions are the primary driving force behind environmental conditions in urban ecosystems, but these conditions cannot be explained by taking separately the behavior of individual agents (e.g., households, businesses, developers) competing in each market (e.g., job market, land and real estate market). Households, which are themselves complex entities, simultaneously compete in the job and real estate markets when deciding where to live. Furthermore, these agents have preferences and make tradeoffs that are highly dependent on biophysical factors. Decisions about land development and infrastructure are strongly influenced by biophysical constraints (e.g., topography) and environmental amenities (e.g., "natural" habitats). From local interactions among these agents eventually emerge metropolitan patterns, which in turn affect both human and biophysical processes. Resulting changes in environmental conditions then strongly influence some important human decisions. Furthermore, in these systems, uncertainty is important, since any departure from past trends can affect system evolution.

Sprawl has important economic, social, and environmental costs (Burchell et al. 2002). It fragments forests, removes native vegetation, degrades water quality, lowers fish populations, and demands high mobility and an intensive transportation infrastructure. Such environmental changes may eventually make suburban sprawl areas less desirable for people and may trigger more development at increasingly remote locations. But urban feedback is changed in form and is phase-lagged, often by decades (e.g., results of decisions on highway development). Municipalities are largely responsible for promoting sprawl. For example, cities often subsidize sprawl by providing public services (schools, waste disposal, utilities) that are priced independent of their real cost and distance from central facilities (Ewing 1997), so that residents in the sprawled periphery usually do not pay the full costs of their own services (Ottensmann 1977).

The "complex system" paradigm provides a powerful approach for studying urban sprawl as an emergent phenomenon and for devising effective policies to control its effects. Complex structures can evolve from multiple agents operating according to simple decision rules (Resnick 1994, Nicolis and Prigogine 1989). Some fundamental attributes of complex human and ecological adaptive systems—multiple interacting agents, emergent structures, decentralized control, and adapting behavior—can help scholars to understand how urbanizing landscapes work and to study urban sprawl as an integrated human–ecological phenomenon. Complex metropolitan systems cannot be managed by a single set of top-down governmental policies (Innes and Booher 1999); instead, they require the coordinated action of multiple independent players operating under locally diverse biophysical conditions and constraints, constantly adjusting their behavior to maintain an optimal balance between human and ecological functions.

A Research Agenda for Urban Ecology

We believe that a radical change is needed in how scholars frame questions about urban ecology. Instead of "How do socioeconomic phenomena affect ecological phenomena?" the question should be "How do humans interacting with their biophysical environment generate emergent collective behaviors (of humans, other species, and the systems themselves) in urbanizing landscapes?" Theories about complex adaptive systems provide tools with which to analyze how landscape-scale organization of structures and processes arises in urbanizing regions; how it is maintained; and how it evolves by local interactions of processes that occur at smaller scales among social, economic, ecological, and physical agents (self-organization). These theories also provide a new framework for understanding how distributed control, information processes, and adaptation in human-dominated systems should guide the development of policies to effectively balance human and ecosystem functions in urbanizing regions. Specifically, urban ecology scholars need to address four fundamental questions:

1. How do socioeconomic and biophysical variables influence the spatial and temporal distributions of human activities in human-dominated ecosystems?
2. How do the spatial and temporal distributions of human activities redistribute energy and material fluxes and modify disturbance regimes?
3. How do human populations and activities interact with processes at the levels of the individual (birth, death, dispersal), the population (speciation, extinction, cultural or genetic adaptation), and the community (competition, predation, mutualism, parasitism) to determine the resilience of human-dominated systems?
4. How do humans respond to changes in ecological conditions, and how do these responses vary regionally and culturally?

Our conceptual framework provides a new theoretical basis to test formal hypotheses about the mechanisms that link urban patterns and ecosystem dynamics at multiple scales and about the influence of these mechanisms on the resilience of urban ecosystems. First, we hypothesize that both biophysical and human agents drive the urban socioeconomic and biophysical patterns and processes that control ecosystem functions. Second, we hypothesize that patterns of development (urban form, spatial organization of land use, and connectivity) influence ecosystem dynamics. Third, since alternative patterns of urbanization affect the ability of a system to maintain a balance between human and ecosystem services, we hypothesize that the patterns generate differential effects on ecological resilience. Fourth, we hypothesize that in complex human-dominated ecosystems, changes at one level of the biological and social organization can alter emergent human-ecological phenomena at another level.

Driver Hypotheses

Urban ecosystems provide an excellent gradient to test hypotheses on emergent human-ecological phenomena. A complex set of social, political, economic, and biophysical factors drives urbanization and affects when, where, how, and at what rate urban development proceeds. In studying interactions between human and ecological processes, researchers need to address explicitly the complexities of many factors working simultaneously on scales from the individual to the regional and global. Consideration solely of aggregated interactions cannot help explain or predict important feedbacks or outcomes, so testable models must be spatially referenced ever more explicitly and finely. Lag times between human decisions and their environmental effects further complicate understanding of these interactions. For instance, in urban ecosystems, land-use decisions affect species composition directly (e.g., introduction and removal of species) and indirectly (e.g., modification of “natural” disturbance agents like fire and flood). If ecological productivity controls the regional economy, interactions between local decisions and local-scale ecological processes can cause large-scale environmental changes (Alberti 1999).

Pattern Hypotheses

A second set of hypotheses that can be effectively tested in urbanizing landscapes concerns the effects of human–ecological patterns on human and ecological processes. Landscape ecologists and urban planners debate relationships between spatial patterns of urban development and ecological conditions, but few empirical studies have provided evidence of mechanisms linking urban patterns to ecological and human functions in urbanizing landscapes. We argue that different urban patterns (i.e., urban form, land-use distribution, and connectivity) generate differential effects on ecosystem dynamics and therefore differ in their ecological resilience. This is because urban development patterns differently affect the amount and interspersion of built and natural land cover as well as anthropogenic demands on ecosystem services. We hypothesize that ecological and socioeconomic conditions can be discriminated across a gradient of urbanization patterns.

Resilience Hypothesis

We hypothesize that resilience in an urban ecosystem depends on multiple human and ecological services provided by natural and human systems. To assess that resilience, researchers must understand how interactions between humans and ecological processes affect the inherently unstable equilibria between the end points of the urban gradient. Over the long term, human services in urban areas (housing, water supply, transportation, waste disposal, recreation) all depend on ecosystem functions for their productivity (Costanza et al. 1997, Daily 1997). Integrating humans into ecology will help identify the thresholds to best balance human and ecosystem services in urban ecosystems.

Scale Hypotheses

One critical problem in urban ecology is understanding how change at one level of biological and social organization will alter emergent patterns or mechanisms at another level. A hierarchical approach has been proposed to better explore the relationship between top-down and bottom-up forces in determining ecosystem dynamics (Wu and David 2002). Urban ecosystems provide the best setting to test hypotheses on the dynamic hierarchical structure of human-dominated landscapes. Such knowledge would make it easier to manage complex, human-dominated ecosystems successfully. For example, in working to maintain biodiversity, managers usually begin at the species level, but this misses the fundamental importance of biodiversity at other scales: Higher-level biodiversity provides interconnections between multiple elements operating at multiple levels and transforms the community from a random collection of species into an ecosystem of interrelated biotic and abiotic parts (Levin 1998).

Practicing a New Urban Ecology

Effective integration of humans into ecological theory, which is both beneficial and necessary in order to better understand ecological systems in general, and human-dominated systems in particular, requires effective team building, interdisciplinary training, and a new dialogue between science and policymaking.

Effective Team Building and Education

Most of today's scientific and social problems lie at the interface of many scientific disciplines. Strategic decisions about how best to address urban growth require the synthesis of extraordinarily complex and rapidly evolving knowledge from a broad range of disciplines (e.g., forestry, fisheries, urban planning, zoology, civil engineering, landscape architecture, geography, political science, sociology, psychology, and economics). Effective approaches require high-performance teamwork. It is naive to assume that scholars trained in a single discipline can successfully create interdisciplinary research teams and teach in interdisciplinary settings. To effectively bridge gaps among disciplines, scientists need to learn new skills with which to frame problems and design solutions that address multiple perspectives simultaneously. To achieve this level of synthesis, scientists need to be aware of their own mental models, disciplinary biases, and group dynamics. This requires (a) investigating differences between disciplines (what the values are; how questions are posed; what constitute valid data; how data are gathered, processed, and reasoned about) and (b) understanding and managing group dynamics.

This awareness comes slowly to established scientists, but it can evolve rapidly if the next generation of urban ecologists is trained in a new way. Our experience suggests that students of urban ecology need strong disciplinary bases, but they especially need qualities rarely developed by traditional graduate programs: interdisciplinary experience, breadth, flexibility, team building, and sophisticated skills in communication and synthesis. These skills can be layered on strong disciplinary foundations by graduate education that emphasizes interdisciplinary and team-based research focused on real-world problems. Students must understand the differences in how social scientists, ecologists, managers, and policymakers formulate and define problems, ask questions, gather and evaluate information, and propose and implement solutions. Students who receive such training will improve relationships among academic, business, regulatory, and urban communities.

A New Relationship Between Science and Policy

Urban ecology ultimately involves studying how to integrate this new interdisciplinary knowledge about urban ecosystems into policymaking processes—to improve interactions between policymakers and scientists so as to help society achieve more sustainable urban forms. Today, the scientific and political communities lack the effective two-way communication and trust that they need to address urban ecological problems. A number of factors contribute to this division between science and policy. Society sets goals through the policy process, which is not solely driven by science's commitment to analytical norms and searching for "truth." Although science can help society formulate a range of options to achieve societal goals, it cannot make value judgments. In addition, scientists often cannot deliver definitive answers to questions posed by policymakers. Scientists often disagree about causes of environmental problems, so policymakers need to act under scientific uncertainty. Policymakers often claim they cannot afford to wait for "scientifically correct" answers to problems. Furthermore, even when causal knowledge exists on environmental problems, it does not necessarily lead to action. The urban ecology's scientific community needs to participate actively to inform policymaking and make scientific results relevant to policy decisions, even though most scientists receive little training on the policy process. In the same way, policymakers must participate in formulating scientific questions and defining priorities if science is to become relevant in decisionmaking. Inviting policymakers into the classroom to help shape graduate research projects helps forge this new relationship.

Toward Consilience?

Urban ecology holds great promise for advancing ecological understanding, providing society with important information that can encourage sustainable development, and allowing social and biological scientists to effectively integrate information. Together, these objectives may lead toward the consilience, or unity of knowledge across fields, that Wilson (1998) argues has eluded science. This unity of sciences and humanities must become the backbone of urban ecology. Without it, socially relevant and ecologically accurate research will not materialize, policy decisions will be made without the full benefit of relevant scientific information, and cities will continue to grow in increasingly unsustainable ways. Employing a unified approach, the next generation of urban ecology scholars can conduct interdisciplinary research, and practitioners can provide society with the tools to set and prioritize goals, make informed tradeoffs, and develop and implement policies toward more sustainable urban development.

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Section III
The Atmosphere, Hydrosphere, and Pedosphere

Section III - *Atmosphere, Hydrosphere, and Pedosphere* - provides readings on the physical underpinnings of urban ecosystems. Interrelated processes among the biotic and abiotic spheres are essential to understanding the ecology *of* a city – considering a city to be one system. Regarding the *pedosphere* subsystem, in his new article about soil sealing Wessolek addresses crucial problems such as the infiltration of rainwater, soil water availability, and soil pollution (e.g., by hydrocarbons, fertilizers, biocides, or heavy metals). Robbins & Sharp (2003) explore new ground in integrated nature-society urban ecology research with their rigorous, empirically- and theoretically-grounded investigation of the complex biophysical, social, economic, marketing, psychological, and aesthetic drivers and dimensions involved in the creation of intensive residential lawn management in the United States.

In all larger cities the *hydrosphere* is of immense importance. In Berlin, Germany, for example, the water supply is exclusively obtained from within the administrative borders of the city by means of wells and riverbank filtration. As a consequence, the availability and quality of groundwater as well as surface water, including more or less natural waterways, shallow lakes, human-made channels and transformed ponds, are of great importance. Paul & Meyer (2001) paper gives an excellent general overview of the impacts of urbanization on rivers and streams, including both geo-morphological aspects and biological effects.

In studies of the urban *atmosphere*, climatological problems, such as human-induced changes in surface radiative properties, discharges of airborne industrial and transport contaminants creating air quality problems, and heat fluxes leading to the formation of the urban heat island, are very important issues. However, global climate change and alterations of the local climate in high-density built-up areas are already leading to severe interferences with the *anthropospheric* subsystem; they especially affect the urban atmospheric system. To address the actual situation and the importance of the urban atmosphere, we place strong emphasis on this subsystem. For example, Kuttler's new paper gives an introduction to urban climate, and Alcoforado & Andrade provide a new summary of urban heat island research. Urban heat islands are, for example, an important factor and driving force for the invasion of alien species. Endlicher et al. (2006) deal with the consequences of heat waves on public health under changing climate conditions. Bell, Davis & Fletcher (2004) paper is dedicated to health perspectives, too, as exemplified by the most famous historic winter smog episode of London in 1952.

Sealing of Soils

Gerd Wessolek

Abstract To judge sealing of soil and interventions in locations in the framework of city and water management planning, one needs quantitative data relating to soil properties, climate and water cycles. The assessment of substances is increasingly important, especially if impacts by emission, motor traffic or by abandoned polluted sites have to be considered. In Germany today sealed and built-up areas occupy a significant and increasing fraction of the surface area (>12 %). In this paper various methods to detect sealing are dealt with in detail. Different measurement techniques can give very different numbers about the degree of sealing. Sealing especially affects the natural water cycle. As a rule, sealing reduces actual evapotranspiration and increases surface runoff (canalizing). Sealing also strongly changes the urban heat balance: the higher the building density, the more heat buildup in summer. As compared with the surroundings, on extreme summer days the temperature may increase by ∼2–3 K and the saturation deficit of air by ∼10–20 %. Sealing of soil also destroys the habitats of fauna and flora. The remaining areas in the city become isolated and develop extreme local conditions. One usually finds a change of fauna and flora, and a shift of the species rank order of abundance. The increased pollutant load in urban spaces, plus the frequent concentration of rainwater into unsealed and partly sealed areas, often causes high pollutant concentrations in the topsoil and in the materials in joints or cracks sin the pavement. High contents of heavy metals in soil and litter may slow down the decomposition of litter, soil respiration, and mineralization.

Keywords: urban soil · urban climate · water management · impervious surfaces · infiltration · road · water balance · runoff · soil temperature

Introduction

Anthropogenic earth filling, excavation, mixing and sealing measures are referred to as "load-casting" of soil. By "sealing" we mean the covering and paving of soil surfaces mostly in urban areas. In urban agglomerations, significant portions of the area may be load-cast and sealed; thus e. g. in Berlin and in the Ruhr region more than 35 % of the area is built-up and paved.

Sealing is not uniformly distributed with regard to soil types and soil quality. Soils occurring in especially good locations are more frequently load-cast and built-up because a favourable location, traffic situation and other properties promote early settlement and economic development. This applies especially to wide river valleys with alluvial loamy soils and river marshes and fertile loess locations where chernozem and luvisols with a high water and nutrient content have developed. As a rule, oligotrophic soils consisting of sand or solid rock (e.g. regosols, podzols) and humid locations (e.g. low moor soils, fens) are less densely populated and sealed (Blume, 1992).

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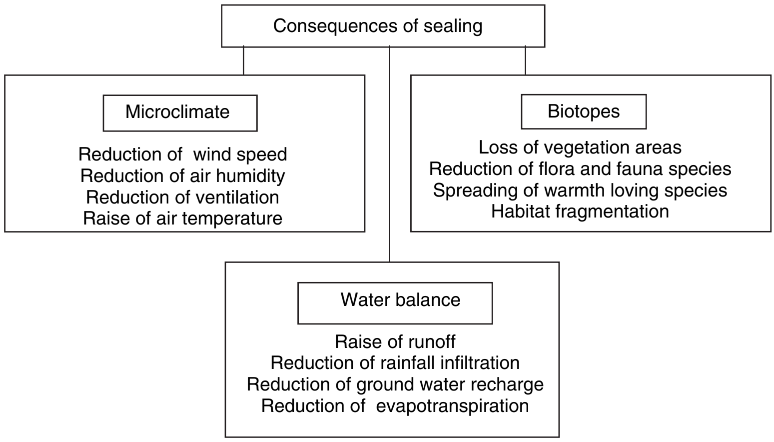

Fig. 1 Effects of sealing (per Muth, 1996)

As compared with naturally developed soils, soils of urban-industrial agglomerations show significantly modified properties such as sealing of surfaces, compaction, increased stone contents, deposition of technogenic (manmade) substrates, increased humus contents up to a depth of 40–50 cm, and changes of the depth of the groundwater table.

We now deal with methods to determine types of sealing and development, their ecological properties, and changes of the water and heat balances. We also consider how biological functions are changed, and the presence of various types of pollution, by considering the results of investigations done in urban spaces. In Figure 1, first of all, the most important effects of sealing have been compiled.

Determination of the Degree of Sealing

Frequently the urban gradient method is used for investigating the degree of sealing. In this method one chooses a space typical for the city (depending on the use, load-casting, soil and groundwater situation) and proceeds from the centre outwards to a peripheral area. Within this space one defines "ideal" types of use patterns, i.e. representative for the respective range of properties and development of a use unit. One uses aerial photos and ground inspection to measure the area of each partial space, and to categorize each space into one of these categories of sealing: buildings, buildings planted with trees, impermeably sealed area, permeably sealed area, unsealed area (Table 1).

Within each partial area the portions of the respective categories are summed and the data are aggregated on the basis of quadrants (1 quadrant = 1600 raster fields). A drawback of this method is the subjectivity in choosing sample types.

Table 1 Classification of urban areas into sealing classes (per Boecker, 1984)

Sealing Classes	Classes in %	Site Characteristics
I	10–50	Low up to moderate sealing, one family house settlements, areas with small gardens, line house settlements; average: 30% = class I
II	45–75	Moderate sealing, block buildings, reconstruction areas after the second world war, average: 60% = class II
III	70–90	Strong sealing, municipal building areas block buildings; former industrial areas, average: 80% = class III
IV	85–100	Very strong sealing, undestroyed areas block buildings of the interior municipal districts and industrial areas, which have arisen or been changed most recently; average: 90% = class IV

Table 2 Development of land use in the Federal Republic of Germany since 1950, data in per cent of the total area (completed according to Deggau et al., 1992)

year	settlement and traffic area	agriculture area	forest area	water area	remaining areas[3]
1950[1]	7,0	57,5	28,4	1,8	5,3
1955[1]	7,1	58,2	28,5	1,6	4,6
1960[1]	7,6	57,7	28,7	1,7	4,3
1965[1]	8,3	56,8	29,0	1,7	4,1
1970[1]	9,3	55,7	28,9	1,8	4,3
1975[1]	10,0	55,0	28,9	1,8	4,2
1981[2]	11,0	55,3	29,5	1,7	2,5
1985[2]	11,6	54,5	29,6	1,8	2,5
1989[2]	12,2	53,7	29,8	1,8	2,5
1997[4]	13,3	52,8	30,2	1,9	1,8
1997[5]	11,8	54,1	29,4	2,2	2,5

[1]Results of land utilization elevations.
[2]Results of the area elevation 1981, 1985, and 1989 respectively deadline 31th December of the previous year. Because of various elevation methods and delimitations of results of the land utilization pre-elevation area elevation only are comparably limited.
[3]Land utilization elevation: uncultivated bog area; anthrosol area; sports-, flight- and armed forces drill grounds. Area elevation: bog; heath; mining country; areas of other use; as of 1989 without inclusion of cemeteries.
[4]Old federal territory
[5]Federal Republic of Germany complete(all data of 1997 from the Statistic Federal Office, 1998)

Because most areas' use changes over time (eg due to development) (Table 2), most urban areas cannot be handled with a single detailed measurement of the degree of sealing. They need a continuous up-dating of the measures. For compiling the sealing map of the environment atlas of Berlin (1993) satellite data (Landsat-5 thematic mapper) were used in combination with infra-red aerial photos and the digital basic maps for West Berlin. On the basis of this preparatory work, we investigated how well Landsat TM satellite photos can be used for detecting and updating the degree of sealing in urban areas. We can make the following statements: Land use classifications of Landsat-TM data used for area-covering water balance investigations can, in principle, allow one to determine the degree of sealing of urban areas. However, one must be careful – assigning a degree of sealing of 100 % or 80% and 95 % to the land use classes "settlement" and "high building density" results in a clear overestimation of the area effective for runoff. In fact, a general assignment of low degrees of sealing to these land use classes provides better results for the inner-city area. A drawback of the methods of classification and inquiry is that similar areas are summed up in various classes (see Fig. 3).

The highly variable data in Table 3 show that it is difficult to specify a single or uniform method of measurement, owing to the heterogeneity of urban structures.

Properties of Sealed Locations

Physical Properties

A list of the physical properties of some sealing materials (Table 4) shows that the total pore volume and the water capacity are relatively low; in particular the proportion of medium-sized pores is small. The same occurs in moistening capacity; it correlates strongly with the quantity of water or precipitation adhering to the material before the runoff begins. On warm summer days the runoff starts later than in the colder winter because the first part of the rain evaporates quickly, and the sealing material cools down.

Table 3 Influence of various methods of inquiry on the degree of sealing (according to Münchow, 1996)

Housing type	A %	B %	C %	D %	E %	F %	* %
Block buildings	75–95	65–85	85–95	70–100	90	78,6	35
Block edge buildings	50–65	65–85	50–65	40–80	65	62,2	45
Line buildings	40–65	36	40	40–70	50	42,5	34
Town hall	35–60	o.A.	60	40–60	50	o.A.	25
Terraced house	30–50	20	o.A.	50–60	40	28,4	40
One-family house	20–40	27	45	20–50	35	39,7	30
Industry/Trade	60–90	75–90	80–90	50–100	80	84,3	40
Parks	5–20	1–20	o.A.	10–40	15	o.A.	35
Streets	80	92–100	o.A.	40–100	o.A.	o.A.	60

A Sealing degree according to Pietsch (1985), from Münchow, 1996
B Sealing degree according to Jentschke & LANGE (1987), from Münchow, 1996
C Sealing degree according to Bundesminister Fur Raumordnung, Bauwesen Und Städtebau Brbs (1988)
D Sealing degree according to Berlekamp & Pranzas (1990)
E Sealing degree according to Strotmann (1991), from Münchow, 1996
F Sealing degree according to Blume (1990), from Münchow, 1996
* Difference of the details in %
o.A.without details

Inner-city soils which developed atop debris and urban ruins usually show a relatively low water capacity owing to their comparatively high fraction of “skeleton” or rubble (eg broken concrete, etc) (Fig. 2). The particle-size distribution of the rubble is critical: the smaller the particles are, the more they may contribute to supplying plants with water. Also the soil components found below sealing materials frequently have highly-modified (as compared with natural soils) or extreme properties (e.g., buried railroad-track or road-beds).

However, the various soil properties are not determined only by the soil materials themselves, but by various compaction processes such as human foot-traffic, vehicular traffic, and construction work. Wessolek & Facklam, 1997). Urban road dust and weathering of the joint material (over decades) can fill joints and cracks and reduce infiltration capacity, e.g., by increasing the proportions of organic substance, clay and silt in the upper few cm (Table 5).

Table 4 Physical properties of sealing materials (Wesolek & Facklam, 1997). Density determined by pyknometer

Material	Density (g/cm^3)	BD (g/cm^3)	Laboratory studies	Field studies					
								required precipitation to start runoff (mm)	
			TPV	pF(1,8)	pF(4,2)	aWC	MC (mm)	warm	cold
Granite pavement	2,61	2,51	3,9	$< 0,5$	$< 0,5$	$< 0,5$	0,4	1,8	1,2
Concrete bond	2,58	2,44	5,1	1	$< 0,5$	1	0,8	n.d.	1,4
Bricks	2,65	2,06	22,2	11,3	10,1	1,2	0,8	n.d.	0,9
Gum, rubber	1,16	0,84	27,6	8,7	4,5	4,2	2	n.d.	1,6
Asphalt	2,4	1,88	21,6	$< 0,5$	$< 0,5$	$< 0,5$	0,4	1,5	1

MC = capacity of moistening (maximal adherent water amount in mm)
BD = bulk density
TPV = Total pore volume
aWC = available water capacity (pF 1,8 – pF 4,2)
n.d. = not determined

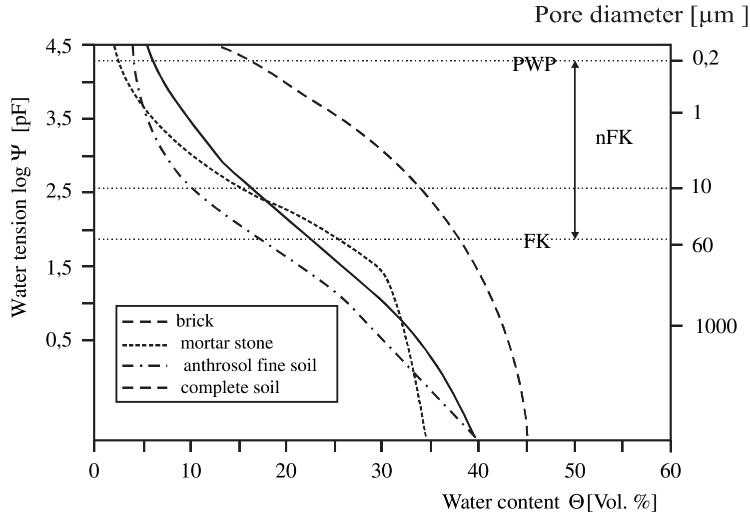

Fig. 2 Relation between water content (Θ) and water tension (ψ) for various skeletal and soil fractions of debris soils from ruins (modified according to Runge, 1978)

In a soil sealing research project implemented by the "Umweltforschungszentrum" Leipzig-Halle GmbH, the hydraulic-physical properties of new and aged area pavements have been measured (Schramm, 1996). It is hard to do a good job of categorizing surface pavements as to their hydrological properties by using parameters that are easy to measure (e.g. % of surface that is actually crack or joint, or grain size distribution) (Table 5). That is why parameters for characterizing infiltration properties of paved areas have to be derived from field or laboratory tests. Field tests are best because aging processes such as settlement and input of dust, mineral, and organics may be included. Irrigation tests are especially suited for determining the infiltration behaviour of covers as the natural infiltration processes may be better reproduced by simulating precipitation than by modelling infiltration processes.

The test results show that by choosing appropriate construction methods high infiltration rates may be reached on paved areas. The infiltration capacity of conventional slab and block pavements is, however, clearly reduced by aging processes or by dust and dirt input, mostly as those materials seal cracks and joints which are the primary routes for infiltration (Fig. 3).

Whenever final infiltration rates are < 10 mm/h, even low intensity rain events cause runoff from older block and slab pavements. The lowest final infiltration rates were measured on two water-

Table 5 Grain size distribution and humus contents of aged[1] and new[2] joint materials (Wessolek & Facklam, 1997)

			Grain size distribution (%)							
			Clay		Silt		Sand fractions			
Location in Berlin	Depth (cm)	Humus (%)	< 2	2–6,3	6,3–20	20–63	63–125	125–200	200–630	630–2000
			μm							
Soldiner Street[1]	0–2	2,56	1,7	2,3	3,6	8,0	15,3	42,7	23,6	2,8
	2–4	0,31	0,1	0,6	0,6	4,5	13,7	48,9	29,1	2,6
Gotenburger Street[1]	0–2	3,30	1,1	1,5	3,2	5,8	10,6	27,2	37,8	12,8
	2–4	0,43	0,7	0,2	0,9	1,8	3,1	14,6	57,6	21,1
Geisbergstreet[1]	0–2	2,78	0,2	3,3	3,7	6,8	11,9	37,2	32,7	4,2
	2–4	0,36	0,5	0,1	0,5	3,9	9,0	41,6	39,4	5,0
Joint material [2]		0,08	3,2				6,3	34 4	48,6	7,5

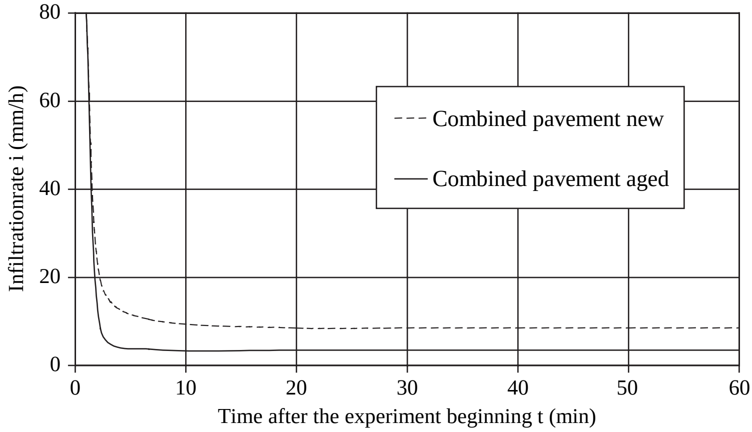

Fig. 3 Infiltration rates of new and aged composite block pavement (modified according to Bischoff, 1996)

bound covers (Table 6). The infiltration capacity of water-bound covers depends strongly on the grain size distribution in the cover and base layers. If the sand or skeletal portion is high and the portion of fine grain sizes low, water-bound covers are comparable to conventional block and slab pavements.

Water Balance

Sealing measures have their strongest effects on the water cycle due to the capillary rise being reduced/interrupted, evapotranspiration being reduced, infiltration being shortened and reduced, and surface runoff being accelerated and increased.

Figure 4 shows a typical cross-section of the hydrological conditions in an urban space. Rainwater falling onto paved ways or roads is either infiltrated or discharged from the surface (canalisation). Given suitable geological conditions the road and roof water runoff is frequently discharged directly into soil

Rainwater not adhering to the sealing material or infiltrating into the joint space is discharged as surface water. Table 7 lists the average coefficients for rainwater runoff from sealed surfaces. These

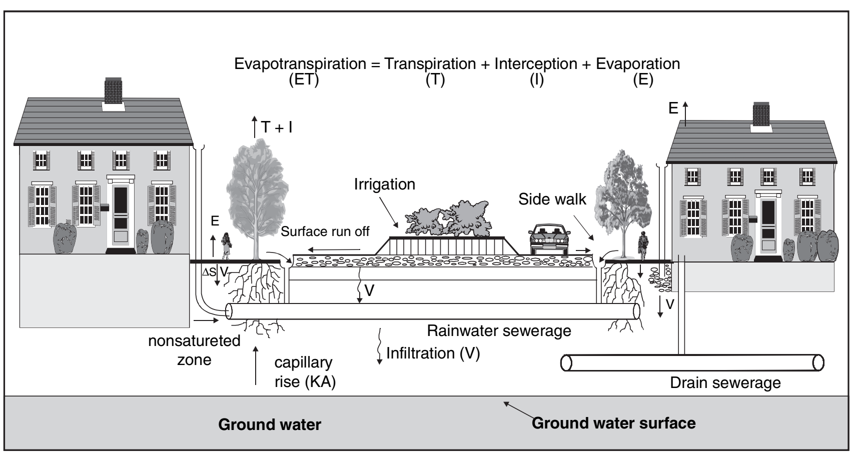

Fig. 4 Schematic illustration of the water balance components in an urban area

Table 6 Comparison of joint portions and final infiltration rates of new and aged surface pavements (from Bischoff, 1996)

Samples	Material	Use	Production year	Remarks	Joint shares (%)	Middle of final infiltration rates (mm/h)
new area						
1	concrete bond plaster, sand joined	parking place	1992	Joints moss-grown	5,3	9,9
2	stones of lawn bars	parking place	1990/91	lawn seed, covering 90 %	27,7	> 70, 0
3	stones of lawn bars	parking place	1990	spontaneous vegetation, covering 25 %	27,7	21,9
4	bond plaster	driving sector	1993	no joints, gravel bedding	4,6	> 70, 0
5	bond plaster with seep opening	place for the car	1993	seeping opening with fine gravel filled, gravel bedding	7,8	> 70, 0
6	bond plaster with seeping opening	parking place	1993	seeping openings only partial filled (top soil), filled with mud, partial spontaneous vegetation	8,9	11,2
7	concrete bond plaster with lawn joints	parking place	1994	lawn seed, covering > 75 %	22,8	19,2
8	concrete plaster with lawn joints	parking place	1994	seed not opened, covering < 5 %	22,8	5,8
9	synthetic resin bound cover	cycle track	1994	geolen-sand mixture	-	45,9
aged area fastenings						
10	large format disks (1,2 × 1,2 m)	sidewalk	1977/78	joint share humos, moss-grown	1,7	5,2
11	concrete sidewalk disks (30 × 30 cm)	sidewalk	1950/60	joint share strong humos,	5,2	5,3
12	Cobble-stone plaster, sand joined	road, side street	about 1944	joint share strong humos,	19,6	9,8
13	Cobble-stone plaster, sand joined	road, side street	about 1944	joint share strong humos,	10,9	4,4
14	mosaic-plaster	sidewalk	about 1900	joint share strong humos, moss-grown	26,2	3,2
15	mosaic-plaster	sidewalk	about 1900	joint share strong humos, moss-grown	22,4	6,2
16	natural stone disks (ca. 0,5 × 1,2 m)	sidewalk	about 1900	Joint share humos, moss-grown	2,3	3,0
waterbounded covers						
17	water bound cover	parking place sidewalk	1975	high skeleton share in cover and carrier course	–	6,2
18		sidewalk	1991			8,4
19.1		(2 locations)	1993			0,8
19.2						
20			1993	high skeleton share in cover course, low silt and clay contents		12,7

Table 7 Runoff coefficients for various soil surfaces (Bischoff 1996)

Type of area	Drain coefficient
1. Waterproofed areas	
– roof areas > 3° inclination	0,9
– concrete areas	0,9
– black cover	0,9
– roof areas < 3° inclination	0,8
– gravel roofs	0,5
– green roof areas:	
– for intensive and extensive planting as of 10 cm structure thickness	0,3
– for extensive plantings less than 10 cm structure thickness	0,5
2. Partial permeable and weak deflected areas	
– concrete stone pavement, in sand or clinker laid	0,7
– areas with pavement, with joint shares > 15 %	0,6
– waterbounded areas	0,5
– sports areas with drainage:	
– synthetic areas, synthetic lawn	0,6
– threshing-floor areas	0,4
– lawn areas	0,3
3. water permeable areas	
– parks and vegetation areas, gravel and clinker floor, rolling gravel, with fastened patches	0,0
– garden-pathes with waterbounded cover	0,0
– entrances and carports with lawn bars stones	0,0

coefficients do not contain information on where the water flows or how much water is discharged through the sewage system, although that is essential to balancing of an area's water regime.

The available water resources are essentially reduced by sealing, which strongly restricts true evapotranspiration. As a rule, evaporation occurs only for the first 1–2 days after precipitation events: it lasts only until the water adhering to the sealing material or stored in the pore space is gone. Thereafter, only the joint spaces participates in evaporation, but in them evaporation decreases quickly because the sandy substructure has only an insignificant capillary conductivity. Figure 5 shows the effects of various water storing capacities (0.2 mm up to 2.0 mm) of the sealing materials on the extent of actual evapotranspiration as a function of the summer precipitation. The data determined for the region Berlin illustrate the variation of the actual evaporation from sealed areas though the absolute differences in the water storage capacity are comparatively small. The evaporation of sealed areas amounts, on an average of the years, between 50 and 120 mm related to the summer half-year (1.4 – 30.9). This example shows also how urban-climatic and hydrological effects may be affected by a purposeful selection of sealing materials.

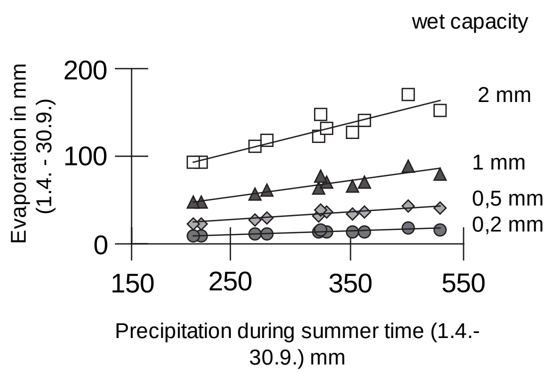

Fig. 5 Cumulative actual evaporation in the summer half-year as a function of the precipitation for various water storing capacities of the sealing material

The effects of use and sealing on annual variations of soil-moisture tension is shown in Fig. 6 for four uses (ornamental lawn, lawn bars, ash cover, asphalt). In the region of Hanover, tensiometer measurements (from Pagel et al., 1993) showed how dessication in summer and rewetting in winter alternate, and how these processes are weakened and delayed (that is, develop time-lags) with increasing depth. A high correspondence of individual water tension phases with the variation of the climatic water balance (CWB) at a depth of 30 cm is striking in the case of ornamental lawns. The CWB was calculated from the difference between precipitation and potential evapotranspiration (per Penman, 1956). Below the ornamental lawn and – restricted – below the lawn paving stones they found a typical periodical variation with low values in summer and high values in winter. In the case of a lawn with lots of pavers the limited transpiration of the sparse grass vegetation causes a comparatively insignificant subsurface desiccation. In the winter, precipitation predominates, producing a climatic water excess, i.e. fast recharge of the soil water stocks and early beginning of infiltration into groundwater. The potentials represented in Figure 6 at a depth of 200 cm show the variation of the groundwater surface.

Below ash dumping grounds, which are covered with an asphalt layer classified as "full sealing", we assume a high surface runoff and essentially no evaporation. In fact, the annual wet-dry cycle is strongly reduced, but NOT absent: the annual variation is quite detectable even at a depth of 20 cm (Fig. 6). Even below this cover the relative desiccation phases in summer (with a high evaporation loss of the heated up asphalt) alternate with a re-wetting in the winter. Obviously, infiltration can happen right through the asphalt layer, e.g. through cracks caused by traffic and other loads, frost or aging of the material. It was possible to prove this also with the aid of infiltration tests. Speer & Schneider (1992) always found hydraulic gradients directed downwards below the road edge or shoulder (the "verge"). It is possible that the temperature-driven transport of vapour is significant. In summer water transport downwards is increased by surface; in winter the opposite transport upwards with a subsequent condensation on the cooler asphalt layer contributes to re-wetting. Comparable processes are assumed for surface sealing of dumping grounds where penetration of surface water is prevented (Vielhaber et al. 1992).

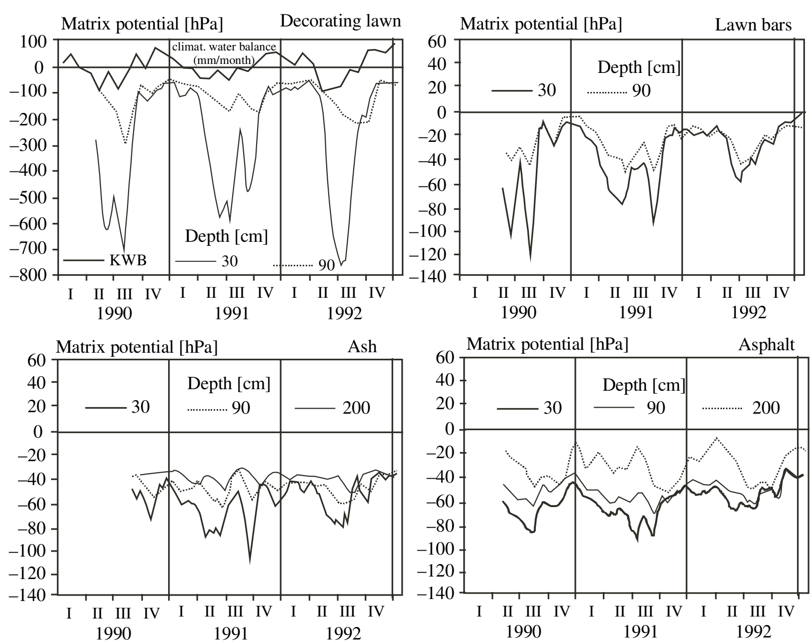

Fig. 6 Annual variation of the matrix potentials (1990-92) at three depths, with use and sealing varying (from Pagel et al. 1993)

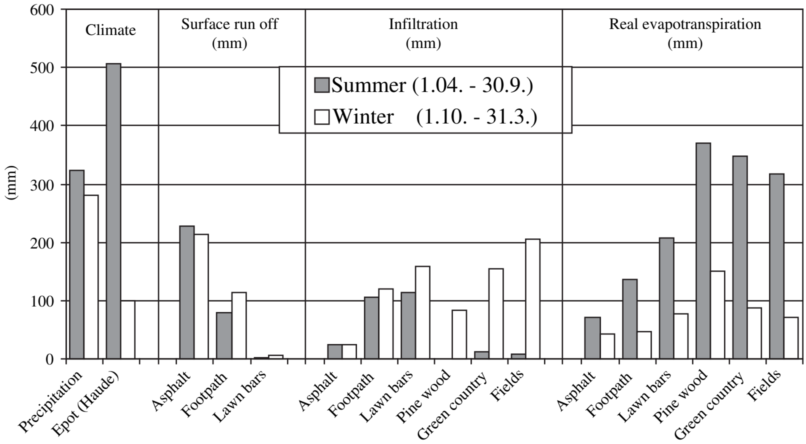

Fig. 7 Water balance components of various land uses and sealing materials (average from three-year lysimeter measurements of the Berliner Wasserbetriebe and water balance investigations)

Even with increasing anthropogenic interference, a typical annual alternation of infiltration and evaporation occurs, as may be still recognized from the distance between the curves for the individual depths in the case of ornamental lawn (Fig. 6). Instead of it an infiltration takes place in the case of lawn grating and ash covering and also below asphalt throughout the year. For partly permeable areas this agrees with the lysimeter results of the Berliner Wasserbetriebe which show that when the % of joint area is high, infiltration or natural groundwater recharge may be higher than below overgrown free space. (Fig. 7). The average, soil-bearing natural groundwater recharge in a built-up area totals ∼ 10–20 % of the average annual precipitation in the North German area. By comparison, the average natural groundwater recharge in a free landscape totals 23–24 % below arable land and 14 to 21 % below forest.

Heat Balance

Various uses and sealings have also affect the heat balance; Figure 8 shows the annual variation of soil temperatures (Pagel et al. 1993). The ornamental lawn shows the typical tendency of an area covered by vegetation throughout the year. Individual phases in the annual variation are influenced by the weather. This becomes evident by the parallel course of the average air temperature per month indicated for 1990–92. As depth increases, the annual amplitude is reduced, and the phase-lag or time delay develops, through which minimums and maximums occur later and later with depth. With anthropogenic influence increasing from ornamental lawn to lawn grating, to ash layer up to asphalt, however, there is a clear increase of the annual amplitude of the temperature cycle. The strong heating of the soil up to a depth of 1.80 m in summer is striking. Soil temperatures under sealed surfaces are clearly higher than the annual average and in the urban landscape heat islands may develop. It is to be expected that the partly extreme temperature gradients affect also the water balance by inducing additionally a transport of water vapour. (Pagel et al. 1993). Water vapor follows the potential gradient, and is always directed from warm to cold. In summer, during daytime warming, a transport of heat and water vapour downwards into deeper soil layers takes place. During the night the flow is reversed in the upper soil region. In winter the flow is predominantly directed upwards into the atmosphere.

Because of strong warming of the sealing materials the runoff of precipitation sets in only later. The first 0.3 to 0.6 mm of rainwater are quickly evaporated – until the temperature difference

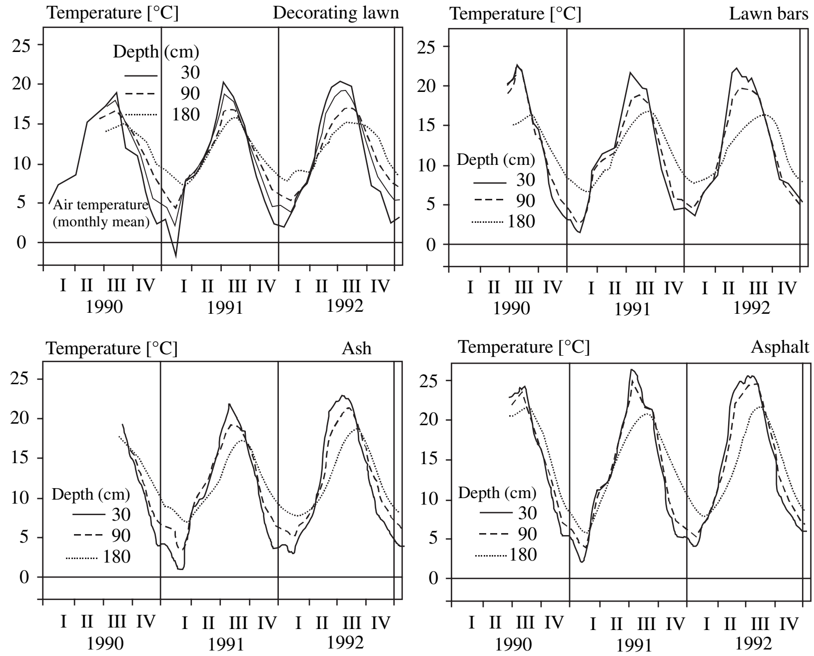

Fig. 8 Annual variation of soil temperatures 1990–92 at three depths, with use and sealing varying (from Pagel et al. 1993)

between sealing material and air or rainwater has been balanced. This is especially evident in rain events after hot summer days. The cooling thus caused is mostly felt to be agreeable, but if there is only a little rain, the temperature of the sealing materials is reduced only insignificantly while atmospheric moisture increases strongly in the short term, thus increasing the humidity and making people uncomfortable.

For sealed areas, their modified warming up and cooling down in connection with the small water resources affects the extent of evaporation. During rainfall, evaporation declines due to atmospheric humidity increasing: it rises again after the rainfall stops. Evaporation stops if there is no longer water in the sealing material and the joints (Glugla & Krahe 1995).

This connection shows clearly the importance of inner-city green spaces to a climatic balance. Positive effects come from green spaces on facades and/or roofs which on built-up, sealed areas may take over at least partly the functions of soils that have been sealed by development. According to Kuttler (1993), on a roof area covered with greenery during a warm summer day, most radiation energy is used for evaporation (585 W/m^2), and only insignificant amounts flow into the building (85 W/m^2) and into the air (70 W/m^2). By comparison, 210 W/m^2 got into the air on a gravel roof and 210 W/m^2 flowed into the building. In addition to climatic balancing effects, green roofs and facades covered with greenery reduce air pollution near the house by increasing deposition speeds of particles, and reducing gaseous trace materials (Kuttler 1993).

Biological Functions

Owing to the central position taken by soil in many ecosystems, sealing causes modifications in substance and energy cycles; in addition it affects fauna and flora and the quality of housing and quality of life of the human population (Fig. 9).

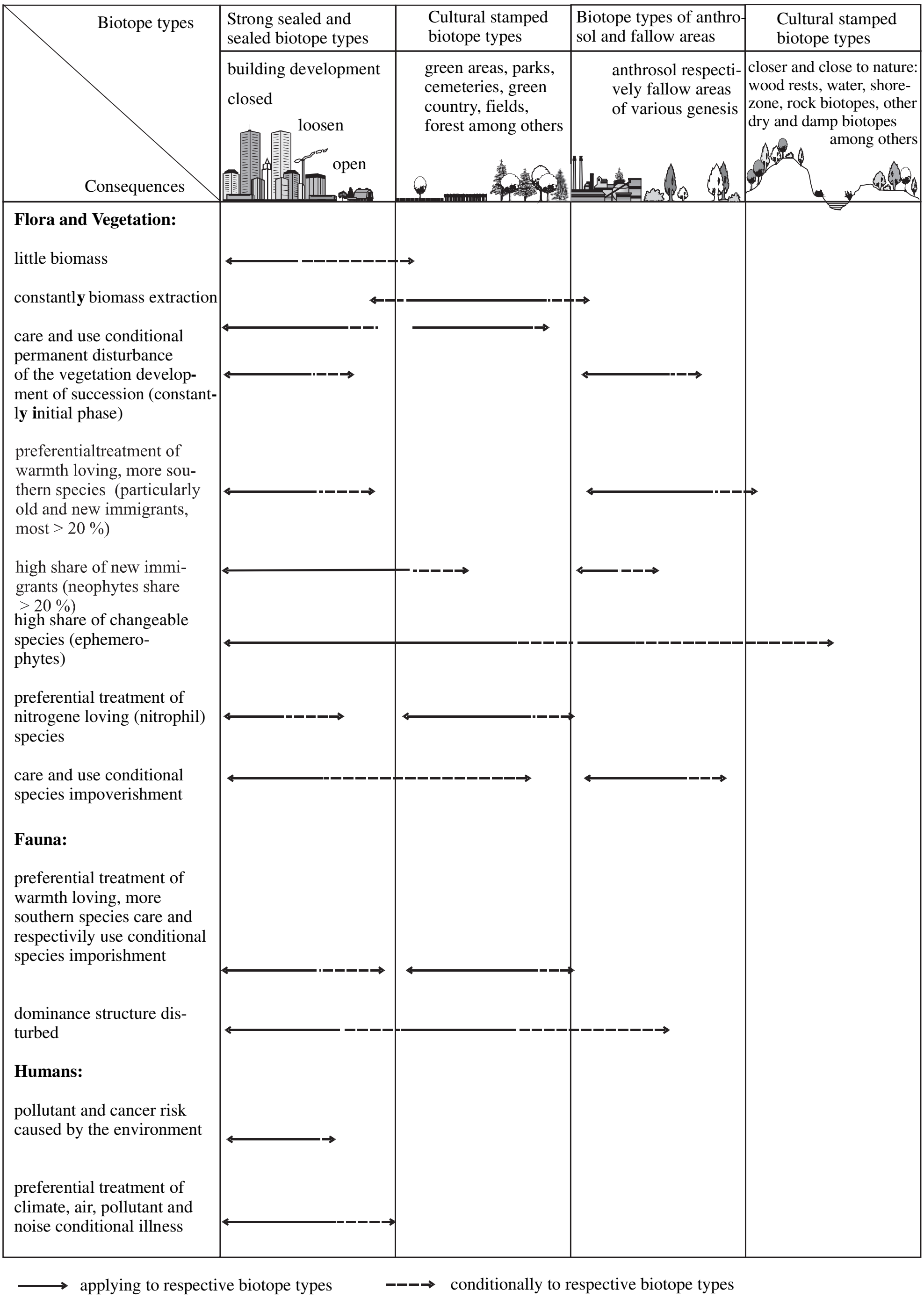

Fig. 9 Changes of the biosphere in cities, their surrounding and in villages (Bflr 1988) multitude of factors affecting fauna and flora which gain directly and indirectly in importance with increasing sealing are summed up in Fig. 9

Full sealing affects fauna and flora by destroying habitat. The remaining unsealed areas in a city become isolated and develop extreme location-specific conditions. A modification of the numbers of animal and plant species and a shift of the species spectrum result. Sealing affects man psychically and physically by:effects upon health (e.g. heat stress), reduced possibilities for shaping one's living and working environment and recreational possibilities, changing the environment's aesthetics, and separating or modifying cycles of materials, and destroying plant habitat.

In settlements, the age of un-built-up areas or the period available for quasi-undisturbed development (e.g. succession), is important, as shown by inventories of small structures relevant to nature conservation. Thus e.g. old, less "transformed" village areas prove to be "rich", however village areas that are more sealed (and also large settlements or development areas and later individual and terraced house areas) are considered as "impoverished" (Bflr 1988).

Often, soil compaction is responsible for strong damage to or destruction of the plant cover in closely built-up regions (e.g. city centres, peripheral zones). In such areas, nearly without exception, there is a problem with compaction of remaining (small) green and open spaces by pedestrians and/or vehicular traffic. The same happens in loosely built-up areas such as road verges, and frequently to tree-grid sides used for traffic. In addition, even in those parts not actually sealed, one often finds stone chippings or sand/gravel substrates low in fine particulates, e.g., on paths, road verges and car parks. Sport and leisure activities (e.g. football games) cause both extensive soil compaction and the development of characteristic vegetation cover that is full of gaps and poor in species (Bflr 1988). Table 8 lists the effects of city- and sealing-dependent factors on organisms (communities).

Owing to the complexity of the systems and their multiple cross-linkages, a simple derivation of threshold values fixing a general "measure of an ecologically compatible sealing" is scarcely possible. In individual science disciplines (botany, zoology, pedology, climatology etc.) we may be able to find threshold values for specific partial phenomena. For example, investigations from plant sociology show that finding $\sim$ 20–25 % of heat loving species (as a rule neophytes) in the total flora of a closely built-up area should be seen as a clear alarming signal, indicating too strong a sealing of soil and its consequent heating up (Bflr 1988).

The division into "nearly natural", suburban, and urban areas reprises in a general way the increase of anthropogenic effects from a historical view as well as the spatial arrangement. In view of the complex effects of anthropogenic influences that result directly from soil sealing, it is clear that not just one factor but rather the synergistic interactions of a multitude of them are responsible for changes in the biocoenosis (Table 9).

Disturbances in the trophic system affect most strongly the predators at the top of the food pyramid. Here, the bigger species of ground beetles and spiders experience a drastic decline due to the reduction of areas in size. In other cases, strong trophic changes may be driven by (e.g.) ants: their omnivory and other demands on a location may actually lead to urban areas being quite hospitable, and their populations may explode, yielding very strong competition to other taxa.

Phytophagous animals, notably plant suckers (e.g. plant lice or aphids), are favoured by the extinction of regulating predators. The bad physiological condition of urban trees provides additional favourable conditions for these fast spreading pests.

Saprophagous animals such as worms and isopods find somewhat more favourable conditions in cities than in the peripheral area. Frequent mowing of park lawns and the big quantity of reduction of the stocks connected with it may result in densities of individuals scarcely reached in an open landscape. Also the resources of lime and food available in a city provide good living conditions for isopods.

The lack exchange of individuals within isolated urban populations intensifies genetic drift, e.g. in changes of the shell design of the land snail *Cepaea*. Only some animals such as flying insects, birds and bats can actively overcome the urban barriers. However, movement of individuals, and thus genetic mixing, may also take place passively, as in wind drifting of spiders and adhering of

Table 8 Effects of city and sealing-dependent factors on organism communities (modified according to Bflr 1988)

Factors complex urban climate, free area and structure	Effects
dry stress by heating, decline of the relative humidity, reduction and cutting of habitat, isolation and habitat fragmentation, strong increase of "minimal habitats" (crevices, joints, pavement chinks, plant tubs, tree discs and similar)	lower biomass changing of species structure: preferential treatment of light and heat liking species (for example old immigrants from submediterranean, mediterranean regions or new immigrants from oversea countries, = most warmth indicators) formation of "continuous initial organisations" due to permanent amplified care of small areas general species poverty
Factors complex hydrology, canalization:	Effects
water infiltration disabled or impossible, fast removal of percipitation water by canalization and discharge system, lowering device of ground water, oxygen exclusion by concrete and asphalt, concentration of melting salt etc. (for example. in tree plates)	damage respectivily dying of plants (for example street trees) modification in species spectrum of cabbage layer (among other things preferential treatment dryness bearing, halophiluos manners partly) vegetation damages by inundations, heavy metal increase in meadow floors
Factors complex other physical and chemical loads	Effects
Increasing of physical and chemical loads caused by more care and use: frequent chop, cut, trim, herbicide application, heavy metal and PAK-entry, eutrophication (by synthetic fertilizer, dust, dog excrement, organic rests or waste)	preferential nutrient loving (nitrophilous) ubiquitos species preferential treatment of raw floor colonialists (pioneer sappers) and rhizome-geophytes resistance formation and formation by mutagenic effect of pollutants general species poverty, partly atrophy each vegetation

small animals, isopods, worms, snails and insects to leaves and fruit. Soil-living animals are often spread over great distances by the transport of garden and native soils (Bflr 1988).

The microbial potential in urban ecosystems represents an important carrier of transformation and buffering abilities for inorganic and organic contaminants whereas in agricultural soil microflora is to be preferably regarded as a biological component of soil fertility (Machulla 1997). High pollutant concentrations are often found in topsoil: they result from an increase in pollutant levels in urban areas generally, and also from the frequent concentration of rainwater infiltration into unsealed areas, where pollution that has been concentrated by runoff is deposited into the soil. A high content of heavy metals in soils and litter may result in a reduced degradation of litter, soil respiration and mineralization. Moderate loads of such metals may contribute to limiting the microbial biomass and various parameters of its activity.

The degree of loading does not only depend on the type and concentration of the pollutant but equally on the specific physical-chemical conditions of the soil (Filip 1995). For instance in soils

Table 9 Effects of an increasing sealing on the fauna (Bflr 1988)

Species composition	decline of nature space typical manners increase ubiquitous manners foreign infiltration with (synanthropic) manners adapted to the human surroundings especially
Trophic structure	failure of levels in the food pyramid increase of phytophagous, saprophagous und omnivorous manners changed dominance structure
Population dynamics	(partly) missing reproduction isolation, genetic drift
Spreading dynamics	anthropogenic spreading

Table 10 Microbial activity in the joint material of a cobblestone pavement road as compared with a heavily polluted sewage field soil and agriculturally used soils (Wesolek, 2001)

	pH $CaCl_2$	SIR µg C_{mic}/g TS 24 h	FEM µg C_{mic}/g TS	Katalase -	ß-GA µg Salicin/g TS 3 h	C_{org} %	C_{mic}/C_{org} -
Joint material	6,70	2781,00	1448,00	67,40	93,00	4,44	3,26
Ap-Loess	7,40	663,00	243,30	8,90	57,80	2,23	1,10
Ah-Sand	5,30	677,00	300,00	8,80	43,00	5,02	1,35
Ap-Loam	6,30	154,90	1930,00	7,70	64,60	1,64	0,95

IR = substrate induced respiration (measure for the breathable biomass)
FEM = chloroform-fumigation-extraction (measure for the whole biomass)
ß-GA = ß-glucosidaseactivity (measure for the aerobic decomposition of cellulose)
Catalase = activity of catalase (measure for the appearance of active aerobic microorganisms)

near a road the microbial activity may not directly depend on the degree of loading of soil by heavy metals; the basic respiration of micro-organisms is elevated near a road, despite the site having a higher load of heavy metals. A potential cause is the high input of organic and mineral substances near a road. However, the degradability of the plant litter supplied there declines as the heavy metal content rises (Post & Beeby 1996).

Comparative investigations of the ratio between organic biomass (C_{org}) and microbial biomass (C_{mic}.) in the joints of a cobblestone pavement road and in the topsoil of a former sewage field showed significantly increased C_{mi}/ C_{org} ratios (as compared with Ap horizons of agricultural loess and sandy soils). The road soil ha very high values, exceeded only by the Ap horizon of a loamy soil (Table 10). The comparison to Machulla (1997) show a non-uniform picture: both anthropogenic soils show strongly increased values of microbial biomass as compared with the values compiled by Machulla whereas the Ah horizons of loess and sandy soil are within the 95 % confidence interval of the correlations between Cmic. and Corg. determined by Machulla. On the other hand, the Ah horizon of the loamy soil shows a value for the microbial biomass far above the range of the remaining samples (Machulla 1997). It has not yet been investigated whether these increased biomass and activity values are accompanied by an increased degradation efficiency of urban soils for organic contaminants or the biomass input.

Soil Load and Transport of Contaminants

Much existing pollution in older, denser residential settlements resulted largely from periods of uncontrolled waste disposal. Wastes included residues of combustion from heating systems, excrements, refuse, and sewage. On industrial land production-specific wastes or pollution were produced by leakage and accidents (Blume 1992).

Inputs of pollutants and excessive nutrients (urine, dogs' excrements) can accumulate on sealed areas, and then during rainfall large quantities at high concentrations may be discharged into the joints and open spaces of the soils and into the sewerage system. The behaviour and transport of the substances is determined by comparable factors as in natural soils, yet their volume and weighting is different. Factors affecting these differences include higher heterogeneity of substrates, heterogeneous disturbances of the soil structure (e.g. compaction), and heterogeneous pollutants in the substrate and in the emissions.

Some recent papers have dealt with the pollution of urban soils by organic and inorganic pollutants, and the effects of animal excrements (dogs' urine) on the vitality of inner-urban trees (Balder 1994). Summaries are in Blume & Scheuss (1997) and Renger et al. 1998.

The pollutant contents within a profile and a spatial soil unit may frequently vary significantly. It depends primarily on the type of substrate (e.g. debris, rubbish, refuse) and the origin of the pollutant load such as waste water, sewage sludge, road traffic (Renger & Mekiffer 1998; Meuser 1996). Table 11 shows an example of the distribution of heavy metals across depth in a road verge profile.

As to heavy metals the mobility and thus the danger potential (Renger & Mekiffer 1998) will depend on the substrate, soil properties that determine mobility (e. g. pH, clay and humus content, DOC, redox potential), origin of the heavy metals, and nteractions with other cations.

Investigations of the mobility of heavy metals in technogenic substrates show that the total content may vary greatly, without considering substrate-specific content ranges or depth gradients. The pH has the most important effects on mobility. Mobilities of the elements mostly about <1 %, and go higher only in a few cases because the urban substrates investigated react predominantly neutral to weakly alkaline. The sorption of the heavy metals to technogenic substrates may be described well by the Freundlich isotherm. Lead is in general sorbed most strongly, nickel most weakly (Kretschmer et al. 1997).

Increased infiltration on partially sealed and unsealed areas may favor a transfer of pollutants. Wessolek & Facklam (1997) show in general only a low tendency to transferring heavy metals owing to high pH-values but clear differences between the enrichment of pollutants in the joint spaces where road water is infiltrated and below coating materials (i.e., in a passive flow regime).

When assessing the hydrophobic polycyclical aromatic hydrocarbons (PAC), four fractions are distinguished, which affect the transfer in the unsaturated soil zone: (1) freely dissolved, (2) DOM-bound, (3) sorbed to solid phase, (4) bound residues. The relation between these four fractions will determine how mobile PACs are and thus how transferable or biologically available or toxically effective (Renger & Mekiffer 1997).

As in debris and refuse soils, a high potential danger may exist in road verge soils because they often have very high PAC contents (above the interference values of the Berlin list). Figure 10 shows the PAC loading of the joint material of a cobblestone pavement road as a function of depth. It is remarkable that the various solubilities of the PACs (declining from fluorine up to benzo(k)-fluoranthene) have not produced an essentially different transfer picture. A first risk assessment whether pollutants are present below partly sealed areas or in the road verge space and of which type they are may be carried out at least partly by means of visual findings (substrates and respective contaminants) with the aid of the substrate key by Meuser (1996). The detection of the spatial distribution of loads in anthropogenically affected substrates is to be considered as a big problem (Blume & Scheuss 1997).

Also the nutrient concentrations and conversion processes in partly sealed or urban soils differ significantly from those in natural soils. Investigations by Stahr et al. (1997) show for the anions

Table 11 Heavy metals (total content) in a depth profile approx. 1.5 m beside an urban main street (Hanover), transferred sand (modified according to Schneider 1994)

Withdrawal depth (cm)	Horizon description	Lead [mg/kg] Pb	Zinc [mg/kg] Zn	Cadmium [mg/kg] Cd	Copper [mg/kg] Cu	Nickel [mg/kg] Ni	pH-value
0–10	jyAh	172	186	1,2	52	20	6,8
10–20	jyAh	84	95	0,5	25	10	6,9
20–25	jyAh	49	58	0,2	15	6	6,7
25–45	jyC	43	60	0	16	9	7,1
45–65	jyC	117	118	0,1	29	11	7,1
65–84	jyC	26	43	0,1	12	7	6,8
84–100	IIjyC	4	15	0	4	5	6,8
100–120	IIjyC	3	11	0	3	4	6,7
120–160	IIIGo	4	11	0,1	3	4	6,9
160–200	IIIGo	6	13	0,1	3	6	6,8

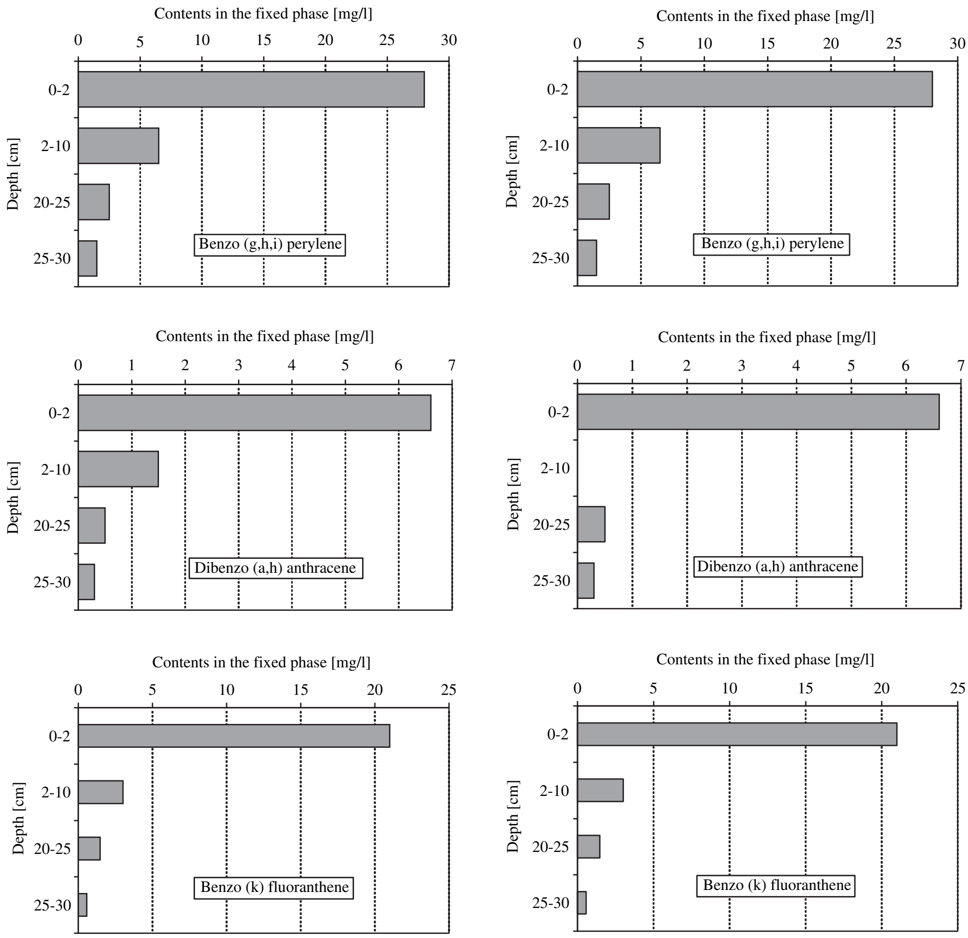

Fig. 10 Easily soluble PAC contents (>1,0 mg/l) (left column) and hardly soluble PAC contents (< 0,0006 mg/l) (right column) in the joint material of a paved road

NO_3, P, B, Mo, S a very wide range between toxic and deficiency situations for plants. The extreme values for phosphorus were 5×, for boron 10× and for sulphur 500× above those of natural soils. Also the nitrate mineralization has clearly changed as compared with natural soils; it may even rise greatly in the case of high humus contents.

The mobility and transfer of organic pollutants below partly sealed areas and in the groundwater zone are determined by two processes: (1) the interactions of pollutant ions and compounds between solid phase and soluble phase (also referred to as retardation), and (2) the transport speed of the substances dissolved depending primarily on the infiltration rate and the field capacity in the unsaturated zone and on the groundwater flowing speed in the groundwater zone.

With the aid of substance transport models, prognoses of long-term pollutant transfer may be prepared if the water balance, the soil physical parameters and the sorption isotherms are known (Stoffregen et al., 1998). If the pollutant quantity in the soil exceeds the soil's "filtering capacity" then one cannot ensure protection of groundwater from toxic pollutants. Given such conditions, then intentional sealing of the soils should be seen positively, because the lost contribution of these areas to groundwater recharge is likely to be smaller than the deterioration of groundwater quality that will happen in their absence.

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Producing and Consuming Chemicals: The Moral Economy of the American Lawn

Paul Robbins, Julie T. Sharp

Abstract The burgeoning application of fertilizers and pesticides to residential lawns, which has begun to offset the gains made in reducing the use of chemicals in agriculture, represents a serious environmental hazard in the United States and elsewhere. Increased use and purchase occur specifically among a sector of consumers who explicitly and disproportionately acknowledge the risks associated with chemical deposition, moreover, and who express concern about the quality of water and human health. What drives the production of monocultural lawns in a period when environmental consciousness has encouraged "green" household action (e.g., recycling)? And why does the production of chemical externalities occur among individuals who claim to be concerned about community family and environment? In this article, we explore the interactions that condition and characterize the growth of intensive residential yard management in the United States. We argue that the peculiat growth and expansion of the moral economy of the lawn is the product of a threefold process in which (1) the lawn-chemical industry has implemented new and innovative styles of marketing that (2) help to produce an association of community, family and environmental health with intensive turf-grass aesthetics and (3) reflect an increasing local demand by consumers for anthentic experiences of community, family, and connection to the nonhuman biological world through meaningful work.

Keywords: political ecology · losins · urban growth consumption

In the United States, the economic boom of the late twentieth century led to an unprecedented level of spending power for the majority of middle-class Americans. The resulting changes in everyday urban practies had profound implications for urban environments in the form of the degradation of the quality of air and water, the disposal of an ever-growing mountain of honsehold waste, and the increases in atmospheric carbon loads through the daily commuting patterns of consumers (Newman and Kenworthy 1996).

One of the most profound though understudied impacts of this growth is the transformation of land cover and ecological change in the wake of urban expansion. Specifically, with a conservative estimate of 23 percent of urban cover dedicated to lawns (Robbins and Birkenholtz 2003) and with 675,000 hectares per year converted to urban development in the United States (Natural Resources Conservation Service 2000), the spread of lawn cover has become a major force for ecological change, blanketing the urban landscapes of the United States (see Fig. 1). Typically, threats to the environment are blamed on agricultural and industrial enterprises. Urban construction and runoff from urban and suburban areas, however, contribute significantly to water pollution, although these

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Fig. 1 Aerial photograph of monocultural lawn space, Columbus, Ohio

sources are often overlooked in regulations (Capiella and Brown 2001; Maddock forthcoming). As a result, one of the most profound but overlooked impacts of changes in land cover has been the spread of the suburban lawn and the concomitant rise in high input systems of management that it demands, including the application of chemical fertilizers, herbicides, and insecticides to residential yards (Jenkins 1994; Robbins, Polderman, and Birkenholtz, 2001).

The scale of the lawn-care industry alone suggests the importance of more thorough analyses. In 1999, Americans spent $ 8.9 billion on lawn-care inputs and equipment. In the same year, 49.2 million households purchased lawn and garden fertilizers, and 37.4 million households purchased insect controls and chemicals. The purchase of these inputs, moreover, steadily increased during the past decade (Butterfield 2000).

This expansion of use and purchase has occurred specifically among a section of consumers who explicitly and disproportionately acknowledge the risks associated with chemical deposition and express concern about the quality of water and human health (Robbins, Polderman, and Birkenholtz 2001). What drives the production of monocultural lawns in a period when environmental consciousness has encouraged "green" household action (e.g., recycling)? And why does the production of chemical externalities occur among individuals who claim to be concerned about community, family and the environment?

To answer these questions we take a political ecological approach to the economic geography of the lawn in the United States. Operating from the understanding that decisions about land management are constrained by “parameters of choice” that are set by larger social, political, and ecological actors and processes, political ecology has been successful in explaining the degradation of agricultural land and water in the global South even while it has demonstrated that global economic and political processes are mediated by local-level ecological realities (Blaikie 1999; Blaikie and Brookfield 1987). This approach we suggest can be equally effective when applied to urban and developed settings. Viewing the use of chemicals in lawn management as the structured environmental decisions of individuals that are embedded in larger social, political, and economic processes at multiple scales, our investigation yields a clearer understanding of the continual and increased use of lawn chemicals by affluent Americans despite widespread knowledge of their possible negative environmental impacts. Moreover, it provides a better picture of the emerging environmental implications of the private consumption and public production of suburban landscapes.

In this article, we explore the interactions that condition and characterize the growth of intensive residential yard management in the United States. We argue that the peculiar growth and expansion of the lawn aesthetic is the product of a threefold cross-scale process in which (1) economic imperatives in the lawn-chemical industry have led to new and innovative styles of marketing that (2) help to produce an association of community, family, and environmental health with intensive turf-grass aesthetics, and (3) reflect an increasing local demand by consumers for authentic experiences of community family and connection to the nonhuman biological world through meaningful work.

Following a brief summary of our methodology we begin our analysis with a demonstration of the distinctiveness of the consumption of lawn chemicals with specific attention to sustained sales of lawn chemicals despite increasing environmental consciousness. This situation points to socioeconomic trends in the industry that demand explanation. In the next section, we briefly examine pressures of production in the agrochemical formulation industry, showing the constraining forces that drive chemical formulator firms to be aggressive in seeking new markets. Next we explore the resulting signification strategies of the industry, showing that formulator marketers work to represent the lawn as a site of community, family, and environment. We then examine the microleyed class-based household and neighborhood processes that drive intensive land uses, demonstrating that the American yard is produced by consumers in an urgent drive to realize the very same things (community, family, and environment). Finally, we argue that the convergence of declining formulator margins, aggressive direct marketing, and social desire together produce the parklike monoculture of intensive lawnscapes, demonstrating the deeply structured nature of the question across scales and troubling some otherwise intuitive notions about the American consumer.

We review the implications of this study in the conclusion. We argue that by examining the question at multiple, mutually constraining locations, we can make the increasingly entrenched normative practices of urban land and water degradation more clear and draw the problematic conceptual divisions between production and consumption and between public and private space into empirical and theoretical question.

Methodology

To explore both the global and local forces at work in the production of the high-input lawn, research required both an industry analysis and a survey and interview component. The industry analysis involved fiscal and historical research on the strategy and logies of lawn-care formulator firms, with attention to their economic performance and the various marketing techniques they use. Specific attention was given to the Scotts Company, the leading marketer of lawn chemicals in the world. With net sales of $ 1.74 billion in fiscal year 2001. Scotts controlled 55 percent of the lawn and garden market in the United States and 25 percent of the market in Europe (Jaffe 1998; Scotts Company 2003).

For the national survey by the Center for Survey Research at Ohio State University during the summer of 2001; researchers conducted 594 phone interviews with respondents across the United States, who were stratified by census region and selected through random digit dialing. The screening sought to identify and interview adults who were “responsible” for the lawn: “a grassy area at the front, the side, or behind your residence.”

These surveys were supplemented with intensive follow-up phone interviews with selected respondents in Ohio, as well as face-to-face interviews with a purposive sample of residents in Columbus, Ohio. The city of Columbus and the state of Ohio more generally represent good sample areas for consumer research on lawns, since they are test markets for major consumer marketing firms and because the headquarters of Scotts, a major lawn chemical formulator and service provider, is located in Ohio. Together, these methods were used to explore and explain the distinctive trends in the production and consumption of lawns and the sustained use of lawn chemicals despite increasing environmental awareness.

Distinctiveness of the Lawn as a Site of Production and Consumption

Intensive lawn maintenance requires the use of diverse inputs, each with specific associated risks. Pesticides pose risks to both human and nonhuman health if they are found in either surface or ground water (Pepper, Gerba, and Brusscau 1996). Excess fertilizers in water supplies lead to the entrophication of water bodies and the contamination of drinking water (U.S. Geological Survey 2001). Common lawn chemicals include both fertilizers and pesticides, a category encompassing insecticides, herbicides, and fungicides. Table 1 shows the major chemicals that were applied on U.S. lawns in 1995, with basic descriptions of the severity and type of risk that they represent according to the U.S. Environmental Protection Agency (EPA).

Although lawn chemicals do not represent an immediate and proximate risk to consumers if they are used properly, the effects of their sustained use over time are less clear. Even Scotts’s CEO admitted that “we cannot assure that our products, particularly pesticide products, will not cause injury to the environment or to people under all circumstances” (quoted in Scotts Company 2002b, 16). As a result, lawn-care chemicals have received increasing attention as a persistent risk. Concern at the EPA has heightened in recent years, and its studies have increasingly pointed to the uncalculated risks of exposure to lawn-care chemicals. Assessments of human exposure have shown that lawn chemicals are more persistent than was previously thought, especially when they are transported into the indoor environment, where their reported laboratory half-lives become relatively meaningless (Nishioka, Brinkman, and Burkholder 1996).

These chemicals track into homes easily and have been shown to accumulate in house dust, especially in carpets, where they are most available for dermal contact and where young children are placed at a significant risk (Lewis, Bond, Fortmann, and Camann, 1991; Lewis, Fortmann, and Camann, 1994; Nishioka, Burkholder, et al. 1996; Nishioka, Burkholder, Brinkman, and Lewis, 1999; Nishioka, Burkholder, Brinkman, and Hines, 1999). Furthermore, the effects of these toxins, especially insecticide neurotoxins, on children is not well understood, but modeled impacts of children’s exposure to chemicals have suggested that serious risks result from normal but persistent exposure (Zartarian et al. 2000). The deposition of common lawn chemicals on clothing during application has also been demonstrated to lead to persistent risks of exposure (Leonas and Yn 1992). Therefore, even apart from the water and energy demands of these high-input systems, intensive lawn management carries with it some dilemmas of chemical use that are serious enough to have engaged the EPA’s attention (Guerrero 1990).

Table 1 Pesticides used on U.S. lawns[a]

Pesticide	Mlb Active[b]	Type	Use	Toxicity (EPA)[c]	Environmental Toxicity
2.4-D	7–9	Systemic Phenoxy Herbicide	General	Slight to High	Birds Fish Insects
Glyphosate	5–7	Nonselective Systemic Herbicide	General	Moderate	Birds Fish Insects
Dicamba	3–5	Systemic Acid Herbicide	General	Slight	Aquatic
MCPP	3–5	Selective Phenoxy Herbicide	General	Slight	NA
Diazanon	2–4	Nonsystemic Organophosphate	Restricted	Moderate	Birds Fish Insect
Chlorpyrifos	2–4	Insecticide Broad-spectrum Organophosphate Insecticide	24-hour Reentry Restricted	Moderate	Birds Fish
Carbarvl	1–3	Wale-spectrum Carbamate Insecticide	General	Moderate to High	Fish Insects
Dacthal (DCPA)	1–3	Phthalate Compound Herbicide	General	Low	Birds Fish

[a]Following Robbins and Birkenholtz (2003)
[b]Millions of pounds of active ingredients used in the United States (U.S. Environmental Protection Agency (EPA) 1996).
[c]Toxicity risks based on the standards of the U.S. EPA (Extension Toxicology Network 2000)

These risks are increasingly well communicated, to the point that formulators worry about the marketability of lawn-care products. For example, William Foley, head of the consumer products division of Scotts Company, reported as early as 1990 that "we're concerned about the overall growing presence of chemophobia in the minds of both the trade and the consumer" (quoted in Cigard 1990, 53). In their forward-looking statements for fiscal 2002, Scotts officials further stated their concern that the public's perception of chemicals could adversely affect business (Scotts Company 2002b). Even so, trends in consumers' use of chemicals show no sign of flagging.

Trends in the Use of Chemicals on High-Input Lawns

In the national survey of U.S. lawn owners, 11 percent of the respondents reported that they had eaten dandelions (*Taraxacum officinale*) from their lawns. This revelation, however astonishing to students of consumer behavior, is not a surprise to major input producers and suppliers, who urge homeowners to treat dandelions more aggressively: "Don't Eat 'Em Defeat' Em" (see Fig. 2). The degree to which the lawn has not yet been fully colonized by intensified, high-input systems marks the extent to which aggressive chemical sales continue to have room to grow and expand to new markets. As a result, the high-input lawn is distinctive in that it represents a case of the expanding use of chemicals, even in the face of the increasing acknowledgment of negative externalities, where all other forms of chemical use are declining.

Fig. 2 "Don't Eat 'Em Defeat 'Em." (Copyright ©2001 The Scotts Company, Marysville, Ohio, reprinted with permission)

Tracking the sales in the yard chemical-formulator industry suggests the degree of recent expansion. Although they do not manufacture the active ingredients in pesticides and fertilizers, formulator firms purchase them from large chemical companies, mix the active ingredients with other solvents, and package and market the final products for retail sale. In this industry, growth is ongoing. Whereas the overall use of agricultural pesticides in the United States has decreased sharply since the late 1970s, as has the use of pesticides in the commercial and governmental sectors, the sales of residential lawn and garden chemicals are stable and increasing. Between 1994 and 1997, national use of agricultural herbicides decreased by 15 million pounds of active ingredient (3 percent of the total) while the use of household herbicides increased by 3 million pounds (or 6.5 percent of the total). Similarly, for insecticides, agricultural usage decreased by 8 million pounds of active ingredient (9 percent of the total) while household usage remained nearly unchanged (Aspelin 1997; Aspelin and Grube 1999). Over the same period, the total retail sales of private lawn-care products increased from \$8.4 million to \$8.9 million, making this segment the largest and fastest-growing one of the entire lawn and garden industry, including ornamental gardening, tree and shrub care, and vegetable gardening (Butterfield 2000). In 1999, 55 percent of U.S. households applied insect controls (some 136 million pounds of pesticides), and 74 percent applied fertilizers (Aspelin and Grube 1999). Even though the total quantity of lawn chemicals applied in the United States remains lower than

Table 2 Chemical deposition: Agriculture versus lawn care, 1997[a]

	Agriculture Deposition 1997 (kg/ha)[b]	Lawn Care Deposition 1997 (kg/ha)	Lawn Care Increase 1982–1997 (kg)[c]
Herbicides	1.612	3.288	2,537,500
Insecticides	0.281	1.141	880,357
Fungicide	0.182	0.537	414,286
Other pesticides	0.566	0.134	103,571
Other chemicals[d]	0.597	4.026	3,107,143
Total	3.238	9.127	7,042,856

[a] Following Robbins and Birkenholtz (2003).
[b] Calculated from total usage (U.S. EPA 1996) by land cover (Natural Resources Conservation Service 2000). Figures are given in kilograms of *active ingredient*. Lawn calculated as 23-percent developed cover.
[c] Calculated from cover change and deposition per hectare.
[d] Chemicals registered as pesticides but often marketed for other purposes, i.e., multiuse chemicals, including sulfur, salt, sulfuric acid, and petroleum products (e.g., kerosene, oils, and distillates).

that applied to agricultural fields, the rate of applications per hectare is far greater (see Table 2). So while total deposition of chemicals applied in agriculture can be estimated to have decreased as a result of land-use conversion, these gains are increasingly offset by the greater use of chemicals on home lawns (Robbins and Birkenholtz 2003).

The environmental and popular presses have increasingly serutinized the conventional yard as a potential environmental problem (Bormann, Balmori, and Geballe 1993; Jenkins 1994; Feagan and Ripmeester 1999). Nevertheless, U.S. homeowners apply chemicals to their lawns even though they understand the negative environmental impacts of doing so. Higher rates of chemical use have previously been shown to correlate with higher incomes, greater levels of education, and high degrees of environmental knowledge and concern (Feagan and Ripmeester 1999; Robbins, Polderman, and Birkenholtz 2001).

In the national survey, the use of a chemical lawn-care company was associated not only with individuals with higher incomes and higher-valued houses, but disproportionately with individuals who believe that homeowners lawn practices and lawn-care services have a deleterious impact on the quality of local water supplies. Moreover, those who use chemicals or chemical application companies are significantly *more likely* to perceive that their own or their neighbors' chemical-use practices have "a negative effect on water quality" than are those who do not use inputs (the survey questions are in Table 3, and the supporting data are in Table 4). Thus, despite greater environmental knowledge and concern, high-energy lawn-care inputs are increasing among the most environmentally sensitive and aware. Therefore, the U.S. lawnscape presents a significant problem, and the increasing production and consumption of these products demands an explanation, as do the complex moral economies that surround them.

Limits to Current Models of Explanation

To date, such questions have been approached largely from an economic perspective, with lawn management described and theorized as individuated, economically rational behavior that is meant to maximize the economic utility of the yard (Templeton, Zilbermand, and Yoo 1998). From this perspective, the use of chemicals has remained steady in the face of wide-spread knowledge of its negative impacts because of (1) broad economic and demographic trends and an increase in the total area of lawns, (2) an increase in the number of people who are likely to use chemicals, and (3) the labor burden that is offset by their use.

In addition to fueling a housing boom, the protracted economic expansion of the late 1990s meant that many Americans had more discretionary income (Cook 1990; Reich 2000), and the low interest

Table 3 Survey questions and response categories

Environment

Impact of home care practices on local water quality
How much do lawn-care practices of home owners, such as the use of fertilizers, weed killers, or bug killers applied to their yards affect water quality near your community? Would you say a lot, some very little, or not at all?
Response Categories: a lot, some, very little, not at all

Impact of lawn-care services on local water quality
How much do companies that provide lawn-care services affect water quality near your community? Would you saw a lot, some, very little, or not at all?
Response Categories: a lot, some, very little, not at all

Community

Impact of neighbors' lawn-care practices on water quality
What kind of effect do you think your neighbors' lawn-care practices have on water quality—a positive effect, a negative effect, or no effect?
Response Categories positive effect, negative effect, no effect

Neighbors' use of lawn chemicals
Do your neighbors use any kind of chemicals on their lawns?
Response Categories yes, no, no neighbors

Knows neighbors by name
How many of your neighbors do you know by name? Would you say . . .
Response Categories all of them, most of them, about half of them, some of them, none of them, no neighbors' don't live in a neighborhood

Family

Learned about yard chemicals from a family member Where did you originally learn about how much to use and how to apply fertilizers weed killers or pesticides?
Response Categories family members, retail salespeople (hardware store for examples, books/magazines from packaging materials instructions included with product, other

Environmental Health

Worry about water supply an index variable
An index variable composed front three survey questions:
Next, I'm going to read von some different statements, and for each one I'd like you to tell me how much you agree or disagree with it
I am concerned that water pollution is a threat to my family's health
I am concerned about pesticides and nutrients in my drinking water
My drinking water is clean
Response Categories; strongly agree, somewhat agree, somewhat disagree, strongly disagree

Willingness to pay for clean water supply
If more tax money were needed to prevent water pollution in your community, about how much more if any, would you be willing to pay per year?
Response Categories: open ended

Socioeconomic Status

Income
And, approximately what was your total household income from all sources, before taxes for 2000?
Response Categories open ended

Education
What is the highest grade or year of school von have completed?
Response Categories: open ended

House value
What is the approximate value of your home?
Response Categories: open ended

Table 4 Status, environment, community, family, environmental health, and lawn chemical use, $N = 594$ (contingency analysis)

	Use lawn-care company		Use do-it-yourself chemicals		Use fertilizers (among chemical users) $N = 271$	
	χ^2	df	χ^2	df	χ^2	df
Status Variables	28.607***	6	15.103**	4	7.466	6
Income						
Education	22.680**	7	12.260	7	14.626*	7
House value	40.078***	5	7.676	5	14.052*	5
Environmental Variables						
Belief that home lawn-care practices have a significant impact on the quality of the local water	15.801**	3	1.350	3	1.427	3
Belief that lawn-care services have a significant impact on the quality of the local water	7.854*	3	6.616	3	1.221	3
Community Variables						
Belief that neighbors' lawn-care practices have a negative impact on the quality of water	6.832*	2	3.319	2	3.410	2
Neighbors use of lawn chemicals	13.452**	2	31.357***	2	3.044	2
Knows neighbors by name	1.423	5	16.621**	5	3.199	4
Family Variable						
Learned about yard chemicals from a family member	–	–	–	–	4.929*	1
Environmental Health Variables						
Worry about home water supply	408	4	4.743	4	13.299**	4
Willingness to pay taxes for clean home water supply	11.495*	4	.842	4	.870	4

* Significant at the .05 level.
** Significant at the .01 level.
*** Significant at the .001 level.

rates of the late 1990s fed the boom in the construction of new single-family homes (see Fig. 3). Simultaneously, there was an increase in the proportion of the lot committed to lawn as ranch-style construction gave way to multistory dwellings (Robbins and Birkenholtz 2003). This increase in gross yard space provided more opportunity for the use of chemicals ("Pesticide Herbicide Innovations" 2000; Reich 2000).

Second, demographic shifts are increasing the number of people who are likely to take yard work seriously enough to use chemicals. Market research has suggested that Americans in their 50s are more likely to work extensively in their yards than are those of any other age (Howell 2000). Disposable income rises dramatically for Americans in their 50s, in particular, and this extra income is often invested in home improvements, including the yard (Assael 1990). People in this cohort are more likely than are others to own homes with yards (Robbins, Polderman, and Birkenholtz 2001). The first baby boomers turned 50 in 1996, bringing the largest group of Americans into their prime yard-management years (U.S. Burean of the Census 1990). As this generation continues to age over the next two decades, the number of people who engage in yard-care activities can be expected to increase dramatically.

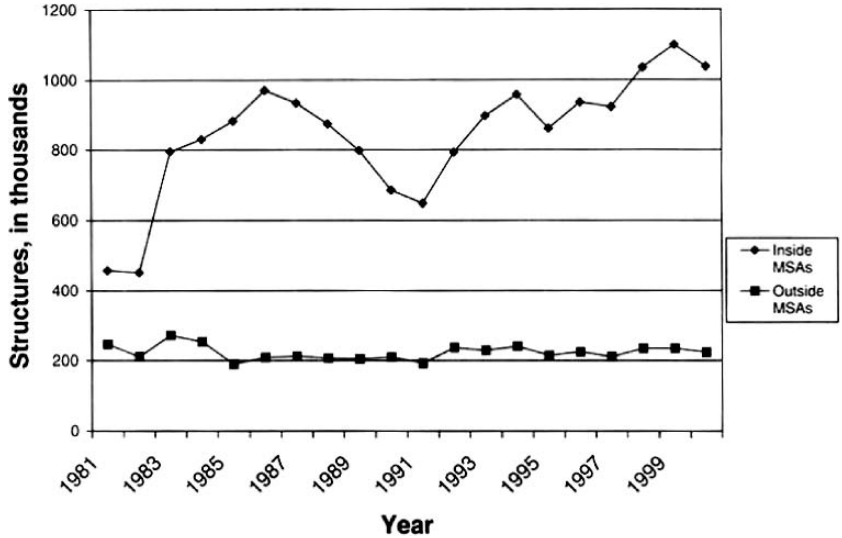

Fig. 3 Single-family housing starts by metropolitan character. *Source*: U.S. Burean of the Census (2001)

Finally, economists and market researchers have asserted that homeowners who engage in yard care tend to use a highly intensive style of management. The geography of new home construction contributes to this intensity, since the majority of new homes that were constructed in the last 20 years were built in metropolitan areas, where residents are more likely to use yard chemicals than people in small towns or rural areas (Cook 1990; Butterfield 2000). In addition, the majority of new homes were built in the southeastern and southern central portion of the United States, where a longer growing season and more virulent insects mean that homeowners have a greater opportunity and incentive to use yard chemicals. These factors, combined with the decreased labor requirements of chemical inputs relative to their manual alternatives, can be used to explain the persistence of the use of lawn chemicals (Templeton, Zilbermand, and Yoo 1998). Thus, many scholars view the continued use of yard chemicals in the United States as being coincident with economic expansion and demographic change and use rational actor models to explain why Americans mow, clip, and spray every Saturday.

These economic explanations, we argue, are insufficient to elucidate fully the sustained and increasing sales of chemicals or to explain the specific cultural economy of the lawn. Describing market behaviors in the absence of a wider economic context, these approaches do not place lawn managers decisions within the social and cultural processes that operate on both the local and global scales. Nor do such explanations posit why specific practices, especially the greater use of chemicals, prevail even in the face of changing public values. They cannot answer why Americans, known for their environmental awareness, continue to manage land in contradiction to their increasing environmental knowledge (Robbins, Polderman, and Birkenholtz 2001).

We suggest, therefore, that a robust explanation of this problem lies in understanding the social construction of the lawn aesthetic within the wider context of the restructuring of the chemical industry at the global scale, as conditioned by residential class and community identity at the local scale. By examining both locations, we find that there is an increasing imperative to produce monocultural lawns and that the demand is created both by formulators who must sell and by local consumers who must buy the monocultural aesthetic of the American lawn. Formulators are forced by declining margins to expand sales aggressively through representational techniques that sign the lawn as part of consumers' identity. Local lawn owners, in contrast, are driven to maintain the monocultural lawn with an urgency rooted in their perceived alienation from community, family, and environment. We consider each process in turn.

The Drive for Markets: Economic Imperatives to Expansion

Over the past 20 years, formulator firms—companies that purchase raw chemical inputs for lawn care, combine them, brand them, and supply them to retail outlets—have faced several new challenges that have forced them to expand and adapt. These challenges have included declines in retail outlets, difficulties in financing, the rising costs of raw materials, and regulatory and ecological limits to production.

First, as in most traditional Fordist industries, the cost of standing warehouse stock, essential in this highly seasonal industry, has become prohibitive for traditional retailers, who rarely want to stock spring merchandise the previous fall. As Scotts executive Charles Berger (quoted in B. Williams 1997, 211) explained. “Nobody wants March merchandise in November.” At the same time, there has been an increasing shift in retail sales toward the cheap prices of mass discount and home improvement stores, which have become the new centers of lawn and garden retailing (Bambarger 1987; Cook 1990).

As a result, hardware stores and nurseries, the traditional outlets for lawn-care products, have lost market share in chemical sales, and formulators have come to rely more heavily on a relatively smaller number of larger-scale customers: home improvement and mass market retailers. Ten North American retailers account for 70 percent of the sales of the Scotts Company, for example. Home Depot, Wal-Mart, Lowe’s, and the increasingly troubled Kmart provide 60 percent of the sales, with Home Depot alone accounting for 28 percent (Scotts Company 2002a). In addition, competition among these retailers is intense. If any of these customers falters, formulators may lose important outlets (Scotts Company 2002b).

At the same time, many formulators are facing increasingly difficult financial situations. Scotts’s leveraged buyout in 1986, as a prominent example, cost $211 million, of which $190 million was financed by an investment banking firm. This debt was blamed for the 11-percent fall in Scotts’s stock share price from June 1999 to June 2000 (“At Scotts They Call It Pull” 2000).

In an attempt to increase sales through acquisition, formulator firms have assumed even more debt with the purchase of new product lines (Baker and Wruck 1991; “Monsanto completes pesticide sales” 1998), resulting in declines in credit ratings. Other expenses in the industry, including company software, the closing of facilities that operate at a loss, severance pay for redundant employees, and costs associated with product recalls, have also driven significant debt (Scotts 2002b). In fiscal year 2000. Scotts spent $94 million on interest payments alone a figure inflated by the high interest rates of that year, requiring Scotts to increase sales to generate a sufficient cash flow (Scotts 2002a).

Formulators must also deal with the increasing costs of raw materials. Scotts sold its professional golf-turf business partly because of the increased cost of raw materials, such as urea and fuel (Scotts Company 2002a). In addition, the costs of most pesticides’ active ingredients have been steadily rising since the early 1980s (Eveleth 1990).

Expiration of patents and environmental regulation further threaten formulators. The patent for glyphosate, for example, the active ingredient in one of Scotts’s best-selling herbicides, expired in September 2000, opening production opportunities and tight competition for market share (Scotts Company 2002b). Neither are formulators immune from the complexities of state regulation. Active ingredients in pesticides can be removed from the market under the Food Quality Protection Act, and companies must be prepared to find new sources of raw materials for all of their pesticides, as well as be prepared for the potential costs of remediation or liability if any pesticide causes harm (Scotts Company 2002b).

In addition, the harvesting of peat for potting soil is subject to environmental regulation in both the United States and Britain. The U.S. Army Corps of Engineers has used Scotts over surface water contamination from its peat-harvesting activities at a New Jersey plant and the company only recently reached an agreement with British environmental officials to close down its peat-harvesting plants at several sensitive sites in the United Kingdom (Scotts Company 2002c). Federal, state and local environmental regulators strictly regulate the disposal of wastes from fertilizer- and pestieide-formulating plants. Scotts has been paying fines and cleanup costs for unlicensed waste disposal and asbestos contamination at several sites in the United Kingdom and Ohio (Scotts Company 2002b).

Moreover, the ecological cycles of the industry pose problems for formulators. The cyclical nature of grass and garden growth means that cash flow is cyclical, since early spring and summer bring the highest sales of lawn and garden products. To meet this demand, fall and winter are the highest production times at formulation plants. Yet fall is the time of the lowest cash flow because receipts from spring and summer sales have not yet been received. As a result, formulators must maintain their highest production at a time when their cash flow is the lowest. This cash-flow "crunch" has implications for financial integrity. The Scotts Company's executives are unsure of their ability to make even minimal debt-service payments in fall and winter when their cash flow is the lowest (Scotts Company 2002b). Because debt service is so crucial, the company must boost sales enough to provide adequate capital even in the fall and winter "crunch."

Changing weather affects formulators, as well. A wet spring slows fertilizer sales but increases pesticide sales. Conversely, a dry spring increases fertilizer sales but decreases pesticide sales, while a cold spring can slow all lawn and garden sales. This is one reason why formulators seek a global market, hedging against a cold spring in the United States with sunny weather in Europe, and vice versa (Scotts Company 2002b).

To pay off their considerable debt; to over-come the rising costs of raw materials, to deal with the possibility of losing an important retail customer, to ensure sufficient funds to acquire new active ingredients, to manage environmental liabilities, to meet winter debt-service obligations, and to compensate for fluctuations in the weather, formulators of the early twenty-first century must increase sales of lawn chemicals and do so at an increasing rate (Robbins and Sharp forthcoming). Fortunately for firms, changes in the retailing environment have led the industry into a new era of consumer advertising, resulting in increased sales, which the industry needs. These direct ("pull") marketing systems depend, to a great degree, on carefully representing the lawn aesthetic to potential consumers.

Signification: Creatively Representing the Lawn Aesthetic

Like most formulators the dominant firm, Scotts, sold yard chemicals prior to 1990 through a "push" strategy, in which incentives were provided to a close network of retailers—usually independent hardware stores, nurseries, and specialty garden stores—who ordered large shipments of merehandise. Products were shipped to stores in the fall and then held by retailers until spring sales, when the retailers were responsible for selling the products, relying on advertising and knowledgeable, motivated sales staffs (B. Williams 1997; Baker and Wruck 1991).

But in 1986, following a leveraged buyout from its parent company ITT, Scotts proceeded with an innovative marketing tactic: "pull marketing" ("Why I Bought the Company" 1989; Baker and Wruck 1991). On the basis of direct advertising to consumers by the formulators themselves, via television, radio, and print advertising, a "pull" approach concentrates on creating demand at the customer level ("At Scotts They Call It Pull" 2000). Rather than rely on a retailer to sell a specific brand, the formulator presents its products by promulgating novel imagery and signifying the lawn (B. Williams 1997).

The move to a pull marketing strategy now extends to new products that attempt to "turn more dirt—branded dirt—into dollars" and to branded plants, sprouted with company products (Scotts Company 2001, 9). Moreover, the current marketing strategy seeks to increase applications per user. "Consider the $900 million lawns category: almost 30% of homeowners are do-nothings! The average do-it-yourselfer still makes fewer than half the recommended product applications each season. If every home-owner made just four applications a year lawns could be a $2.8 billion market!" (Scotts Company 2001. 12). In addition, advice is readily available through toll-free help lines, in-store counselors, extensive web pages, and e-mail reminder services, all backed by a guaranter of satisfaction for markets in both the United States and Europe (Hagedorn 2001; Scotts Company 2001a).

This emphasis on marketing is capital intensive. After purchasing a new pesticide line. Scotts spent twice as much as its former owner to advertise it (Reich 2000). In 1998, with only about half the market share in yard-care supplies. Scotts spent 75 percent of all the advertising dollars in the sector, and in 2002, it purchased $8 million of television airtime (Jaffe 1998; Scotts Company 2002a). In the process, the lawn-care industry has reinvented itself as a directly marketed aesthetic tied to community values: family, nature, and collective good.

Traditional images generally showed manicured yards, menacing magnified pests that inhabited lawns that received poor care, and specific branded compounds that provided solutions. Recent images have followed this approach, but with many new symbols apparent. Specifically, the lawn is increasingly represented as something that transcends personal value to managers of home lawns with collective activities and pride among neighbors much more prominent, as in Fig. 4, where the lawn becomes a community-activity space for parents and children.

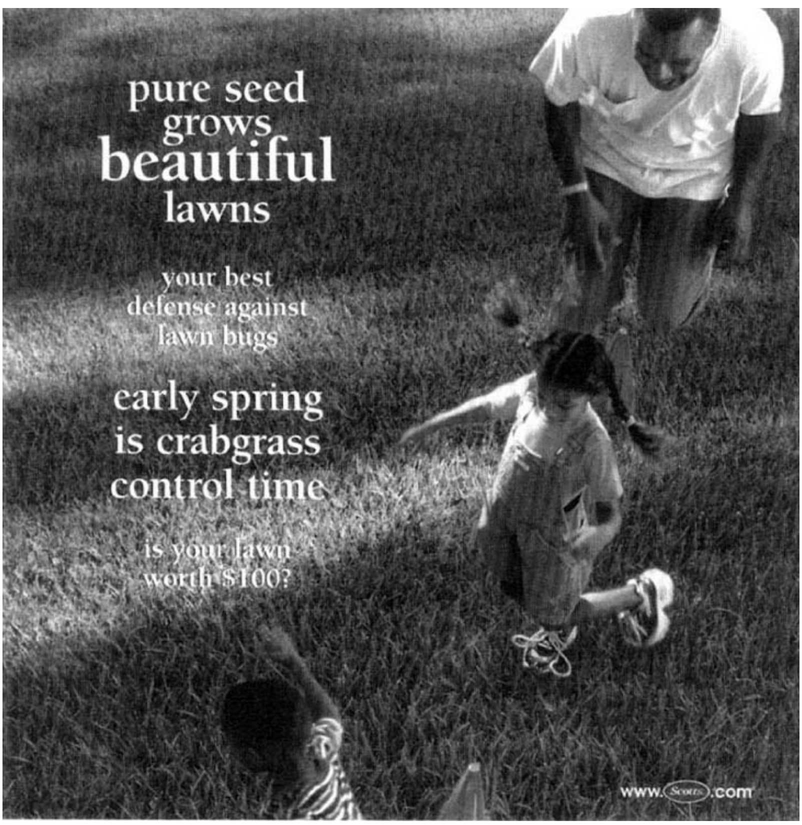

Fig. 4 Advertising the lawn as activity space. (Copyright © 2001. The Scotts Company, Marysville, Ohio, reprinted with permission)

These advertisements reinforce the lawn as a symbol both of collective consumption and of competitive social status. One typical advertisement simply shows a square of turf grass in a saturated green shade, accompanied by the caption, "Not only will your grass be this color, your neighbors will be too." As typically depicted, the monocultured lawn is pictured as an enviously ("green") coveted symbol of community status where the goal is to produce collective social value and to accrue individual merit.

Second, the lawn continues to be depicted as a way to participate in and strengthen the traditional nuclear family. The lawn-advertising images shown in Fig. 4 depict typically young, heterosexual couples working and relaxing amid a healthy sward of monoculture lawn and portray the lawn as a place where children learn to play and share and where turf grass bonds and secures the relationships between fathers and their offspring. Advertisements for lawn chemicals never depict two men or two women, a child with an elderly person, or a group of schoolchildren at play. It is always the nuclear family that enjoys the lawn.

Finally, lawn chemical companies build on and reinforce the sense of lawn management as a bridge to the biotic, nonhuman natural world. Fig. 5 shows the yard as a place in which the suburban homeowner engages in the timeless human activity of planting growing things in the soil. By sowing and nurturing living green plants, these advertisements suggest, modern suburbanites can reconnect with the soil, sun, and water.

Thus, the formulator industry, under pressure to find new markets in an increasingly constricted sales environment has come to produce and reproduce an aesthetic of the lawn that associates the

Fig. 5 Advertising the lawn as family space. (Copyright ©2001 The Scotts Company. Marysville, Ohio, reprinted with permission)

preservation of community, family, and environment with the production of monoculture. The rising lawn-chemical purehases by consumers, in a period when other chemical uses are decreasing coincides with a period of more aggressive and creative marketing.

But to imagine that the formulator industry, by itself is responsible for the increasing consumption of chemicals and the specific aesthetic and social association of lawnscape production is to overlook a highly localized moral economy. Increased advertising budgets cannot explain the specific character of the emerging aesthetic and its appeal, particularly to collective values. To do so requires a reorientation to local-scale analysis and an understanding of the classed character of consumer communities.

Lawn Aesthetics: Classed Expressions of Community Desire

For the most part, yards are not managed for physical resources, such as food, fuel, or shelter, or for environmental purposes like the control of erosion. Instead, yards are managed for other qualities. Specifically, lawn managers overwhelmingly associate their lawns with the preservation of community, family, and environment.

Lawnscape as Participation in Community

The results of the survey indicate that lawn managers who use chemicals are more likely to know their neighbors by name than are those who do not use chemicals (see Tables 3 and 4). Moreover, the majority of the respondents reported that their neighbors used chemicals. The resulting perceived impacts of neighbors' practices are shown in Table 5. While most of the respondents reported that their neighbors' practices had no impact or negative impact on the quality of water most agreed that these practices had a positive impact on both property values, and more significantly, "neighborhood pride."

As a result, intensive lawn management tends to cluster. If one's neighbors use lawn chemicals, then one is more likely to engage in a number of intensive lawn-care practices, including hiring a lawn-care company, using do-it-yourself fertilizers and pesticides, and applying chemicals more frequently if one uses chemicals (see Table 4).

In addition, lawn management, in general, is associated with positive neighborhood relations. People who spend more hours each week working in their yards and report greater enjoyment of lawn work, feel more attached to their local community (see Table 6). Data from the interviews supported the idea that people associate yard work with a sense of neighborly attachment. One 29-year-old male informant explained: "It's nice just to have people on either side of us who are out doing things in their yard, whether they're planting flowers or whatever they're doing.... It's more about the neighborhood. That's kind of the reason why we like where we are."

This sense of community action was supported in follow-up interviews, in which the respondents commonly reported an "obligation" to manage their private lawns intensively, not only in defense

Table 5 Kind of impact resulting from neighbor's lawn-care practices, $N = 594$ (percentage)

	Kind of Impact		
Impact of neighbors lawn-care practices on	Negative	None	Positive
Quality of water	34.0	48.5	11.9
Property values	6.2	41.2	49.8
Neighborhood pride	4.9	18.4	73.1

Note: Rows do not total 100 percent because of missing data.

Table 6 Community attachment and lawn-care practices, $N = 594$ (pearson's correlation coefficients)

	Number of Hours per Week Spent on Lawn Management	Enjoyment of Lawn Work
Sense of attachment to the local community	134**	278***

** Significant at the .01 level.
*** Significant at the .001 level.

of their neighbors' property values, but also in support of "positive neighborhood cohesion." "participation," and "holding the neighborhood together." Thus, yard management is not an individual activity; rather, it is done for social purposes: the production of community.

Lawnscape as Family Space

Second, lawn managers see their yard work as a way to strengthen the traditional nuclear family; family members play an important role in passing on the correct way to maintain a lawn. Among the respondents who used do-it-yourself chemicals of some sort, those who learned about lawn management from a family member were more likely to use chemical fertilizers regularly than were those who learned about lawn management from a family member were more likely to use chemical fertilizers regularly than were those who learned about lawn management from another source (see Table 4). Data from the interviews also revealed the importance of nuclear family relationships, and many respondents explained the importance of the lawn for family life. When asked why he maintained a lawn, the 29-year-old male respondent did not espouse the virtues of higher property values: instead, he suggested that he managed his lawn for his wife: "Jenny [his wife] loves the house and the neighborhood. She just really enjoys going out and being in the sun and pulling weeds. I guess that's why we have the lawn."

Lawnscape as Environment

The third distinctive feature of the production and consumption of lawn is its strong and increasing association as a form of care for the environment. In this regard, chemical users were more likely to worry about the quality of the water and reported a greater willingness to pay for clean water than did nonusers (see Table 4). Lawn managers who seemed the most concerned about the quality of the water were also the most likely to contribute potential pollutants to the water supply in the form of lawn chemicals.

In addition, the respondents often considered working in their yards to be a link between people and the nonhuman world. The belief in yard work as a connection to nonhuman nature was expressed by the respondents, regardless of their lawn-management style. One 47-year-old woman told us: "I really do enjoy being outside. That's really why I do it.... It's fun to watch things grow.... I'm sure when I retire. I'll find some reason to get out there in the lawn and dig around. It just makes you feel better to be out—in the earth." Another woman in her late 50s described her yard work in glowing terms: "I see my yard as... my communion with our environment. It's the way I mark the seasons, the way I reaffirm who I am as a member of the community of earth. I go out and ... benefit from the wonderful fragrance of the fresh air.... The birds just find our yard to be a fabulous feast.... I just love the capacities to ever tune in to the way the world works."

In this way, the practices and aesthetics of lawn managers who use inputs are associated with a concern for community, family, and environment. Herbicides that flow off lawns and represent a risk to the good of the community are seen as fundamental to proper community behavior. Lawn chemicals that are potentially harmful to children and collect in carpet dust are viewed as important for the family. Lawn chemicals with potentially detrimental impacts on the ambient environment are understood as taking care of the environment. Chemical users are more likely to be concerned about their neighbors' values and feelings. They are more likely to get their lawn-management information from family members. They are more likely to be concerned about the quality of water. These local actors thus closely resemble the figures who occupy the frozen advertising images of the formulator industry, whose drive for new markets has led to the direct marketing of just such practices and aesthetics. What does this coincidence tell us?

The Moral Economy of the Lawn: Production and Consumption, Public and Private

The relationship of chemical firms and suburban consumers to one another and to the environment and the apparently contradictory practices of lawn production and consumption that result suggest several conclusions both for explaining environmental behaviors and for interpreting landscapes like the lawn. First, the case exposes the difficulty in identifying and isolating sites of production and consumption, showing the mutual coercion and constitution of chemical formulators and lawn managers. Second, it suggests the porousness of public and private spaces and the degree to which their intermingling is fundamental to contemporary cultural and economic practices, characterized by a moral economy of the lawn.

Producing/Consuming Authentic Community, Family, and the Environment

Materialist theory suggests that consumer desires are forged in the production sector of the economy to maintain levels of surplus (Schnaiburg 1980; Galbraith 1958; Debord 1983). Support for such a view may be drawn from the case of the lawn. Chemical marketers are implicated in the production of desire for the monocultural lawn through unique and powerful marketing programs, essential to the survival of their economic enterprises. Their receipts depend on successfully conveying to nonusers the personal, social, and environmental importance of proper lawn care: formulators produce images that lawn managers consume.

At the same time, however, the drive for the manicured lawn is clearly a result of localized desires for conspicuous performance of class identity: the use of lawn chemicals is positively correlated with household income, level of education, and market value of the home (see Table 4). The creative lawn industry may therefore be seen as simply responding to patterns of locally classed conspicuous consumption whereby the ability to refrain from productive work is demonstrated through rituals that suggest freedom from labor (Gidwani 2000; Veblen 1899). Agrochemicals, it may be argued, are an essential tool in such rituals, allowing homeowners to project a social landscape of laborions hours spent pulling dandelions, removing insects, and reseeding yards. Lawn managers produce the monocultural lawn in the formation of identity.

The identity derived from production of the lawn is of a specific sort, however, one that accrues status as much to community and collectivity as to individuality. Unlike many conspicuously consumed goods, such as automobiles, the lawn carries the moral weight of participating in a greater community or polity, touching on the relationship of the consumers not only to their families and neighborhoods, but to the broader natural world, over which high-input lawn managers express an explicit sense of stewardship.

These specific goals—community, family, and a "green" environment—may be viewed as a triumvirate of alienated desires. As local communities expire, families fragment, and natural environments are lost, the desire for them grows in an increasingly alienated and individuated society that "goes bowling alone" (Putnam 2000). What is more critical, these desires may be seen as reconsumed fetishes, marketed back to increasingly alienated consumers in the form of "natural" products (Smith 1996). In this view, the human need for creativity and productive work, frustrated by the contemporary system of production, makes consumption the last-resort source of personal identity: the lawn is an arena in which alienated homeowners use energy and skill in yard management to recover personal and community identity based on the creation of a specific commodified landscape: the manicured lawn (Miller 1987).

Yet this interpretation requires that one posit an "authentic" moment from which family, community, and nature have been lost. Careful historical and ecological scrutiny do not support the existence of any such condition (Hull, Robertson, and Kendra 2001; Coontz 1992; Botkin 1990). yet the recapture of that moment commands the attention and capital of homeowners who establish monocultural landscapes for personal and collective gain. That urge to restore lost social-environmental relationships is also a goal of the formulators, whose narrowing margins depend on the increased demand for these moral landscapes of collective desire.

Therefore, the production of authentic community, family, and environmental practice must be seen as the fundamental and common cultural and economic currency of *both* lawn managers and chemical formulators. Both are deeply invested in the establishment of this authentic landscape in the discursive-material realm of suburban space, realized as turf grass. Hence, both formulators and consumers collectively produce the larger system in which each is constrained and directed. Local lawn managers produce the moral imperative to invest in and recover collective landscapes, while formulators produce the specific normative and common images of healthy relationships among neighborhoods, children, and the nonhuman world, made to order, providing an imaginary back-drop to which intensive and earnest local practices can be directed.

Production and consumption are in this way entangled through the simultaneous cultural and economic impulses of the suburban landscape. Their distinction is difficult on many levels. Is the lawn consumed or produced by lawn managers? Are its collective community and environmental values consumed or produced in the process of nurturing turf grass? A convincing explanation of the lawn requires that all these things are simultaneously acknowledged.

Privately Producing Public Space

In much the same way that production and consumption are enmeshed in the lawn and made difficult to distinguish, the public and private spheres of producer-consumer activity are also hard to distill. By examining the porousness of this boundary, some otherwise-intuitive notions about the American consumer can be called into question.

Specifically, in surveys or interviews, lawn managers rarely emphasize issues of rights to private property, concepts usually stressed in analyses of alienated, industrial, suburban consumers. Rather, it is the public nature of the front yard that is most often championed. Lawn managers consistently speak of "obligations," "community," and "neighborhood pride," rather than "rights," in their explanation of activities and choices. Private spaces are managed as public goods.

Drawing on research in cultural and political ecology, this kind of action, with its simultaneously redistributive and disciplinary implications, recalls the actions of peasants in the "moral economy." According to Scott (1976), the moral economy is the collective culture of redistributive obligation in which the risks of failed harvests and poverty are spread through extended villages and families by shared harvests and shared poverty. This economy transcends the neoclassical model of rational household action, demonstrating the emergence of a shared culture of collective good. Ecologies are managed with sensitivity to collective, as well as individual, needs.

Although this analysis depends on an explicitly economic rationale of collective good through shared risk, it makes a convincing bridge between private land management and shared-collective good with implications for the lawn. In their consistent, indeed emphatic, insistence on collective good, suburban peasants redistribute public value through private investment in collective monoculture. As seen from the air (see Fig. 1), the unbroken spaces of the lawn do, indeed, suggest community parkland, rather than private holdings.

In the process, the lawn inverts the traditional "tragedy of the commons" scenario (Hardin 1968). Rather than create collective externalities in pursuit of personal gain lawn managers create personal externalities in pursuit of collective gain. The apparent contradiction (why do lawn owners who are environmentally concerned use chemicals?) is resolved if the actions are seen as resolute markers of collective responsibility.

The implications of this moral economy are several. First, they suggest the harsh and disciplinary regimes that prevent the changing of land-management practices at the local level. The reluctance to disintensify inputs or to allow more heterogeneous ecological landscapes (e.g., dandelions) to flourish is not simply a product of individual choices, time, the optimization of resources, or even individual planning and desire. Rather, it is bound up in the social obligations for the production of public space in the form of the lawn, a landscape that is under constant collective scrutiny and carries great moral weight. In this way, it is difficult to reform.

On the other hand, it opens up opportunities for reform of action, consciousness, and change, especially when we consider cases of other ecological practices in the urban system. Recycling, for example, has been criticized for moving from a rational exercise to one with moralistic overtones, an irrational crusade that brainwashes unsuspecting citizens, especially children (Tierney 1996). Yet it is the moral reversal, from an ethic of disposal by which materials disappear into the waste stream, to an ethic of recycling, by which geographic and ecological consciousness follows materials through complex life cycles, that makes radical changes in behavior possible (Ackerman 1997). Therefore, the profoundly moral character of the lawn economy may yet be leveraged to produce changing consciousness and practice in the suburbs. Organic lawn-care options increasingly appear both in do-it-yourself form (organic fertilizers, how-to books, and nonpower mowers), as well as in the commercial lawn-care sector, where dozens of organic lawn-care companies have sprung to life in recent years. Thus, the blurred line between the public and the private breaks both ways, allowing both the colonization of "private" life worlds by economic imperatives traveling in the guise of community and new community visions to disturb the status quo of accumulation. An understanding of the political ecology of the lawn thus enables the critical possibility of uniting *meaningful work* with sustainable suburban landscapes.

Conclusion: Toward a First-World Political Ecology of Meaningful Work

To date, rigorous and extensive research on urban ecology has traveled in several disparate directions. Ecological explorations of the city have been effectively centered on the simultaneous creation of urban and nonurban spaces as sites of *production* (Cronon 1992 and R. Williams 1973 are notable examples). This research has been supplemented with a greater focus on the city as a site of *consumption* of alienated "second" nature (Smith 1996). At the same time, there has been an increasing emphasis on the actions of individuals on *private* land and the aggregation of decision making in the expansion of urban environments (Brabec and Schulte 2002; Capiella and Brown 2001), combined with a wealth of research on ecological management and struggles for justice over the environment of *public* space (Pulido 2000).

This exploration of the lawn, however, suggests the convergence of these themes—production and consumption, public and private—posing a landscape privately produced for collective consumption, balanced between economic imperatives of accumulation and moral imperatives to recover community, family, and environment. Moreover, the lawn may not be unusual in this regard. Sustained attention to urban ecologies may yet reveal the plural character of many urban spaces, from parks and gardens to sidewalks and houses. The results of this study have implications for any such future investigation.

For economic geography, the results suggest that lessons can be learned from political ecology, where local people perform as land managers and local agents are as much producers as they are consumers of landscapes. Furthermore, these local land managers work not only to produce private property for personal gain and satisfaction, but also to create collective moral spaces for public consumption. But homeowners are not the only agents in the production of lawn landscapes and externalities, even if they are the most proximate ones. Rather, they are the local participants in a convergent process of production and consumption (see Fig. 6). As such, they are driven by a range of contextual forces, from the increasing costs of chemical inputs to embodied practices of class status, constrained by a range of truncated opportunities.

For political ecologists, the extension of land-management approaches to the urban First World demonstrate that lawn owners are not necessarily unique specimens of ecology, geography, or history, but can be seen instead as land managers of the most traditional sort, living within a broader *moral economy* (Scott 1976; Chayanov 1986): tending to status as well as production (Gidwani 2000): and seeking collective good, however disciplinary or constraining (Fig. 7). As for their primary-producer counter-parts, lawn owners are ecological managers, under the collective obligations of a community, operating in a class process, although one not solely capitalist in nature (Gibson-Graham 1996).

In addition, as Blaikie and Brookfield (1987) insisted in their discussion of traditional peasant practices, state and academic castigation of such land managers as “ignorant,” “stupid,” and “conservative” misses the point. Insofar as producers operate within boundaries and constraints, “where there is a known set of practices and behavioral responses, it is . . . much easier for the [manager] to

Fig. 6 Advertising the lawn as human/nature space. (Copyright © 2001 The Scotts Company, Marysville, Ohio, reprinted with permission)

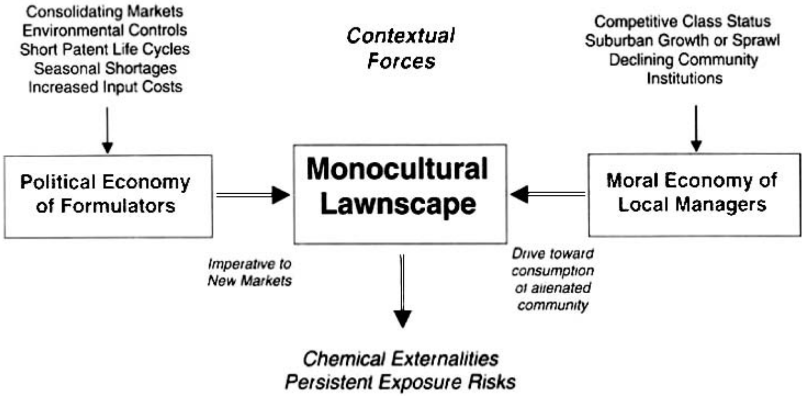

Fig. 7 The political ecology of lawn chemical production and consumption

adhere to an established pattern than to make changes" (p. 35). This is as true for the lawn manager in Ohio as it is for the grower of millet in India. Political ecology unites our understanding of lawns and fields, villages and suburbs.

We have not argued that the position of a subsistence producer, living at the edge of hunger, is the same as that of an affluent suburban dweller. Nor have we argued that both these agents are equally able to deploy alternative strategies for land management. As private producers for public consumption, lawn managers differ significantly from subsistence producers, who arguably publicly produce for private consumption. The visual nature of the lawn landscape, moreover, conveys status through pride of place that would be unknown to many "peasant" producers, who prefer conspicuous poverty and hide their wealth "behind mud walls" (Wiser and Wiser 1989).

We have argued, however, that in either case, whether among subsistence land managers or those of the suburbs, "there is no 'correct' scale for an investigation of land managers and their decisions" (Blaikie and Brookfield 1987, 68). As viewed in a political ecological framework, the discursive-economic rationalities of land managers everywhere are conditioned by the structures of experience that dictate the logics of production. Bemoaning the mindset of individuals is therefore an antisystemic and politically fruitless effort.

Nor have we argued that the goals stated by lawn managers—increased engagement with community family, and the natural world—are in and of themselves problematic or undesirable. There is no doubt that lawns, like other landscape features from parks to porches (Brown, Burton, and Sweaney 1998), actually do produce the effect of putting community members in touch with one another and literally in touch with the soil. These are individuals and communities on a real and profoundly urgent search for *meaningful work*, which will connect them to things, people, and places outside themselves. Even so, we demonstrated that it is the *coupling* of this search with a high-input system of turf management that sets the terms for specific, and potentially pernicious, ecological practices. Through the onmipresent signification practices of input marketing—in which the urgency of public desire is unnecessarily tied to the urgency of constricted chemical markets-community, family, and the biological environment become 2.4-1), glyphosate, and dicamba.

Similarly, we have *not* argued that yard managers' decisions are simply manipulated by chemical company executives. Rather, we have shown that contextual conditions and reduced alternatives constrain the logics of all players, resulting in emergent properties of the system: magnification and acceleration of the persistent chemical hazards. Following Foucault's (1990) "rule of immanence," we suggest that local class processes condition the situation of chemical markets as much as the reverse and that hierarchy of scale is not necessarily hierarchy of causal force. It is the specific power-knowledge relationship between formulators and consumers that produces landscapes, not simply the force of one over the other.

The possibilities for practical progressive policy should operate with these lessons in mind. Assuming that the overuse of lawn chemicals is a problematic practice for both human and non-human health and operating under the normative goal of sustainable urban environments, practical efforts must simultaneously address the structures of chemical supply and the demands of chemical consumption, taking seriously the confounding entanglement of production and consumption, public and private space.

In the case of supply, it is evident that the pressures of chemical production will increase the aggressive promulgation of lawn formulations: even when biotechnologically advanced turf grasses can be produced, they cannot be imagined in the absence of chemical inputs, at least insofar as the producers of seed and the producers of chemicals are the same firms or are contractually linked. Local- and regional-level action to limit the availability of such inputs is viable, however, especially since the private domain of the lawn is viewed generally in such public terms. Therefore, the model of Canadian municipalities offers a possible direction forward. Since 1991, when the Montreal suburb of Hudson outlawed the use of pesticides on lawns, dozens of municipalities have followed suit, enacting bans across the country (Bailey 2002). Although opposed by the landscaping and chemical industries (Carmichael 2002), these bans are broadly supported—precisely because of increased concerns by consumers about environmental conditions, the sustainability of communities, and the health of their families. The driving local forces for consumption may yet be turned to local-level resistance, inverting the social pressures that account for the use of chemicals. Moreover, such a municipal approach stresses the public urgencies associated with private property, departing significantly from "green city" planning, which concentrates on public works and public space.

In the case of demand, the wealth of alternative landscapes that does not require high-input, monocultural management continues to grow. In many communities, violation of local "weed laws," which ban perennial grasses and other alternatives, is increasingly common. Municipalities rarely pursue such cases, since such landscaping decisions rarely represent a credible health risk (Crumbley 2000; Crumbley and Albrecht 2000; Long 1996). Such forms of resistance are increasingly well organized, with groups forming to either coordinate action or simply provide viable landscaping alternatives (Robbins and Sharp forthcoming). They succeed, however, again precisely because they allow homeowners to produce landscapes without necessarily consuming chemicals, inverting the pressures that direct the construction of monoculture, and drawing on people's drive from meaningful work to create new and alternative landscapes.

Therefore, the problem of lawn-chemical externalities, although structured into vast cultures and economies, is not intractable insofar as there are multiple locations of intervention and action. This problem, which continues to pose a technological risk especially for vulnerable populations, can be approached locally, as well as federally. Municipal codes, for example, can be changed, and moral quandaries about the use of chemicals can be more publicly explored, just as lawn products like Diazanon have already received federal regulatory serutiny. The vastness of the "nexus of production relations" (following Yapa 1996) that govern such a problem is therefore a cause for optimism, rather than for political paralysis; environmental challenges become opportunities when they are rendered clear through political ecological analysis. Moreover, such an exploration of the context of daily action opens the door for a hard and personal look at our own situated ecologies, even and especially for those of us in the middle class. These everyday geographies are no more or less than suburban moral economies.

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Streams in the Urban Landscape

Michael J. Paul and Judy L. Meyer

Abstract The world's population is concentrated in urban areas. This change in demography has brought landscape transformations that have a number of documented effects on stream ecosystems. The most consistent and pervasive effect is an increase in impervious surface cover within urban catchments, which alters the hydrology and geomorphology of streams. This results in predictable changes in stream habitat. In addition to imperviousness, runoff from urbanized surfaces as well as municipal and industrial discharges result in increased loading of nutrients, metals, pesticides, and other contaminants to streams. These changes result in consistent declines in the richness of algal, invertebrate, and fish communities in urban streams. Although understudied in urban streams, ecosystem processes are also affected by urbanization. Urban streams represent opportunities for ecologists interested in studying disturbance and contributing to more effective landscape management.

Keywords: impervious surface cover · hydrology · fluvial geomorphology · contaminants · biological assessment

Introduction

Urbanization is a pervasive and rapidly growing form of land use change. More than 75% of the U. S. population lives in urban areas, and it is expected that more than 60% of the world's population will live in urban areas by the year 2030, much of this growth occurring in developing nations (UN Population Division 1997, US Census Bureau 2001). Whereas the overall land area covered by urban growth remains small (2% of earth's land surface), its ecological footprint can be large (Folke et al. 1997). For example, it is estimated that urban centers produce more than 78% of global greenhouse gases (Grimm et al. 2000) and that some cities in the Baltic region claim ecosystem support areas 500 to 1000 times their size (Boland & Hanhammer 1999).

This extensive and ever-increasing urbanization represents a threat to stream ecosystems. Over 130,000 km of streams and rivers in the United States are impaired by urbanization (USEPA 2000). This makes urbanization second only to agriculture as the major cause of stream impairment, even though the total area covered by urban land in the United States is minor in comparison to agricultural area. Urbanization has had similarly devastating effects on stream quality in Europe (House et al. 1993).

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Despite the dramatic threat urbanization poses to stream ecosystems, there has not been a thorough synthesis of the ecological effects of urbanization on streams. There are reviews discussing the impacts of a few aspects of urbanization [biology of pollution (Hynes 1960), physical factors associated with drainage (Butler & Davies 2000), urban stream management (Baer & Pringle 2000)] and a few general reviews aimed at engineers and invertebrate biologists (House et al. 1993, Ellis & Marsalek 1996, Suren 2000), but the ecological effects of urban growth on stream ecosystems have received less attention (Duda et al. 1982, Porcella & Sorensen 1980).

An absolute definition of urban is elusive. *Webster's New Collegiate Dictionary* defines urban as "of, relating to, characteristic of, or constituting a city," where the definition of city is anything greater than a village or town. In human population terms, the U. S. Census Bureau defines urban as "comprising all territory, population, and housing units in urbanized areas and in places of 2,500 or more persons outside urbanized areas," where urbanized areas are defined as places with at least 50,000 people and a periurban or suburban fringe with at least 600 people per square mile. The field of urban studies, within sociology, has a variety of definitions, which all include elements of concentrated populations, living in large settlements and involving some specialization of labor, alteration of family structure, and change in political attitudes (Danielson & Keles 1985). In this review, we rely on the census-based definition, as it includes suburban areas surrounding cities, which are an integral part of many urban ecological studies and represent, in many cases, areas that will develop into more densely populated centers. However, many industrial/commercial/transportation areas that are integral parts of urban and urbanizing areas have low resident population densities, but are certainly contained within our view of urban areas.

Ecological studies of urban ecosystems are growing (McDonnell & Pickett 1990, USGS 1999, Grimm et al. 2000). A valuable distinction has been drawn between ecology in cities versus ecology of cities (Grimm et al. 2000). The former refers to the application of ecological techniques to study ecological systems within cities, whereas the latter explores the interaction of human and ecological systems as a single ecosystem. Although our review focuses on stream ecology in cities, it is our hope that it will provide information of value to the development of an ecology of cities. The goal of this review is to provide a synthesis of the diverse array of studies from many different fields related to the ecology of urban streams, to stimulate incorporation of urban streams in ecological studies, and to explore ecological findings relevant to future policy development. This review is a companion to the review of terrestrial urban ecosystems by Pickett et al. (2001). The review is structured in three parts that focus on the physical, chemical, and biological/ecological effects of urbanization on streams.

Physical Effects of Urbanization

Hydrology

A dominant feature of urbanization is a decrease in the perviousness of the catchment to precipitation, leading to a decrease in infiltration and an increase in surface runoff (Dunne & Leopold 1978). As the percent catchment impervious surface cover (ISC) increases to 10–20%, runoff increases twofold; 35–50% ISC increases runoff threefold; and 75–100% ISC increases surface runoff more than fivefold over forested catchments (Fig. 1) (Arnold & Gibbons 1996). Imperviousness has become an accurate predictor of urbanization and urban impacts on streams (McMahon & Cuffney 2000), and many thresholds of degradation in streams are associated with an ISC of 10–20% (Table 1) [hydrologic and geomorphic (Booth & Jackson 1997), biological (Klein 1979, Yoder et al. 1999)].

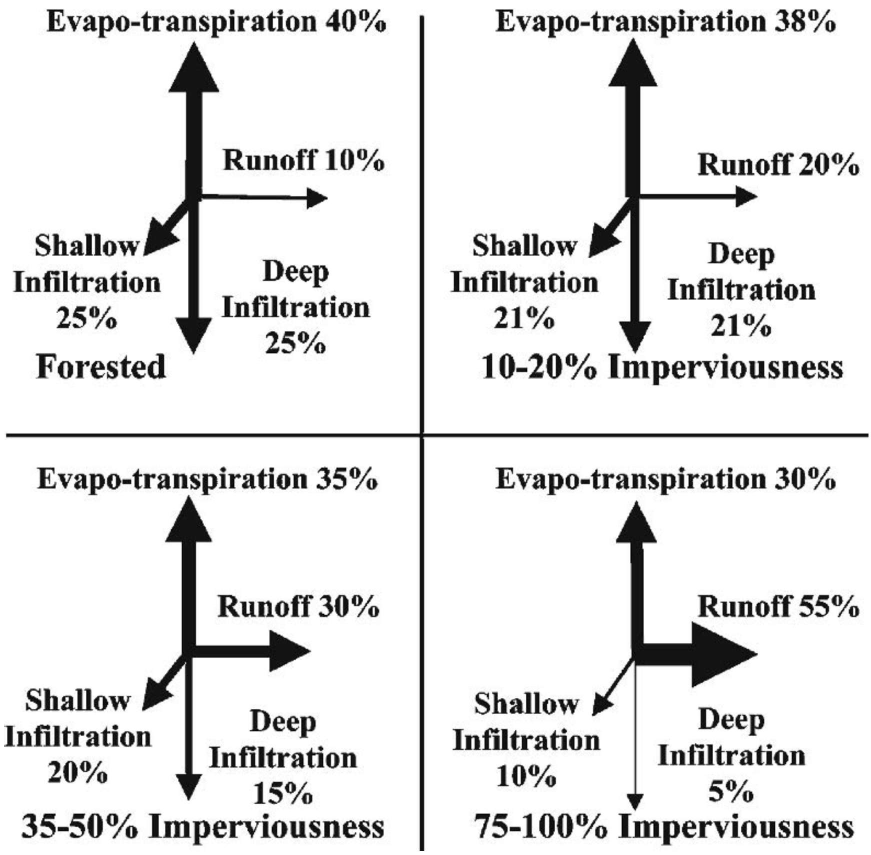

Fig. 1 Changes in hydrologic flows with increasing impervious surface cover in urbanizing catchments (after Arnold & Gibbons 1996)

Various characteristics of stream hydrography are altered by a change in ISC. Lag time, the time difference between the center of precipitation volume to the center of runoff volume, is shortened in urban catchments, resulting in floods that peak more rapidly (Espey et al. 1965, Hirsch et al. 1990). Decreases in flood peak widths from 28–38% over forested catchments are also observed, meaning floods are of shorter duration (Seaburn 1969). However, peak discharges are higher in urban catchments (Leopold 1968). Flood discharges increase in proportion to ISC and were at least 250% higher in urban catchments than forested catchments in Texas and New York after similar storms (Espey et al. 1965, Seaburn 1969). Flood discharges with long-term recurrence intervals are less affected by urbanization than more frequent floods, primarily because elevated soil moisture associated with large storms results in greater surface runoff in forested catchments (Espey et al. 1965, Hirsch et al. 1990). Some exceptions to these observations have been noticed, largely depending on the location of urbanization within a catchment. If the ISC occurs lower in a catchment, flooding from that portion can drain faster than stormflow from forested areas higher in the catchment, leading to lower overall peak flood discharge and increased flood duration (Hirsch et al. 1990). In addition, blocked culverts and drains, swales, etc. may also detain water and lower peak flood discharges (Hirsch et al. 1990).

A further result of increased runoff is a reduction in the unit water yield: a greater proportion of precipitation leaves urban catchments as surface runoff (Fig. 1) (Espey et al. 1965, Seaburn 1969). This reduces groundwater recharge and results in a reduction of baseflow discharge in urban streams

Table 1 Effects of impervious surface cover (ISC) resulting from urbanization on various physical and biological stream variables[a]

Study subject	Findings	Reference
Physical responses: hydrology		
Streams in Texas	Peak discharge increases and lag time decreases with ISC.	Espey et al. 1965
Streams in Pennsylvania	Bankfull discharge increases and lag time decreases with catchment ISC.	Leopold 1968
Review	Surface runoff increases and lag time decreases with increasing ISC (see Fig. 1).	Arnold & Gibbons 1996
Streams in Washington	Increase in bankfull discharge with increasing ISC. At 10%, 2 y urban flood equals a 10 y forested flood.	Booth & Jackson 1997
Physical responses: geomorphology		
Streams in Pennsylvania	Channel enlargement increases with increasing ISC.	Hammer 1972
Streams in New York	Channel enlargement begins at 2% ISC.	Morisawa & LaFlure 1979
Streams in New Mexico	Dramatic changes in channel dimensions at 4% ISC	Dunne & Leopold 1978
Streams in Washington	Channels begin widening at 6% ISC; channels universally unstable above 10% ISC	Booth & Jackson 1997
Physical responses: temperature		
Streams in Washington, DC	Stream temperatures increase with increasing ISC.	Galli 1991
Biological responses: fish		
Streams in Maryland	Fish diversity decreased dramatically above 12–15% ISC and fish were absent above 30–50% ISC.	Klein 1979
Streams in Ontario, Canada	Fish IBI decreased sharply above 10% ISC, but streams with high riparian forest cover were less affected.	Steedman 1988
Streams in New York	Resident and anadromous fish eggs and larvae densities decreased to 10% urban land use and then were essentially absent.	Limburg & Schmidt 1990
Streams in Maryland	Fish diversity decreased dramatically above 10–12% ISC.	Schueler & Galli 1992
Streams in Wisconsin	Fish IBI decreased rapidly at 10% ISC.	Wang et al. 1997
Streams in Ohio	Fish IBI decreased rapidly between 8% and 33% urban land use.	Yoder et al. 1999
Biological responses: invertebrates		
Streams in Maryland	Invertebrate diversity decreased sharply from 1% to 17% ISC.	Klein 1979
Streams in Northern Virginia	Insect diversity decreased between 15% and 25% ISC.	Jones & Clark 1987
Streams in Maryland	Insect diversity metrics moved from good to poor at 15% ISC.	Schueler & Galli 1992
Streams in Washington	Insect IBI decreased sharply between 1% and 6% ISC, except where streams had intact riparian zones.	Horner et al. 1997
Streams in Ohio	Insect diversity, biotic integrity decreased between 8% and 33% ISC.	Yoder et al. 1999

[a]IBI, index of biotic integrity.

(Klein 1979, Barringer et al. 1994). However, this phenomenon has been less intensively studied than flooding, and the effects of irrigation, septic drainage, and interbasin transfers may mitigate the effects of reduced groundwater recharge on baseflow (Hirsch et al. 1990). Baseflow may also be augmented by wastewater treatment plant (WWTP) effluent. The Acheres (Seine Aval) treatment plant, which serves 8.1 million people, discharges 75 km west of Paris and releases 25,000 liters/s during low flow periods (Horowitz et al. 1999), increasing baseflow discharge in the Seine by up to 40% during low flow periods. More strikingly, wastewater effluent constitutes 69% annually and at times 100% of discharge in the South Platte River below Denver, Colorado (Dennehy et al. 1998). In our experience, high percentage contributions of wastewater discharge to urban rivers are not uncommon.

Geomorphology

The major impact of urbanization on basin morphometry is an alteration of drainage density, which is a measure of stream length per catchment area (km/km^2). Natural channel densities decrease dramatically in urban catchments as small streams are filled in, paved over, or placed in culverts (Dunne & Leopold 1978, Hirsch et al. 1990, Meyer & Wallace 2001). However, artificial channels (including road culverts) may actually increase overall drainage densities, leading to greater internal links or nodes that contribute to increased flood velocity (Graf 1977, Meyer & Wallace 2001).

A dominant paradigm in fluvial geomorphology holds that streams adjust their channel dimensions (width and depth) in response to long-term changes in sediment supply and bankfull discharge (recurrence interval average = 1.5 years) (Dunne & Leopold 1978, Roberts 1989). Urbanization affects both sediment supply and bankfull discharge. During the construction phase erosion of exposed soils increases catchment sediment yields by 10^2–10^4 over forested catchments and can be more exaggerated in steeply sloped catchments (Wolman 1967, Leopold 1968, Fusillo et al. 1977). Most of this export occurs during a few large, episodic floods (Wolman 1967). This increased sediment supply leads to an aggradation phase as sediments fill urban channels (Fig. 2). During this phase stream depths may decrease as sediment fills the channel, and the decreased channel capacity leads to greater flooding and overbank sediment deposition, raising bank heights (Wolman 1967). Therefore, overall channel cross-sections stay the same or even decrease slightly (Robinson 1976). Ironically, the flooding associated with aggradation may help attenuate increased flows resulting from increased imperviousness by storing water in the floodplain, temporarily mitigating urban effects on hydrography (Hirsch et al. 1990).

After the aggradation phase sediment supply is reduced and geomorphic readjustment initiates a second, erosional phase (Fig. 2). High ISC associated with urbanization increases the frequency of bankfull floods, frequently by an order of magnitude or, conversely, increases the volume of the bankfull flood (Leopold 1973, Dunne & Leopold 1978, Arnold et al. 1982, Booth & Jackson 1997). As a result, increased flows begin eroding the channel and a general deepening and widening of the channel (channel incision) occurs to accommodate the increased bankfull discharge (Hammer 1972, Douglas 1974, Roberts 1989, Booth 1990). Increased channel water velocities exceed minimum entrainment velocities for transporting bed materials, and readily moveable sediment is lost first as channels generally deepen (Leopold 1973, Morisawa & LaFlure 1979). Channels may actually narrow during this phase as entrained sediment from incision is deposited laterally in the channel (Dunne & Leopold 1978). After incision channels begin to migrate laterally, bank erosion begins, which leads to general channel widening (Booth 1990, Booth & Jackson 1997, Trimble 1997).

During the erosional phase channel enlargement can occur gradually if increases in width and depth keep pace with increases in discharge associated with increasing ISC. In this case the channel enlargement may be barely noticeable (Booth 1990). However, erosion more commonly occurs disproportionately to discharge changes, often leading to bank failure and catastrophic erosion in urban streams (Neller 1988, Booth 1990). In developed urban catchments, as a result of this erosional readjustment phase, the majority of sediment leaving the catchment comes from within-channel erosion as opposed to hillslope erosion (Trimble 1997). The magnitude of this generalized geomor-

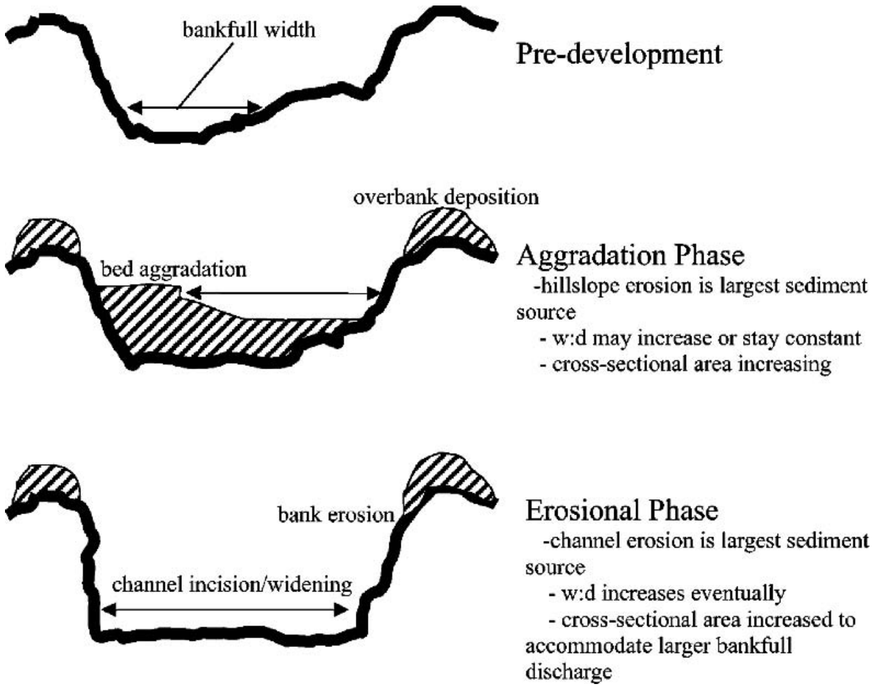

Fig. 2 Channel changes associated with urbanization. During the construction phase of urbanization, hillslope erosion increases sediment supply leading to bed aggradation and overbank deposition. After construction ceases hillslope sediment supply is reduced, but bankfull flows are increased owing to increases in imperviousness. This leads to increased channel erosion as channel incision and widening occur to accommodate increased bankfull discharge

phic response will vary longitudinally along a stream network as well as with the age of development, catchment slope, geology, sediment characteristics, type of urbanization, and land use history (Gregory et al. 1992).

Urban streams differ in other geomorphic characteristics from forested catchments as well. The spacing between pool-riffle sequences (distance between riffles) is generally constant at 5–7 times channel width in forested catchments (Gregory et al. 1994). Generally, this ratio stays constant in urban channels as they widen, which means the absolute distance between pool-riffle units increases, although there is some evidence that this spacing may decrease to 3–5 times channel width (Gregory et al. 1994).

Changes in sediment supply may also alter channel pattern. Increased sediment supply during construction has converted some meandering streams to braided patterns or to straighter, more channelized patterns (Arnold et al. 1982). In the latter case, channelizing leads to increased slope and therefore higher in-stream velocities, especially where artificial channel alteration is carried out to increase the efficiency of the channel in transporting flows (Pizzuto et al. 2000).

Urbanization can also alter sediment texture. Less fine sediment, increased coarse sand fractions, and decreased gravel classes have been observed in urban channels as a result of alteration of sediment supply and altered velocities (Finkenbine et al. 2000, Pizzuto et al. 2000). In addition to sediment changes, large woody debris is also reduced in urban channels. Catchments in Vancouver, British Columbia with greater than 20% ISC generally have very little large woody debris, a structural element important in both the geomorphology and ecology of Pacific Northwest stream ecosystems (Finkenbine et al. 2000).

Other geomorphic changes of note in urban channels include erosion around bridges, which are generally more abundant as a result of increased road densities in urban channels (Douglas 1974).

Bridges have both upstream and downstream effects, including plunge pools created below bridge culverts that may serve as barriers to fish movement. Knickpoints are another common feature of urban channels. These readily erodeable points of sudden change in depth are created by channel erosion, dredging, or bridge construction and are transmitted throughout the catchment, causing channel destabilization (Neller 1988). Other features include increased tree collapse, hanging tributary junctions as a result of variable incision rates, and erosion around artificial structures (e.g., utility support pilings) (Roberts 1989).

Changes in the hydrology and geomorphology of streams likely affect the hydraulic environment of streams, altering, among other things, the velocity profiles and hyporheic/parafluvial dynamics of channels. Such changes would affect many ecological processes, from filter-feeding organisms (Hart & Finelli 1999) to carbon processing and nutrient cycling (Jones & Mulholland 2000).

Temperature

Stream temperature is an important variable affecting many stream processes such as leaf decomposition (Webster & Benfield 1986) and invertebrate life history (Sweeney 1984). Urbanization affects many elements of importance to stream heat budgets. Removal of riparian vegetation, decreased groundwater recharge, and the "heat island" effect associated with urbanization, covered more fully in a companion review (Pickett et al. 2001), all affect stream temperature (Pluhowski 1970), yet very little published data exists on temperature responses of streams to urbanization. In one study on Long Island urban streams had mean summer temperatures 5–8°C warmer and winter temperatures 1.5–3°C cooler than forested streams. Seasonal diurnal fluctuations were also greater in urban streams, and summertime storms resulted in increased temperature pulses 10–15°C warmer than forested streams, a result of runoff from heated impervious surface (Pluhowski 1970). Similar effects on summer temperatures and daily fluctuations have also been observed elsewhere (Table 1) (Galli 1991, Leblanc et al. 1997).

Chemical Effects of Urbanization

Chemical effects of urbanization are far more variable than hydrologic or geomorphic effects and depend on the extent and type of urbanization (residential versus commercial/industrial), presence of wastewater treatment plant (WWTP) effluent and/or combined sewer overflows (CSOs), and the extent of stormwater drainage. Overall, there are more data on water and sediment chemistry in urban streams than any other aspect of their ecology. This is aided by several very large national datasets of stream chemistry that focus in whole or in part on urbanization [e.g., National Urban Runoff Program (United States), National Water Quality Assessment Program (USGS 2001), Land-Ocean Interaction Study (UK) (Neal & Robson 2000)].

In general, there is an increase in almost all constituents, but consistently in oxygen demand, conductivity, suspended solids, ammonium, hydrocarbons, and metals, in urban streams (Porcella & Sorensen 1980, Lenat & Crawford 1994, Latimer & Quinn 1998, USGS 1999). These increases can be attributed to both WWTP effluent and non-point source (NPS) runoff. Many countries have accomplished significant reductions in chemical constituents as a result of adopting better WWTP technologies (e.g., Krug 1993, Litke 1999). However, treatment cannot remove all constituents from wastewater, treatment systems fail, and permitted discharge limits are exceeded. There are more than 200,000 discharges subject to permitting in the United States (USEPA 2001), and of 248 urban centers studied, 84% discharge into rivers (40% of those into rivers with mean annual discharges less than 28 m^3/s) (Heaney & Huber 1984). In addition, CSO systems are still common, in which stormwater and untreated sewage are combined and diverted to streams and rivers during storms. At least 28% of the urban centers mentioned above contained CSOs, and in the United Kingdom 35% of the annual pollutant discharge comes from CSOs and storm drains during less than 3% of the time (Heaney & Huber 1984, Faulkner et al. 2000). In addition, illicit discharge connections, leaking sewer systems, and failing septic systems are a large and persistent contributor of pollutants to urban streams (Faulkner et al. 2000). In the Rouge River catchment in Detroit, Michigan, the focus of an intense federal NPS management program, septic failure rates between 17% and 55% were reported from different subcatchments, and it was estimated that illicit untreated sewage discharge volume at more than $193,000\,m^3/yr$ (Johnson et al. 1999). The ubiquitous nature of small, NPS problems in urban catchments has led some to suggest that the cumulative effect of these small problems may be the dominant source of biological degradation in urban catchments (Duda et al. 1982).

Nutrients and Other Ions

Urbanization generally leads to higher phosphorus concentrations in urban catchments (Omernik 1976, Meybeck 1998, USGS 1999, Winter & Duthie 2000). An urban effect is most often seen in total phosphorus as a result of increased particle-associated phosphorus, but dissolved phosphorus levels are also increased (Smart et al. 1985). In some cases increases in phosphorus can even rival those seen in agricultural catchments both in terms of concentration and yield (Omernik 1976). Even an attempt to understand the agricultural contribution to catchment phosphorus dynamics in a midwestern catchment discovered that urbanization was a dominant factor (Osborne & Wiley 1988). Even though urban areas constituted only 5% of the catchment area and contributed only a small part to the total annual yield of dissolved phosphorus, urban land use controlled dissolved phosphorus concentration throughout the year.

Sources of phosphorus in urban catchments include wastewater and fertilizers (LaValle 1975). Lawns and streets were the primary source of phosphorus to urban streams in Madison, Wisconsin as a result of fertilizer application (Waschbusch et al. 1999). Soils are important in phosphorus dynamics, and the retention of groundwater phosphorus from septic fields affects stream phosphorus concentrations (Hoare 1984, Gerritse et al. 1995). Phosphorus stored in soils as a result of fertilization, however, can be mobilized by soil erosion and contribute to eutrophication of receiving waters. This effect has been called the “chemical time bomb” and is of particular concern when previously agricultural land is cleared for urban growth (Bennett et al. 1999).

Although phosphorus concentrations are elevated in urban streams, the effective increase is not as great as that observed for nitrogen. Urban centers have been shown to increase the nitrogen concentration in rivers for hundreds of kilometers (Meybeck 1998, USGS 1999). Increases have been observed for ammonium as well as nitrate (McConnell 1980, Hoare 1984, Zampella 1994, Wernick et al. 1998). The extent of the increase depends on wastewater treatment technology, degree of illicit discharge and leaky sewer lines, and fertilizer use. As with phosphorus, nitrogen concentrations in streams draining agricultural catchments are usually much higher (USGS 1999), but some have noticed similar or even greater levels of nitrogen loading from urbanization (Omernik 1976, Nagumo & Hatano 2000). Soil characteristics also affect the degree of nitrogen retention, of importance when on-site septic systems are prevalent (Hoare 1984, Gerritse et al. 1995).

Other ions are also generally elevated in urban streams, including calcium, sodium, potassium, and magnesium (McConnell 1980, Smart et al. 1985, Zampella 1994, Ometo et al. 2000). Chloride ions are elevated in urban streams, especially where sodium chloride is still used as the principal road deicing salt. A significant portion of the more than 100,000 tons of sodium chloride applied in metropolitan Toronto annually for deicing enters long-turnover groundwater pools and is released slowly, raising stream chloride concentrations throughout the year (Howard & Haynes 1993). The combined effect of heightened ion concentrations in streams is the elevated conductivity observed in most urban streams. The effect is so common that some have suggested using chloride concentration or conductivity as general urban impact indicators (Wang & Yin 1997, Herlihy et al. 1998).

Metals

Another common feature of urban streams is elevated water column and sediment metal concentrations (Bryan 1974, Wilber & Hunter 1977, Neal et al. 1997, Horowitz et al. 1999, Neal & Robson 2000). The most common metals found include lead, zinc, chromium, copper, manganese, nickel, and cadmium (Wilber & Hunter 1979), although lead has declined in some urban river systems since its elimination as a gas additive (Frick et al. 1998). Mercury is also elevated in some urban streams, and particle-bound methyl-mercury can be high during stormflow (Mason & Sullivan 1998, Horowitz et al. 1999). In addition to industrial discharges, there are many NPSs of these metals in urban catchments: brake linings contain nickel, chromium, lead, and copper; tires contain zinc, lead, chromium, copper, and nickel; and metal alloys used for engine parts contain nickel, chromium, copper, and manganese among others (Muschak 1990, Mielke et al. 2000). All of these metals accumulate on roads and parking lots (Sartor et al. 1974, Forman & Alexander 1998). Many other metals have been found in elevated concentrations in urban stream sediments including arsenic, iron, boron, cobalt, silver, strontium, rubidium, antimony, scandium, molybdenum, lithium, and tin (Khamer et al. 2000, Neal & Robson 2000). Not surprisingly, it appears that NPSs of metals are more important than point sources in urban streams (Wilber & Hunter 1977, Mason & Sullivan 1998).

The concentration, storage, and transport of metals in urban streams is connected to particulate organic matter content and sediment characteristics (Tada & Suzuki 1982, Rhoads & Cahill 1999). Organic matter has a high binding capacity for metals, and both bed and suspended sediments with high organic matter content frequently exhibit 50–7500 times higher concentrations of zinc, lead, chromium, copper, mercury, and cadmium than sediments with lower organic matter content (Warren & Zimmerman 1994, Mason & Sullivan 1998, Gonzales et al. 2000). Sediment texture is also important, and metal concentration in sediments was inversely correlated to sediment particle size in several urban New Jersey streams (Wilber & Hunter 1979). In addition, geomorphic features have been shown to influence metal accumulations. Higher sediment metal concentrations were found in areas of low velocity (stagnant zones, bars, etc.) where fine sediments and organic particles accumulate, whereas areas of intermediate velocities promoted the accumulation of sand-sized metal particles, which can also be common in urban streams (Rhoads & Cahill 1999).

Several organisms (including algae, mollusks, arthropods, and annelids) have exhibited elevated metal concentrations in urban streams (Davis & George 1987, Rauch & Morrison 1999, Gundacker 2000), and ecological responses to metals include reduced abundances and altered community structure (Rauch & Morrison 1999). It is important to note that the route of entry appears to be both direct exposure to dissolved metals and ingestion of metals associated with fine sediments and organic matter. This has led a few researchers to suggest that metal toxicity is most strongly exerted through the riverbed rather than the overlying water (Medeiros et al. 1983, House et al. 1993), although only dissolved metal concentrations in the water column are regulated in the United States.

Pesticides

Pesticide detection frequency is high in urban streams and at concentrations frequently exceeding guidelines for the protection of aquatic biota (USGS 1999, Hoffman et al. 2000). These pesticides include insecticides, herbicides, and fungicides (Daniels et al. 2000). In addition, the frequent detection of banned substances such as DDT and other organochlorine pesticides (chlordane and dieldrin) in urban streams remains a concern (USGS 1999). Most surprising is that many organochlorine pesticide concentrations in urban sediments and biota frequently exceed those observed in intensive agricultural areas in the United States (USGS 1999), a phenomenon observed in France as well (Chevreuiel et al. 1999). Additionally, it is estimated that the mass of insecticides contributed by urban areas is similar to that from agricultural areas in the United States (Hoffman et al. 2000).

There are many sources of pesticides in urban catchments. Urban use accounts for more than 136,000 kg, which is a third of U.S. pesticide use (LeVeen & Willey 1983). They are frequently applied around homes (70–97% of U.S. homes use pesticides) and commercial/industrial buildings and are intensively used in lawn and golf course management (LeVeen & Willey 1983, USGS 1999). Areal application rates in urban environments frequently exceed those in agricultural applications by nearly an order of magnitude (Schueler 1994b). For example, pesticide application rates on golf courses (including herbicides, insecticides, and fungicides) exceed 35 pounds/acre/year, whereas corn/soybean rotations receive less than 6 pounds/acre/year (Schueler 1994b). However, unlike agricultural use, urban pesticide application rates are generally not well documented (LeVeen & Willey 1983, Coupe et al. 2000).

As with metals, the main vector of transport of pesticides into urban streams appears to be through NPS runoff rather than WWTP effluent (Foster et al. 2000). A strong correlation between particle concentration and pesticide concentration was found in the Anacostia River basin in Maryland and the San Joaquin River in California, suggesting NPS inputs are most important (Pereira et al. 1996, Foster et al. 2000). Volatilization and aerosol formation contributed to higher pesticide concentrations, including atrazine, diazinon, chlorpyrifos, p,p′-DDE (a DDT metabolite), and other organochlorines, in precipitation in urban areas and may contribute directly to greater pesticide concentrations and yields in urban areas (Weibel et al. 1966, Coupe et al. 2000).

Other Organic Contaminants

A whole suite of other organic contaminants are frequently detected in urban streams, including polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), and petroleum-based aliphatic hydrocarbons (Whipple & Hunter 1979, Moring & Rose 1997, Frick et al. 1998). PCBs are still frequently detected in urban areas of the United States, even though their use in manufacturing was outlawed because of their carcinogenic effects. These compounds are very stable and are still found in fish at concentrations exceeding consumption-level guidelines in urban rivers such as the Chattahoochee River below Atlanta, Georgia (Frick et al. 1998). PCB concentrations were highly correlated with urban land use in the Willamette Basin in Oregon as well (Black et al. 2000). As with metals and pesticides, PCBs are primarily particle associated, and in the absence of industrial point sources, it is assumed that stormwater runoff is the major route of entry (Foster et al. 2000).

PAHs are a large class of organic compounds that include natural aromatic hydrocarbons but also many synthetic hydrocarbons including organic solvents with different industrial uses (Yamamoto et al. 1997). For this reason, the unnatural PAHs are probably derived from industrial effluent or episodic spills. Very little is known about these compounds in urban streams. In Dallas–Fort Worth, Texas streams, 24 different industrial PAHs were detected, including 4 of the top 10 U. S. Environmental Protection Agency (EPA) most hazardous substances, and at concentrations exceeding human health criteria (Moring & Rose 1997). In Osaka, Japan streams, 55 PAHs were detected, including 40 EPA target compounds. Organic solvents (e.g., toluene, trichloroethane, and dichlorethane) were most common (Yamamoto et al. 1997).

It is difficult to find automobile parking spaces without oil stains in any city. The result of these leaky crankcases is a cornucopia of different petroleum-based aliphatic hydrocarbons in storm runoff associated primarily with particles (Whipple & Hunter 1979). Although there are natural aliphatic hydrocarbons in streams, these are generally overwhelmed by petroleum-based compounds in urban stream bed and water-column sediments (Hunter et al. 1979, Mackenzie & Hunter 1979, Eganhouse et al. 1981). Evidence suggests that these are frequently at concentrations that are stressful to sensitive stream organisms (Latimer & Quinn 1998). Most striking is the yield of these compounds from urban catchments. An estimated 485,000 liters of oil enters the Narragansett Bay each year, a volume equal to nearly 50% of the disastrous 1989 *World Prodigy* oil spill in that same bay (Hoffman et al. 1982, Latimer & Quinn 1998). Similarly, it is estimated that the Los Angeles River alone contributes about 1% of the annual world petroleum hydrocarbon input to the ocean (Eganhouse et al. 1981).

Lastly, recent data suggest pharmaceutical substances from hospital effluent may contribute an array of different chemical compounds into streams. Detectable levels of antiobiotics, genotoxic chemotherapeutic drugs, analgesics, narcotics, and psychotherapeutic drugs have been reported from effluent and/or surface waters (Halling-Sorensen et al. 1998). Although there is some information on the toxicity of these different compounds from laboratory studies, there are insufficient data on the nature or extent of the threat they pose to urban stream biota.

Biological and Ecological Effects of Urbanization

The ecological implications of urbanization are far less studied than the chemical effects, an absence noted in several studies (Porcella & Sorensen 1980, Duda et al. 1982, Medeiros et al. 1983). Nevertheless, much is known about the response of stream organisms, especially invertebrates, to urbanization; far less is known about urban effects on fish (Mulholland & Lenat 1992). Of even greater concern is the lack of mechanistic studies; few studies analyze whether physical habitat, water quality, or food web disturbances (either resource effects or altered community interactions) are the cause of biological degradation in urban streams (Suren 2000). Grossly underrepresented are studies of population dynamics, community interactions, and ecosystem ecology of urban streams, which is surprising given the level of knowledge within the field (Allan 1995). Lastly, very little information has been gathered on biological monitoring of restoration or best management practice implementation in urban catchments (Riley 1998). Most studies assess performance based on stream channel condition or pollutant reduction; few, if any, monitor biological response (Benke et al. 1981, Center for Watershed Protection 2000). In this section, we discuss the effects of urbanization on microbes, algae, macrophytes, invertebrates, and fish.

Microbes

Bacterial densities are usually higher in urban streams, especially after storms (Porcella & Sorensen 1980, Duda et al. 1982). Much of this is attributable to increased coliform bacteria, especially in catchments with wastewater treatment plant (WWTP) and combined sewer overflow (CSO) effluent (Gibson et al. 1998, Young & Thackston 1999). In Saw Mill Run, an urban stream near Pittsburgh, Pennsylvania, fecal coliform colony–forming units (CFU) increased from 170–13,300 CFU/100 ml during dry weather to 6,100–127,000 CFU/100 ml during wet weather (Gibson et al. 1998). CSOs contributed 3,000–85,000 CFU/100 ml during wet weather. These data indicate that non–point sources (NPSs) as well as point sources contribute to fecal coliform loads in urban streams. High values during dry weather are not uncommon in urban streams and may indicate chronic sewer leak-age or illicit discharges. Storm sewers were also a significant source of coliform bacteria in Vancouver, British Columbia; stormwater there contained both human and nonhuman fecal coliform bacteria (Nix et al. 1994). Other pathogens, including *Cryptosporidum* and *Giardia*, have also been associated with CSOs (Gibson et al. 1998).

Increased antibiotic resistance has been seen in some urban bacterial populations (Goni-Urriza et al. 2000). Increased resistance to several antiobiotics, including nalidixic acid, tetracycline, beta-lactam, and co-trimoxazole, has been observed from several enteric as well as native stream species isolated from a river downstream of a WWTP discharge in Spain. It may be that resistant bacteria are passing through the treatment process and conferring resistance to native bacteria. Recent evidence suggests that metal toxicity may also be indirectly involved in increasing antibiotic resistance in stream bacteria. Bacterial resistance to streptomycin and kanamycin were positively correlated with sediment mercury concentration in streams below nuclear reactors and industrial facilities, a result of indirect selection for metal tolerance (McArthur & Tuckfield 2000). Metals may also affect bacterial enzyme activity in urban streams. Enzyme levels were inversely correlated to sediment metal concentration in an urban stream, and this was especially pronounced below an industrial effluent (Wei & Morrison 1992).

Nitrifying bacteria, responsible for the oxidation of reduced nitrogen, are also influenced by urbanization. WWTP effluent can represent a significant source of nitrifying bacteria to urban streams (Brion & Billen 2000). These bacteria are used to oxidize ammonium during the treatment process, but escape into streams in effluent and contribute to the high nitrifier activity observed below some WWTP discharges (Jancarkova et al. 1997). Nitrification rates were as much as six times higher in treated effluent entering the Seine than in receiving river water upstream (Brion & Billen 2000). Ironically, because so many nitrifiers entered the Seine River in France via untreated sewage historically, the reduction in untreated sewage via improved sewage design contributed to a reduction in ammonium oxidation rates in the river from 1.5 μmol/liter/h in 1976 to 1.0 μmol/liter/h in 1993 (Brion & Billen 2000). In addition to nitrifiers, iron-oxidizing bacteria are often abundant in urban streams, especially where reduced metals emerge from anoxic urban groundwater or storm sewers (Dickman & Rygiel 1998).

Algae

The use of algae to indicate water quality in Europe and the United States has a long history (Kolkwitz & Marsson 1908, Patrick 1973). As a result, information exists on algal species and community responses to organic pollution; however, the response of algae to all aspects of urbanization is far less studied. The increasing proportion of urban land use in a catchment generally decreases algal species diversity, and this change has been attributed to many factors including water chemistry (Chessman et al. 1999). Elevated nutrients and light levels typically favor greater algal biomass, which has been observed in many urban streams, where algae do not appear to be nutrient limited (Chessman et al. 1992, Richards & Host 1994). However, the shifting nature of bed sediment in urban streams, frequent bed disturbance, and high turbidity may limit algal accumulation (Burkholder 1996, Dodds & Welch 2000). In addition, several algal species are sensitive to metals, and stream sediment metal accumulation can result in reduced algal biomass (Olguin et al. 2000). Lastly, the frequent detection of herbicides in streams, some with known effects on algae (Davies et al. 1994), will undoubtedly affect stream algal communities

Macrophytes

Little has been written on macrophyte response to urbanization. Most of the work has been done in New Zealand and Australia, where bed sediment changes, nutrient enrichment, and turbidity all contribute to reduced diversity of stream macrophytes (Suren 2000). Exotic species introductions in urban streams have also resulted in highly reduced native macrophyte diversity (Arthington 1985, Suren 2000). Excessive macrophyte growth as a result of urbanization has not been observed in New Zealand, even though nutrient and light levels are higher (Suren 2000).

Invertebrates

Literature searches revealed more studies of urban effects on aquatic invertebrates than on any other group, and the available data are being expanded by groups biomonitoring urban systems (e.g., USGS National Water Quality Assessment, U.S. EPA, state agencies, and others). All aspects of aquatic invertebrate habitat are altered by urbanization. One of the historically well-studied aspects has been the effects of organic pollutants (especially WWTP effluent) on invertebrates. Organic pollution generally reduces invertebrate diversity dramatically, resulting in a community dominated by Chironomidae (Diptera) and oligochaetes (Campbell 1978, Seager & Abrahams 1990, Wright et al. 1995). However, general effects of urbanization on stream invertebrates have also been studied and general invertebrate responses can be summarized as follows: decreased diversity in response to toxins, temperature change, siltation, and organic nutrients; decreased abundances in response to toxins and siltation; and increased abundances in response to inorganic and organic nutrients (Resh & Grodhaus 1983, Wiederholm 1984).

Studies of the effects of urban land use on invertebrates can be divided into three types: those looking along a gradient of increasing urbanization in one catchment, those looking at an urbanized versus a reference catchment, and large studies considering urban gradients and invertebrate response in several catchments. All single catchment gradient studies find a decrease in invertebrate diversity as urban land use increases, regardless of the size of the catchment (Pratt et al. 1981, Whiting & Clifford 1983, Shutes 1984, Hachmoller et al. 1991, Thorne et al. 2000). Decreases were especially evident in the sensitive orders—Ephemeroptera, Plecoptera, and Trichoptera (Pratt et al. 1981, Hachmoller et al. 1991). Most of these studies observed decreases in overall invertebrate abundance, whereas the relative abundance of Chironomidae, oligochaetes, and even tolerant gastropods increased (Pratt et al. 1981, Thorne et al. 2000). Comparative catchment studies show the same trends with increasing urbanization as those observed in single catchment studies: decreased diversity and overall abundance and increased relative abundance of tolerant Chironomidae and oligochaetes (Medeiros et al. 1983, Garie & McIntosh 1986, Pederson & Perkins 1986, Lenat & Crawford 1994).

The multi-catchment studies attempt to relate differing amounts of urbanization in many catchments to particular invertebrate community responses, often using a gradient analysis approach. As discussed above, all find decreases in diversity and overall invertebrate abundance with increased urbanization. This response is correlated with impervious surface cover, housing density, human population density, and total effluent discharge (Klein 1979, Benke et al. 1981, Jones & Clark 1987, Tate & Heiny 1995, Kennen 1999). Klein (1979) studied 27 small catchments on the Maryland Piedmont and was among the first to identify impervious surface cover (ISC) as an important indicator of degradation. Invertebrate measures declined significantly with increasing ISC until they indicated maximum degradation at 17% ISC (Table 1). Degradation thresholds at ISC between 10 and 20% have been supported by numerous other studies for many different response variables (see Schueler 1994a). Residential urbanization in Atlanta, Georgia had dramatic effects on invertebrate diversity, but there were very few clues as to the mechanisms responsible, although leaky sewers were implicated in these and other urban residential catchments (Benke et al. 1981, Johnson et al. 1999).

Few studies have considered specific mechanisms leading to the observed effects of urbanization. This is a difficult task because of the multivariate nature of urban disturbance. Increased turbidity has been associated with higher drift densities of insects (Doeg & Milledge 1991), but more work has focused on the instability of smaller and more mobile bed sediments associated with urban sedimentation. In general, the change in bed sediments favors species adapted to unstable habitats, such as the chironomid dipterans and oligochaete annelids (Pedersen & Perkins 1986, Collier 1995). Where slopes are steeper, and smaller sediments are removed by increased water velocities, localized areas of higher invertebrate diversity are observed within the coarser sediments (Collier 1995). Pools are particularly affected by sediment accumulation in urban streams, and invertebrate communities within these habitats are degraded (Hogg & Norris 1991). Lastly, sedimentation associated with urban streams reduces available refugial space, and invertebrates are more susceptible to drift when refugial space is limited during the frequent floods characteristic of urban environments (Borchardt & Statzner 1990). Storm-flows in urban streams introduce the majority of pollutants and also move the bed sediment frequently. The mortality of *Pteronarcys dorsata* (Plecoptera) in cages in urban streams was attributed to sedimentation associated with storms (Pesacreta 1997).

Sediment toxicity has also been explored. As mentioned above, benthic organic matter binds many toxins and is also a major food resource for many stream invertebrates (Benke & Wallace 1997). Mortality of aquatic invertebrates remains high in many urban streams even during low flow periods, suggesting that toxicity associated with either exposure in the bed or ingestion of toxins associated with organic matter contributes to invertebrate loss (Pratt et al. 1981, Medeiros et al. 1983).

Riparian deforestation associated with urbanization reduces food availability, affects stream temperature, and disrupts sediment, nutrient, and toxin uptake from surface runoff. Invertebrate bioassessment metrics decreased sharply in Puget Sound, Washington tributaries with increasing ISC (Horner et al. 1997). However, streams that had higher benthic index of biotic integrity scores for a given level of ISC were always associated with greater riparian forest cover in their catchment, suggesting that riparian zones in some urban catchments may buffer streams from urban impacts. Above 45% ISC, all streams were degraded, regardless of riparian status. The value of riparian forests is also reduced if the stormwater system is designed to bypass them and discharge directly into the stream.

Road construction associated with urbanization impacts stream invertebrates. Long-term reductions (>6 y) in invertebrate diversity and abundances were observed in association with a road construction project in Ontario (Taylor & Roff 1986). General effects of roads on streams has been reviewed recently (Forman & Alexander 1998).

Very little ecological data beyond presence/absence or abundance data have been reported for urban stream invertebrates. Aquatic insect colonization potential was reported to be high in some urban streams, suggesting restoration efforts would not be limited in this regard (Pedersen & Perkins 1986), but little is known about colonization or adult aquatic insect ecology in urban streams. Urban stream restoration work focuses largely on channel geomorphological stability, with relatively little attention given to biological restoration (Riley 1998), although restoration of Strawberry Creek on the campus of the University of California at Berkeley has resulted in detectable increases in invertebrate diversity and abundance (Charbonneau & Resh 1992). Drift of aquatic invertebrates is a well studied phenomenon in streams, but with one exception (Borchardt & Statzner 1990), little has been published on insect drift in urban streams. We found no published work regarding life cycle ecology (e.g., voltinism or emergence timing), population dynamics, behavioral ecology, community interactions, or production of aquatic invertebrates in urban streams.

Fish

Less is known about fish responses to urbanization than about invertebrates, and a general response model does not exist. However, the Ohio Environmental Protection Agency has a very large database of land use and fish abundance from around their state and has suggested three levels of general fish response to increasing urbanization: from 0 to 5% urban land use, sensitive species are lost; from 5 to 15%, habitat degradation occurs and functional feeding groups (e.g., benthic invertivores) are lost; and above 15% urban land use, toxicity and organic enrichment result in severe degradation of the fish fauna (Table 1) (Yoder et al. 1999). This model has not been verified for other regions of the country, where studies have focused on various aspects of urbanization. Here we consider three types of urban land use studies with regards to fish: gradients of increasing urbanization within a single catchment, comparing an urban and reference catchment, and large, multi-catchment urban gradient studies.

Along urban gradients within single catchments, fish diversity and abundances decline, and the relative abundance of tolerant taxa increases with increasing urbanization (Table 1) (Onorato et al. 2000, Boet et al. 1999, Gafny et al. 2000). Invasive species were also observed to increase in more urbanized reaches of the Seine River, France, and this effect extended more than 100 km below Paris (Boet et al. 1999). Summer storms in that river were associated with large fish kills as a result of dissolved oxygen deficits, an effect also observed for winter floods in Yargon Stream, the largest urban stream in Israel (Gafny et al. 2000). Comparisons with historical collections, an approach used commonly with fish studies, revealed that several sensitive species were extirpated from the Upper Cahaba River system in Alabama between 1954 and 1995, a period coinciding with the rapid growth of Birmingham, Alabama (Onorato et al. 2000). Extirpation of fish species is not uncommon in urban river systems (Ragan & Dietmann 1976, Weaver & Garman 1994, Wolter et al. 2000).

Comparative catchment studies also find dramatic declines in fish diversity and abundances in urban catchments compared with forested references (Scott et al. 1986, Weaver & Garman 1994, Lenat & Crawford 1994). Kelsey Creek, a well-studied urban stream in Washington, is unusual in that it has sustained salmonid populations, especially cutthroat trout (*Oncorhynchus clarki*), even though coho salmon (*Oncorhyncus kisutch*) and many nonsalmonid species have disappeared (Scott et al. 1986). Salmonids in the urban stream actually grow more rapidly and to larger sizes, increasing fish production up to three times that in the forested reference site, presumably a result of warmer temperatures and greater invertebrate biomass in the urban stream. However, the population size structure is different in the two streams, with year 0 and 1 cutthroat underrepresented in the urban stream (Scott et al. 1986).

Large multi-site studies of fish responses to urban gradients also find dramatic decreases in diversity or fish multimetric indices [index of biotic integrity (IBI)] with increasing ISC or other urban land use indicators (Table 1) (Klein 1979, Steedman 1988, Wang et al. 1997, Frick et al. 1998, Yoder et al. 1999). Similar to effects observed for invertebrates, these studies also find precipitous declines in fish metrics between 0 and 15% ISC or urban land use, beyond which fish communities remain degraded (Klein 1979, Yoder et al. 1999). The effect of urbanization on fish appears at lower percent land area disturbed than effects associated with agriculture. In Wisconsin and Michigan few fish community effects were observed in agricultural catchments up to 50% agricultural land use in the catchment (Roth et al. 1996, Wang et al. 1997), and mixed agriculture and urban catchments had significantly lower IBI scores than strictly agricultural catchments (Wang et al. 2000). This suggests that although total urban land use occupies a smaller area globally, it is having disproportionately large effects on biota when compared with agriculture. However, it is crucial to recognize that all urban growth does not have the same effects. Extensive fish surveys in Ohio suggest that residential development, especially large-lot residential development, has less of an effect on stream fishes than high-density residential or commercial/industrial development (Yoder et al. 1999). They hypothesize that riparian protection and less channel habitat degradation are responsible for protecting the fauna in these streams, even up to 15% urban land use. Similar benefits of riparian forests to fish in urban streams were observed in the Pacific Northwest (Horner et al. 1997).

Few studies have explored specific mechanisms causing changes in fish assemblages with urbanization. Sediment is presumably having effects on fish in urban streams similar to those observed in other systems although toxin-mediated impacts may be greater (Wood & Armitage 1997). Road construction results in an increase in the relative abundance of water-column feeders as opposed to benthic feeders, likely a response to a decrease in benthic invertebrate densities (Taylor & Roff 1986). Benthic feeders quickly reappeared as sedimentation rates declined after construction. Flow modification associated with urbanization also affects stream fish. In the Seine, modification of flow for flood protection and water availability has affected pike (*Esox lucius*) by reducing the number of flows providing suitable spawning habitat. With urbanization, the river contains enough suitable spawning habitat in only 1 out of 5 years as opposed to 1 out of every 2 years historically (Boet et al. 1999). Last, WWTP effluent clearly affects fishes. Reductions in WWTP effluent have been associated with the recovery of the fish community in a River Trent tributary near Birmingham, England (Harkness 1982). After nearly 250 years of degradation, effluent reductions, improved treatment, and construction of run-of-the river purification have resulted in an increase in fish diversity and abundances.

A few studies have actually examined ecological factors regulating stream fish populations and communities in urban streams. Recruitment of anadromous fish in the Hudson River Basin in New York was limited by suitable spawning habitat as a result of urbanization (Limburg & Schmidt 1990). Numbers of alewife (*Alosa pseudoharengus*) eggs and larvae in tributary streams decreased sharply between 0 and 15% urban land use. Beyond 15%, no eggs or larvae were found. The Kelsey Creek study discussed above showed impacts on salmonid population structure associated with urbanization, suggesting that urban streams may serve as population sinks for cutthroat, and that fish populations in those streams are dependent on recruitment from source populations with normal population age structures (Scott et al. 1986). Few data on the diet of fish in urban streams have been published, although a shift in diet was observed for fish along an urban gradient in Virginia (Weaver & Garman 1994).

Introduced fish species are also a common feature of urban streams. As a result of channelization, other river transportation modifications, and voluntary fisheries efforts in the Seine around Paris, 19 exotic species have been introduced, while 7 of 27 native species have been extirpated (Boet et al. 1999). The red shiner (*Cyprinella lutrensis*), a Mississippi drainage species commonly used as a bait fish, has invaded urban tributaries of the Chattahoochee River in Atlanta, Georgia where it has displaced native species and now comprises up to 90% of the fish community (DeVivo 1995).

As observed above for invertebrates, real gaps exist in our understanding of fish ecology in urban streams. The effects of urbanization on fishes have focused primarily on patterns of species presence, absence, or relative abundance. We found no published information on behavioral ecology, community interactions, or the biomass and production of nonsalmonid fishes in urban streams.

Ecosystem Processes

Ecosystem processes such as primary productivity, leaf decomposition, or nutrient cycling have been overlooked in urban streams, although they have been extensively studied in other types of stream ecosystems (Allan 1995). A few studies have considered organic matter in streams. WWTP effluent and CSO discharges can dramatically increase dissolved and particulate organic carbon concentrations, especially during storms (McConnell 1980). However, much less is known about baseflow concentrations of particulate and dissolved carbon in urban streams—natural or anthropogenic. The carbon inputs associated with sewage are generally more labile than natural transported organic matter and they affect dissolved oxygen in streams. Oxygen deficits associated with high biological oxygen demand during and after storms are common (McConnell 1980, Faulkner et al. 2000, Ometo et al. 2000). In addition, nonrespiratory oxygen demands associated with chemical oxidation reactions are also elevated in urban streams and can be much higher than biological oxygen demand in stormwater runoff (Bryan 1972). These inputs explain in part why more than 40% of 104 urban streams studied in the United States showed a high probability of greater than average oxygen deficits, with dissolved oxygen concentrations below 2 mg/liter and daily fluctuations up to 7 mg/liter not uncommon (Keefer et al. 1979). In a comparison of 2 forested and 4 urban catchments, average organic matter standing stocks were significantly lower in urban streams near Atlanta, Georgia (Paul 1999). This was attributed to greater scouring of the highly mobile sandy substrates in urban channels as a result of more severe flows.

Organic matter quality has been characterized in a few urban streams. In Kelsey Creek, particulate organic matter (POM) carbohydrate concentrations were higher than in POM in a nearby forested reference stream, suggesting that urbanization affects the nature of transported organic matter as well (Sloane-Richey et al. 1981). In addition to differences in organic matter quantity and quality, urban streams also differ in organic matter retention. Coarse and fine particles released to measure organic matter transport in Atlanta, Georgia streams traveled much farther before leaving the water column in urban streams than in forested streams (Paul 1999). Combined with the data from benthic organic matter (BOM) storage, these data indicate that these urban streams retain less organic matter, a fact that could limit secondary production in these urban streams (Paul 1999).

Ecosystem metabolism has also been measured in a few urban streams. In a comparison of three rivers in Michigan the urban river had higher gross primary production and community respiration than the forested river (Ball et al. 1973). In addition, the gross primary productivity to community respiration (P/R) ratio in the urban river without municipal effluent was greater than the forested stream and greater than 1.0, indicating that autotrophy dominated organic matter metabolism. However, in a downstream reach of the urban river receiving effluent, respiration was higher and the P/R ratio less than the forested river and far less than 1.0, indicating that heterotrophic metabolism predominated. Similar results were observed for urban streams in Atlanta, where gross primary production and community respiration were higher in urban streams than forested streams, and urban streams had more negative net ecosystem metabolism (gross primary production–community respiration), indicating greater heterotrophy (Paul 1999). However, because carbon storage was far less in the urban streams, carbon turnover was faster, supporting the hypothesis that respiration in urban streams was driven by more labile sources of carbon, such as sewage effluent.

Decomposition of organic matter has been measured in a few urban streams. Willow leaves decayed much faster in two suburban New Zealand streams than ever reported for any other stream; this occurred regardless of whether shredding insects were present or absent (Collier & Winterbourn 1986). The same results were observed for chalk maple (*Acer barbatum*) decay in urban streams in Atlanta, where rates were far faster in urban streams than rates observed for any woody leaf species in any stream (Paul 1999). Fungal colonization of leaves was only slightly lower in the urban streams, but there were no shredding insects associated with packs. These results suggested that higher stormflow was responsible for greater fragmentation of leaves in the urban streams, resulting in faster decay rates (Paul 1999).

Removal of added nutrients and contaminants is an ecological service provided by streams and relied upon by society. Although nutrient uptake in flowing waters has been extensively studied in forested ecosystems (Meyer et al. 1988, Stream Solute Workshop 1990, Marti & Sabater 1996), urban settings have been largely ignored. Studies in enriched reaches of river below the effluent from wastewater treatment plants have provided opportunities to examine patterns of denitrification in rivers (e.g., Hill 1979) and seasonal patterns of phosphorus removal and retention in a eutrophic river (e.g., Meals et al. 1999). Recently, ecologists have used the nutrients added by a wastewater treatment plant to measure nutrient uptake length, which is the average distance downstream traveled by a nutrient molecule before it is removed from the water column (Marti et al. 2001, Pollock

& Meyer 2001). Uptake lengths in these rivers are much longer than in nonurban rivers of similar size, suggesting that not only is nutrient loading elevated in urban streams, but also nutrient removal efficiency is greatly reduced. The net result of these alterations in urban streams is increased nutrient loading to downstream lakes, reservoirs, and estuaries.

Opportunities and Imperatives for an Ecology of Urban Streams

Urban streams are common features of the modern landscape that have received inadequate ecological attention. That is unfortunate because they offer a fertile testing ground for ecological concepts. For example, hydrologic regime is a master variable in streams (Minshall 1988), influencing channel form, biological assemblages, and ecosystem processes. As discussed in this review, impervious surfaces result in characteristically altered and often extreme hydrologic conditions that provide an endpoint on a disturbance gradient and that offer opportunities to quantify the relationships between channel form, biological communities, and ecosystem processes (Meyer et al. 1988). Does a continuous gradient of impervious surface cover result in a similar gradient of ecological pattern and process or are there thresholds? Answering that question is of both theoretical and practical interest. Developing a mechanistic understanding of the linkages between urbanization and stream ecosystem degradation is elusive but essential if ecologists hope to understand the nature of ecological response to disturbance and if they want to contribute to the development of scenarios that can guide planning decisions.

Many urban centers developed around rivers, which were the lifeblood of commerce. These commercial uses of rivers ignored and degraded the ecological services rivers provide, a phenomenon continuing today as urban sprawl accelerates. Despite widespread degradation, urban rivers and streams offer local communities an easily accessible piece of nature. Most people live in urban areas, and many children first encounter nature playing in urban streams. Hence, urban streams offer opportunities for ecological outreach and education that ecologists are only beginning to explore. The meteoric rise in numbers of local catchment associations and adopt-a-stream monitoring groups is testimony to an audience eager for ecological insights.

Urban streams also offer ecologists an opportunity to test concepts of system organization through restoration projects. The field of urban stream restoration is dominated by physical scientists and engineers and rarely extends beyond stormwater management and bank stabilization with a goal of reestablishing a channel geomorphology in dynamic equilibrium with the landscape (e.g., Riley 1998). Little attention is given to restoration of a native stream biota or the ecological services streams provide. Urban stream restoration offers challenges not only in integrating physical, chemical, and biological processes to rehabilitate impaired ecosystems, but also requires an attention to esthetics and human attitudes toward the landscape. This offers an opportunity for the integration of ecological and social sciences with landscape design, which if successful will provide an avenue for ecologists to participate in the creation of the sustainable metropolitan centers of the future.

Cities have been a part of human history for millenia, and projections suggest most humans will live in cities in the future. Hence, urban areas lie at the intersection of human and ecological systems. If we are to succeed in that often-stated goal of incorporating humans as components of ecosystems, cities and their streams can no longer be ignored.

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The Urban Climate – Basic and Applied Aspects

Wilhelm Kuttler

Keywords: urban heat island · precipitation · fog · radiation · wind · urban atmosphere · urban climate · climate change · air pollution

Introduction, Definition and Features of Urban Climate

Towns and cities are the most densely populated areas on Earth and will continue to be the artificial landscapes most widely used by the greater part of the Earth's population in the future. In 2030 more than 60% of humans will live in cities. Changes in urban conditions have often caused deterioration in environmental quality and may result in damage to the health of city-dwellers. The differences between the climate of a city and the climate of its surroundings are referred to as the "urban climate". The most important features of urban climate include higher air and surface temperatures, changes in radiation balances, lower humidity, and restricted atmospheric exchange that causes accumulations of pollutants from a variety of sources. Although these changes mainly affect local or regional conditions, persistent substances released into the atmosphere may also affect larger areas or even the global climate.

Causes of Urban Climate

The four main causes of urban climate, which result from different uses of built-up areas, are:

1. replacement of natural soil by sealed surfaces, mostly artificial and having a strong 3-D structure;
2. reduction of the surface area covered by vegetation;
3. reduction of long-wave emission of the surface by street canyons and
4. release of gaseous, solid and liquid atmospheric pollutants, and waste heat.

These factors all have severe impacts on radiation and thermal properties such as evapotranspiration, water storage, and atmospheric exchange in near-surface layers. Properties of urban climates throughout the world are generally comparable. However, the regional and local situation of an urban area, the infrastructure available and local economic structures all modify the local anthropogenic climate (Wienert 2002). Here, we will discuss only the most important characteristics of urban climate, and only for mid-latitude settlements.

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Table 1 Thermal properties of typical "urban surfaces" (asphalt) and "natural bare surfaces" (loamy soil) (after Zmarsly et al. 2002)

Material	Density ρ/kg m^{-3}	Specific heat c/J kg^{-1} K^{-1}	Heat capacity cρ/J m^{-3} K^{-1}	Thermal conductivity λ/W m^{-1} K^{-1}	Thermal diffusivity a/m^2 s^{-1}	Thermal admittance b/J s$^{-0,5}$ m^{-2} K^{-1}
Asphalt	2,100	920	$2.0 \cdot 10^6$	0.75	$0.4 \cdot 10^6$	1,200
Loamy soil (40 % pore space; dry)	1,600	900	$1.4 \cdot 10^6$	0.25	$0.2 \cdot 10^6$	600
Ratio Asphalt/ Loamy soil	1.3	1.02	1.4	3.0	2.0	2.0

Thermal and Hydrological Properties of Urban Surfaces

The thermal behaviour of sealed surfaces is largely determined by the density, heat capacity, thermal conductivity, thermal diffusivity and thermal admittance coefficients of the materials used (see Wesolek, this volume and a summary in Table 1).

The behaviour of sealed surfaces with respect to water drainage and seepage is highly heterogeneous, because porosity and water-bearing properties may fluctuate severely as a function of capillarity and pore volume. The large-scale use of impermeable materials for the almost complete sealing of urban surfaces normally means that precipitation is drained rapidly through underground sewers which are protected against evaporation, and that exposed surfaces are only wetted for a very short time. As a result of reduced evaporation, more energy is available for long-wave emission, sensible heat flux and conduction to the subsurface, and latent heat flux of evaporation is severely reduced.

Structure of Urban Atmosphere and Wind Conditions

The 3-D relief of cities multiplies effective surface area. The aerodynamic properties of surfaces can be characterized by the roughness length (z_0) and the zero plane displacement (d_0), which together measure the unevenness of a surface with respect to the surface's effects on the horizontal and vertical wind vectors. Although disturbances of the near-surface boundary layer can be measured at up to 500 m above ground in densely built-up areas, a largely undisturbed wind field is usually found at 400 m in suburbs and at 300 m in the surrounding areas, both of which are less intensively developed.

The structure of the urban atmosphere depends on the type, size and arrangement of obstacles to air flow and the resulting mechanical and thermal turbulence. As air flows from the countryside to the city it encounters a new and very different set of boundary conditions. That is why during calm and clear weather conditions an internal boundary layer develops downwind from the leading-edge of the city. The *urban boundary layer* is a local to meso-scale phenomenon whose characteristics are governed by the nature of the general urban 'surface'. Beneath roof-level is the *urban canopy layer*, which is produced by micro-scale processes operating in the streets ('canyons') between the buildings (Fig. 1). Its climate is an amalgam of microclimates each of which is dominated by the characteristics of its immediate surroundings. In the case of severe convection in the daytime, the total thickness of the urban boundary layer may reach several hundred metres. During the night,

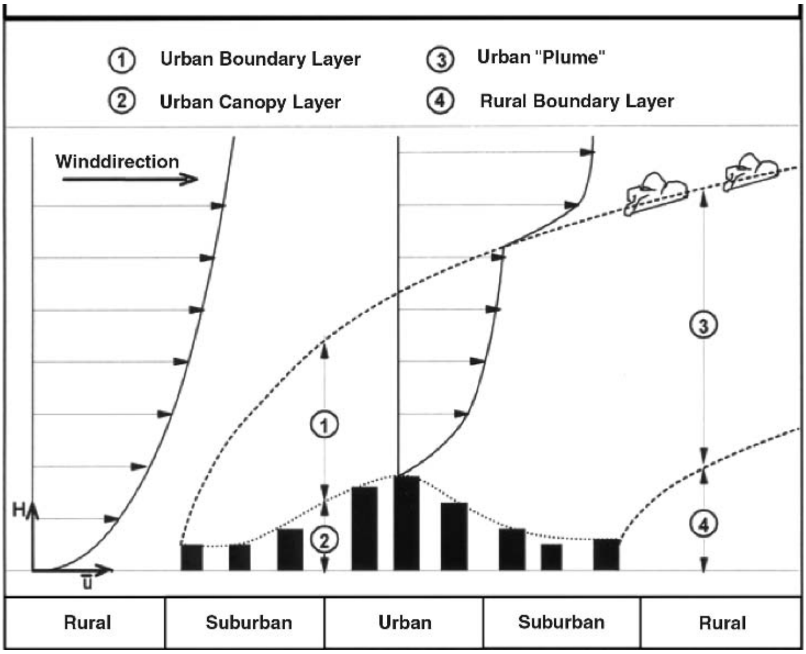

Fig. 1 Features of the urban boundary layer (adapted from Kuttler 1997, after Helbig 1987)

the layer is only a few decametres thick. In these weather conditions, radiation inversions may form; in the surrounding area, these inversions may be more intensive because of severe cooling and the lack of heat emitted by the urban fabric. On the lee side of a city, the urban plume may extend several km downwind, resulting in conditions similar to the urban climate, but less pronounced.

Normally, urban windspeed is lower than that of the countryside. But the surface-level windspeed may be significantly higher in built-up areas than in surrounding areas during calm and cloudless weather conditions (especially at night and in the early morning) when Urban Heat Island ("UHI") values are high and the overall airflow is in transition. The transitional speed depends on the size and structure of the city. Such thermally-generated airflows are called country breeze systems, and are characterized by a relatively low-speed (and normally intermittent) air flow in a layer of a few metres above the surface. In ideal conditions, this country breeze flows radially into the city, supplying cold air to the urban area (Barlag & Kuttler 1990, 1991). The pressure gradient required for air flow between the surrounding area and the city is created by differences in heating between city and surrounding areas. These wind systems can be relevant for urban planning if they improve the air quality in the city, for example by eliminating thermal stress and facilitating renewal or exchange of the urban atmosphere (Weber & Kuttler 2003). However, this is only possible if cold air from the surrounding area can flow freely into the city centre (Weber & Kuttler 2004). General features of such urban air paths (UAP) are (after Matzarakis & Mayer 1992):

- $\mathbf{z_0} < 0.5\,\mathrm{m}$
- $\mathbf{d_0}$: negligible
- length $\geq 1.000\,\mathrm{m}$
- width $\geq 50\,\mathrm{m}$ (depends on lateral obstacles)
- width of obstacles within UAP is 2 – 4 times the height of the lateral obstacles (min. 50 m)
- height of obstacles within UAP $\leq 10\,\mathrm{m}$

- orientation of largest width of obstacles within UAP should be parallel to axis of UAP
- single obstacles within UAP should have a ratio of height of obstacle to horizontal distance between two successive obstacles of 0.1 to 0.2

Urban air paths can be classified into ventilation paths (having different thermal regimes and air pollution levels), clean air paths (differing thermal levels, and without air pollution), cold air paths (differing air pollution levels, without thermal differences) and bioclimatic clean and cold air paths (without thermal differences and without air pollution). Examples of urban air paths (after Kuttler 2000) include:

MAIN TRAFFIC ROUTES

- low z_0 – values
- relatively warm surfaces
- possible high shearing stress
- potential air pollution by cars and domestic heating
  - Evaluation: only ventilation paths because of potential air pollution

RAILWAY TRACKS

- low z_0 – values
- relatively cool surface
- small shearing stress
- air pollution if Diesel-engines operate in goods depots
  - Evaluation: Clean air paths/Bioclimatic clean and cold air paths if Diesel-engines don't operate in goods depots

VEGETATED AREAS/PUBLIC PARKS

- z_0-, d_0 – values depend on vegetation height and density
- relatively cool surface, stable atm. conditions near the ground
- no release of anthropogenic air pollutants, but possibly of
  - biogenic emissions (VOC)
- filtering aerosols and gases
  - Evaluation: Bioclimatic clean and cold air paths with regard to prevailing low vegetation height

RIVERS and URBAN WATERS

- very low z_0-values
- relatively warm surface
- no release of anthropogenic emissions
- absorber of atm. aerosols and gases
  - Evaluation: Clean air paths; thermal effects can be reduced because of relatively warm waterbody

Radiation and Heat Balance

Compared with the surrounding countryside, the radiation and heat balances of an urban area are subject to a wide variety of effects. These are caused by gaseous, particulate and liquid air pollutants, which reflect, scatter and absorb radiation, and the type, structure, use and exposure of urban surfaces. The short and long-wave albedo long wave emission, effective radiation and thermal behaviour of the surface are determined by the composition of the urban atmosphere and the city's surface conditions. During low advection conditions without precipitation, the radiation and heat balance of the ground/air boundary layer is:

$$Q^* + q_m + q_a + q_g + q_l + q_s = 0 \quad (\text{all in W m}^{-2})$$

wherein:
Q^* = radiation balance /W m^{-2}
q_m = metabolic heat flux density / W m^{-2}
q_a = artificial heat flux density /W m^{-2}
q_g = heat flux density in the ground /W m^{-2}
q_l = turbulent latent heat flux density /W m^{-2}
q_s = turbulent sensible heat flux density /W m^{-2}

Conservation of energy demands that the total of the individual components of the radiation *and* heat balance must be zero. The radiation balance (Q^*) is:

$$Q^* = (I + D)(1 - \rho_s) - A(1 - \rho_l) + E/\text{W m}^{-2}$$

wherein:
I= direct solar radiation / W m^{-2}
D= diffuse solar radiation / W m^{-2}
ρ_s = short-wave reflectivity coefficient of surface / -
ρ_l = long-wave reflectivity of surface / -
E= long-wave counter radiation / W m^{-2} , and
$E = \sigma \cdot \varepsilon \cdot T_0$
A= long-wave radiation from the ground / W m^{-2}
σ = Stefan-Boltzmann constant ($5.67 \cdot 10^{-8}$ W m^{-2} K^{-4})
ε = emissivity / -
T_0 = surface temperature / K

By convention, radiation and energy flux densities are positive if they are directed towards the surface being considered. Because the urban atmosphere is normally polluted, the ratio of direct (I) to indirect solar radiation (D) is low. The reflected short-wave radiation intensity is a function of sun elevation angle (γ) and the characteristics and exposure of city surfaces. For European and North American cities, the usual value of short-wave reflectivity is ~10 %. Long-wave reflectivity (ρ_l) reaches ~5 % and is normally not taken into consideration, given the error in determining the other factors. The long-wave *radiation flux densities* expressed in the second part of the above equation are largely determined by surface and air temperatures, air humidity and CO_2 content and the corresponding emissivity values (ε). The long-wave counter-radiation (E) is affected by the atmosphere and very strongly by the raising of the horizon (i.e., the artificial "horizon" seen from within the city – that is, building roof-lines), which can be expressed in terms of the relationship between the width of roads and the height of road-edge buildings. In the case of high ratios, i.e. wide roads and low buildings, about 90 % of the energy input is still available in suitable weather conditions. With low ratios (e.g. 0.5) < 30 % of the energy is emitted. As regards long-wave radiation flux densities, cities have higher counter-radiation values (more intense greenhouse effects) and greater emission from the ground.

The *urban energy balance* includes the factors q_m, q_a, q_g, q_l and q_s. Metabolic heat production (q_m) comes from living organisms (in cities, mostly humans). Since q_m is quite small relative to the other flux densities, it is normally not taken into consideration (although it is about 100 watts/person).

Artificial heat production (q_a), from vehicles, power stations, industrial plants, domestic heating systems, air conditioning systems, and the like can reach widely variable, and sometimes significant, values depending on the city's geographical location and topographical situation. For easier handling, the ratio proposed by Bowen (Bo) is often used to characterize the main factors q_s and q_l which describe the climate of a city in thermal terms:

$$\mathrm{Bo} = \frac{q_s}{q_l}$$

Surfaces with less evaporation emit energy mainly via q_s and have Bo-values > 1, whereas values <1 are reached where turbulent latent heat flux predominates. The Bowen ratio allows a detailed classification of areas that have different land uses with respect to the predominant heat transfer mechanism. On this basis, negative values are only found over green spaces, early in the morning, in the evening and especially at night. During the daytime, both the sensible and latent heat fluxes are mainly directed away from the surface. During temperature inversions, heat transfer direction may be reversed, giving negative Bo-values. Because of the heat island effect, inversions are relatively rare in urban areas, hence the Bo-ratio may fall below 1 but normally does not become negative. Negative values may happen in the surrounding area if the q_s-transfer towards the surface is combined with the q_l-transfer away from the surface to the atmosphere.

A major factor in the urban energy balance is the heat-flux density in the ground q_g. In view of the high thermal conductivity and heat capacity values of construction materials, the ground and the buildings store significant heat. In dry conditions, q_g accounts for up to 50 % of the urban heat balance. During the night, the energy reserve formed in this way fuels the heat island effect (Parlow 2003).

To summarize, the components D, E, A, q_a, q_g and q_s of the radiation and energy balance normally play a more important role in the city, whereas I and q_l are less important.

Urban Moisture Environment, Precipitation Amount, Fog Density

The urban atmospheric moisture environment depends on the water balance of the urban area:

$$\mathrm{p} + \mathrm{J} + \mathrm{F} = \mathrm{E} + \Delta\mathrm{r} + \Delta\mathrm{S} + \Delta\mathrm{A} \quad \text{(all units in mm time}^{-1}\text{)}$$

wherein:
p = precipitation
J = water supply from rivers and reservoirs
F = water released to air by combustion (cars, industry, domestic heating)
E = evapotranspiration
Δr = net runoff
ΔS = moisture storage
ΔA = net moisture advection

Precipitation patterns are odd in urban areas - most rain falls on the lee side of the city (see Fig. 2), due to the urban heat island effect, the urban emission of particles which become condensation nuclei and, the higher roughness length which causes rain not to fall vertically (Schütz 1996).

Relative humidity is affected by the pattern of rainfall. During the day it will be lower in the city than in the surroundings because of higher runoff, sealed surfaces and loss of vegetation. But

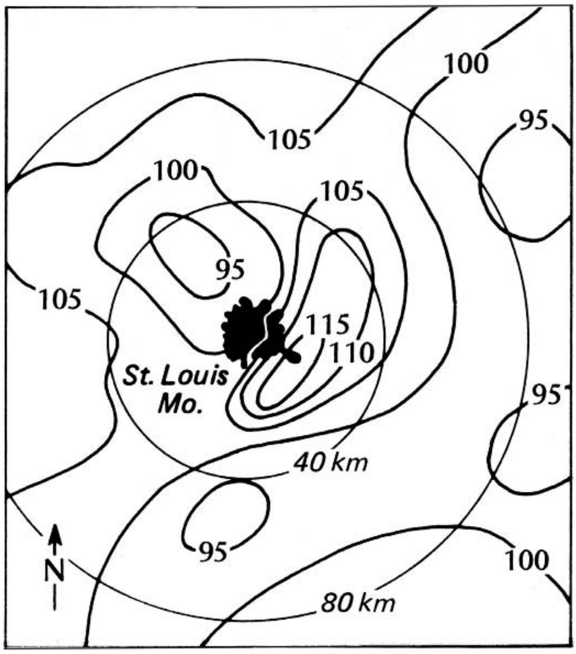

Fig. 2 Average rural/urban ratios of summer rainfall near St. Louis, 1949–1968 (after Changnon et al. 1971). The Rural/Urban ratio is the ratio of the precipitation recorded at any station to that at two stations near the urban center

at night the city can be more humid because of higher temperatures and less dew-fall than in the countryside (Fig. 3). In former days towns and conurbations in industrialized countries had more fog days than did their surroundings. But the number of foggy days is decreasing, due to cleaner air and a stronger urban heat island effect caused by a decrease in the percentage of natural surfaces within the cities.

Urban Heat Island Effect

Air and surface temperatures in cities are normally higher than in the surrounding areas. Urban excess heating is the result of the different importance of the various factors in the energy balance and air movement in cities and surrounding areas. The severity of excess heating, known as the Urban Heat Island Intensity (UHII), is normally expressed in terms of the horizontal temperature difference ($\Delta t_{u\text{-}r}$) between the city (u) and the surrounding area (r). Mechanisms producing the Urban Heat Island Effect include (after Oke 1979):

Urban Boundary Layer

- Anthropogenic heat from roofs and stacks
- Entrainment of heat from warmer canopy layer
- Entrainment of heat from overlying stable air by the process of penetrative convection
- Shortwave radiative flux convergence within polluted air

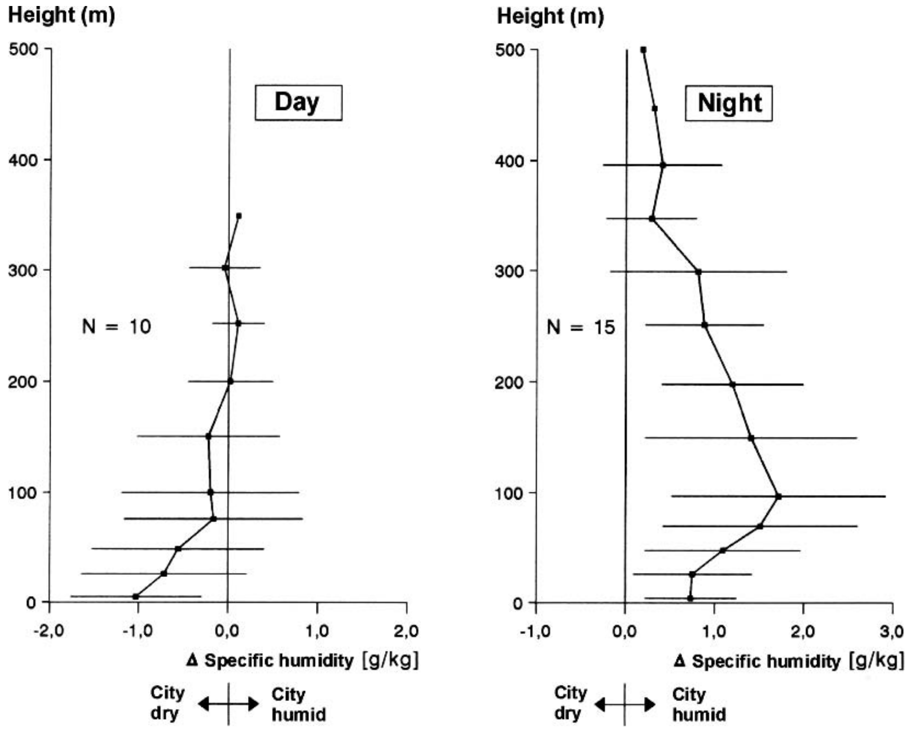

Fig. 3 Variations of mean nocturnal urban excess humidity with height over Christchurch during clear, calm weather (after Tapper 1990)

Urban Canopy Layer

- Anthropogenic heat from building sides
- Greater shortwave absorption due to canyon geometry
- Decreased net long-wave loss due to reduction of sky view factor by canyon geometry (sky-view factor is the ratio of the amount of the sky "seen" from a given point on a surface to that potentially available)
- Greater daytime heat storage (and nocturnal release) due to thermal properties of building materials
- Greater sensible heat flux due to decreased evaporation resulting from removal of vegetation and surface waterproofing
- Convergence of sensible heat due to reduction of wind speed in canopy

Several types of heat islands may occur at different times and cover different areas.

Surface heat islands are affected by the ground and caused by high surface temperatures. They mainly occur in built-up areas and therefore have clearly defined boundaries. They can be demonstrated on the basis of surface temperatures measured directly or by infrared photographs.

The *heat islands of the urban canopy layer* affect the atmosphere between the surface and mean roof height. *Urban boundary layer heat islands* form above the canopy layer as a result of heat transfer (q_s, q_l), artificial heat input (q_a), and increased absorption of radiation by atmospheric pollutants with resulting thermal re-emission. This type of heat island already extends so far upwards into the atmosphere above a city that it is propagated downwind by the overall wind patterns and gives the well-known "urban plume".

Factors Affecting Urban Heat Islands

Thermal conditions in conurbations are determined by land use, the structure of buildings and the size of the city. Especially severe urban excess heating occurs during calm and cloudless weather. During about 80% of the hours in a year, central European cities differ from their surroundings by $\overline{\Delta}t_{u\text{-}r} \geq 1\,K$. The UHII is affected especially strongly by wind speed, and less by cloud cover.

There are also relationships between the maximum intensity of heat islands and the size and functioning of a city. For assessing UHII, a city's population is often used as a surrogate for city size, because it is relatively easy to measure. The sealing of surfaces and the geometry of street canyons also affect UHI intensity. For example, the sky view factor (SVF) is the ratio of the actual visible area of sky to the potentially visible area. Low values (SVF ~0,1 – 0,4) indicate higher UHII (Blankenstein & Kuttler 2004).

UHII not only depends on these local factors but also on the city's geographic and regional location. For example, coastal cities have much less severe UHII than cities located in valleys and depressions or inland (Park 1987) and the more northern a city is located the higher the UHI.

In the case of human-bio-meteorological problems, city dwellers are exposed to greater thermal stress in the summer (due to higher air and radiation temperatures and reduced near-surface air exchange) than are the inhabitants of surrounding areas. This is confirmed by the higher incidence of heart and circulation problems in conurbations during hot spells (Kalkstein et al. 1996). But there are also positive effects of the urban heat island, e.g a reduction in the energy required for space heating.

It would be beyond the scope of this chapter to discuss the other changes caused by urban excess heating. These include an increase in the length of the growing season, a shift in vegetation growth-phases and larger numbers of immigrant or imported plant species, and animals from warmer areas. And of course, apart from a reduction in the number of frosty and icy days, a reduction in the number of snowy days reduces costs for snow and ice clearance.

Urban Air Pollution

Urban air quality is mainly determined by road traffic, industry, power and heating plants and households. The most important pollutants in urban atmospheres include NO, NO_2, CO, NMVOC (Non-Methane Volatile Organic Compounds), O_3, SO_2, dust and soot. The urban CO_2 (which is not pollution but is an important determinant of the atmosphere's infra-red behavior) concentrations are enhanced in sealed areas (Henniger & Kuttler 2004). Depending on the degree of industrialization, economic conditions and the geographic location of conurbations, different pollutants predominate, resulting in varying air quality problems. For example, NO_x, CO und O_3 tend to predominate in areas with high traffic density, whereas $SO_{2,}$ dust and soot are the main pollutants associated with coal and oil heating, power generation systems, and lower traffic densities. An analysis for central Europe shows that road traffic accounts for ~50% of NO_x, 60% of CO and > 30% of NMVOC emissions. Among other things, these pollutants are ozone precursors and may result in high urban ozone concentrations during sunny summer conditions. In some countries, the air quality of conurbations is also affected by emissions from biomass combustion or dust carried in by the wind from deserts and elsewhere.

Table 2 shows the general situation of air pollution in individual cities. The various individual species of pollutants include:

SO_2: Over the past few decades, SO_2 concentrations especially in Middle Europe and North America have generally been reduced some cities, by installation of filters and flue gas scrubbers on industrial plants and power stations, use of low-sulphur fuels, and changes in consumer behaviour (use of gas instead of coal and oil). However, relatively high concentrations still severely impair air quality in some cities, especially where coal or peat are common fuels ([e.g. Beijing (90 $\mu g\ m^{-3}$), Shanghai (79 $\mu g\ m^{-3}$), Manila (55 $\mu g\ m^{-3}$)].

NO_2: Worldwide, the main source of NO_2 pollution is internal-combustion vehicles. Areas where catalytic converters are not widely used or traffic densities are high often have severe NO_2 concentrations, e.g., Sofia, Athens, Los Angeles, London and São Paulo, where concentrations are sometimes $> 75\ \mu g\ m^{-3}$.

O_3: Ozone concentrations in the cities listed in Table 2 range from 10 $\mu g\ m^{-3}$ (Vancouver BC Canada) up to 69 $\mu g\ m^{-3}$ (Mexico City). In Los Angeles, where high values were still measured a few years ago, pollution control action has reduced the concentration of precursor substances and now limits ozone concentrations.

CO: In most conurbations, CO concentrations are 1 – 2 mg m^{-3}. Significantly higher values sometime occur, e.g. São Paulo (5.9 mg m^{-3}), Athens (5.1 mg m^{-3}) and Sydney (5.0 mg m^{-3}).

SST and PM 10: Particulate sources not only include industry, domestic heating systems and road traffic but also the deflation (aerial erosion) of soil material, which is a main reason for the high values recorded in several cities of Asia (e. g. Calcutta, New Delhi, Beijing).

Soot: Only limited data are available on soot concentrations. Soot comes mostly from diesel engines and the combustion of coal and oil. These are probably the main reasons for the high concentrations in Quito (120 $\mu g\ m^{-3}$), Athens (99 $\mu g\ m^{-3}$) and Mendoza (94 $\mu g\ m^{-3}$).

Most air quality problems are caused by SO_2, NO_2 and SST/PM 10. In western industrialised countries 'classical' air pollutants such as SO_2 are no longer a serious problem in most areas.

Urban Climate and Global Climate Change: the Future, as Best we can Guess it Today

Because the concentration of CO_2 in the atmosphere will probably double in the next 100 years, global average temperatures will likely increase ~2 K over those of 1985 (Houghton et al. 2001). This prediction is based on numeric simulations which lead to different results depending on the scenario selected. It is beyond the scope of this article to discuss the validity of the input data used or the uncertainty of the results. Here, this prediction is assumed to be correct: we also assume there will be no change in the main factors which directly or indirectly affect urban development. In other words, the current situation is assumed to be frozen: the only variable considered is the global temperature increase as it affects urban climate.

Table 2 Annual average concentrations of atmospheric trace substances for selected cities in different continents (all figures in μg m-3; except CO in mg m-3; SST = total suspended solids, PM10 = respireable particulate matter ≤ 10 μm; – = value not available; bold type indicates values in excess of WHO guidelines (1997) for SO2 = 50 μg m-3, NO2 = 40 μg m-3; a relates to the year of measurement period) (after Kuttler 2000 and Healthy Cities – Air Quality Management Information System – AMIS 2.0, 1998, WHO, Geneva (from Dr. Mücke, WHO, Berlin)

Continent/city	SO_2	NO_2	O_3	CO	SST	PM_{10}	Soot	Year
EUROPE								
Athens	44	**95**	25	5.1	–	–	99	95
Brussel	22	**50**	43	–	68	20	–	95
Edinburgh	29	**49**	29	0.7	–	–	9	95
Geneva	13	**58**	28	1.2	29	–	–	95
Helsinki	4	**42**	38	1.0	92	31	–	95
Copenhagen	11	**69**	24	1.7	78	–	35	94
London	26	**88**	20	2.1	–	28	–	95
Madrid 1	3	25*a*	–	–	–	–	85	96*a*/97
Sofia	39	**208**	–	–	308	–	–	96
Vienna	11	33	41	0.9	–	36	–	96
AFRICA								
Durban	30	–	–	–	–	–	–	95
Johannesburg	22	40	54	1.8*a*	106	–	61	94*a*/95
Cape Town	21*a*	**72**	41	–	–	27	–	94*a*/95
ASIA								
Bangkok	17*a*	32*a*	–	–	138	–	–	93*a*/95
Calcutta	31	35	–	–	542	378	–	96
Manila	**55**	–	–	–	164	–	–	93
New Delhi	24	**77**	–	–	459	246	–	96
Osaka	19	**62**	57	1.6	40	–	–	94
Beijing	**90**	–	–	–	343	–	–	94
Pusan	**66**	**55**	34	1.3	93	–	–	95
Seoul	35	**67**	24	1.5	–	78	–	96
Shanghai	**79**	–	–	–	289	–	–	94
Tokyo	18	**70**	31	0.9	–	48	–	95
NORTH AMERICA								
Los Angeles	3	**92**	39	2.3	–	43	–	95
Montreal	10	**41**	–	0.8	39	–	–	93
Vancouver	21	**55**	10	1.5	39	–	–	93
CENTRAL AMERICA								
Mexico City	**53**	**81**	69	3.2	180	61	–	96
SOUTH AMERICA								
Caracas	35*a*	**79**	–	–	63	–	–	94*a*/95
Mendoza	6	**76**	–	–	61	–	94	97
Quito	22	–	–	–	154	57	120	95
Santiago	23	**86**	12	2.8	91	–	–	95
Sao Paulo	34	**88***a*	65*a*	5.9	116	89	–	91*a*/95
AUSTRALIA/NEW ZEALAND								
Sydney	–	33	18	5.0	70	31	–	95
Auckland	7	20	–	–	33	25	9	96

Table 3 Climatological event days for the greater Berlin area with present and changed climatological conditions (according to Wagner 1994 and Hupfer 1996, with some changes)

Event	Present [Number]	Modelling, scenario A, for end of 21st century (ECHAM I, T21) [Number]	Change [Number]
Extremely hot days, $t_{max} > 39\,^{\circ}C$	0.01	0.04	+0.03
Hot days, $t_{max} > 30\,^{\circ}C$	5.4	11.7	+6.3
Summer days, $t_{max} > 25\,^{\circ}C$	27.2	41.8	+14.0
Frosty days, $t_{min} > 0\,^{\circ}C$	56.6	38.6	−18.0
Ice days, $t_{max} > 0\,^{\circ}C$	22.0	8.8	−13.2
Extremely cold days, $t_{max} > -10\,^{\circ}C$	0.7	0.11	−0.59

Thermal Environment and Near-Surface Exchange Conditions

The climate in a city is largely determined by thermal conditions and changes in the wind field. These factors affect not only the human bioclimate but also energy consumption, emissions of anthropogenic and biogenic hydrocarbons, and the formation of secondary pollutants. For Berlin, as an example, Table 3 indicates the thermal changes expected for days with different weather. Whereas the winter will become less severe (18 fewer frosty days and 13 fewer icy days), there will be 14 additional summer days and 6 additional hot days per year. Less severe winter weather will probably lead to a reduction in energy consumption for heating (this will be dealt with later). On the other hand, an increase in the number of summer days could lead to rising energy consumption for air conditioning. The temperature thresholds mentioned confirm a reduction in the recurrence cycle for hot days and the less frequent occurrence of frosty, icy and extremely cold days. Town and country planners should take action at an early stage to counteract the increased thermal discomfort occurring during the summer. For example, highly reflective colours should be used for facades, vegetation should be planted on the roofs of buildings, and adequate ventilation channels should be provided to allow of the flow of cooler air from the surrounding countryside as far as possible into the city centre.

As a result of the surface roughness caused by buildings, wind speeds in urban areas are normally lower than in the surrounding countryside. If near-surface air-exchange is restricted by low-wind conditions and extremely stable atmospheric stratification (temperature inversion), there may be an increase in atmospheric pollutant levels. The question therefore arises as to whether global warming leads to changes in ventilation conditions in cities. For Berlin, Gross (1996) predicted that thick temperature inversion layers ($>$ 300 m) would occur 20 % more frequently under the conditions mentioned than in 1985. On the other hand, less severe (thinner) temperature inversions will probably occur less frequently under the same conditions. This is because of a change in the characteristics of the air masses. Thicker inversions are more stable than thinner ones, therefore air pollution is likely to be worse during such episodes as they will probably last longer.

Power Consumption

Because power consumption is determined largely by climatic conditions, the location of a city is a major factor in its energy use. Essen, Germany (pop = 590,000; temperate zone) and Los Angeles, USA (3.5 M pop, subtropical zone) are good examples. For Essen, as expected, power consumption

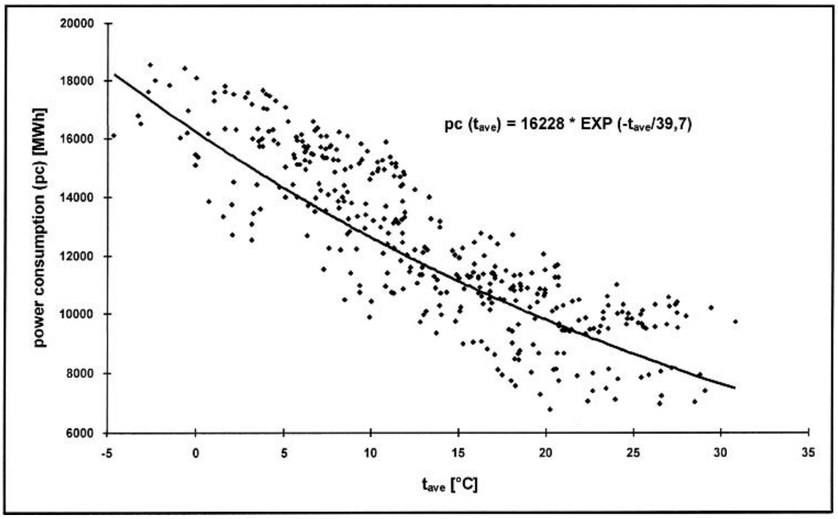

Fig. 4 Dependence of average daily power consumption (pc) on mean daily temperatures (t_{ave}) for the supply areas of Essen and adjacent cities; (after Kuttler 2001)

is inversely proportional to temperature (Fig. 4). At temperatures <0 °C, a temperature change of 1 K results in a change in power consumption of ∼400 MWh/K, or twice as much as the change in power consumption at summer temperatures around 25 °C (∼200 MWh/K) – people are more sensitive to cold than they are to excess warmth. Table 4 indicates the monthly percentage of annual total power consumption. If the annual average temperature rises by 2 K, there will be a fall of about 8 % in overall power consumption, entirely as a result of higher winter temperatures and the reduced demand for heating. There will be no difference in power consumption in the summer. Of course, this assumes that there will be no change in consumer behaviour (= business as usual) as a result of higher temperatures, i. e. there will no increase in use of air conditioning in the summer. The predicted reduction in power consumption should also lead to reduced emissions, especially of carbon dioxide, by power stations.

In contrast to the situation in the temperate zone, higher temperatures are predicted to cause increased power consumption in western subtropical cities. Winter power consumption is only of secondary importance in such areas, so the increase in power demand for air conditioning will be the major parameter. Taking weekday power consumption in Los Angeles as an example, Oke (1994) demonstrated that power consumption remains roughly constant at temperatures between 15°C and 20°C, but then rises by about 33% between 20°C and 25°C. Apart from intensifying the urban overheating effect, the higher power consumption results in more severe air pollution and increased use of all resources needed for power generation.

Table 4 Relative portions of monthly power consumption in annual power consumption of the city of Essen[1)] under present (1995, (a)) and changed (b) climatic conditions [1995: 4,523 GWh[2)] = 100%[3)]] (after Kuttler 2001)

Month	I	II	III	IV	V	VI	VII	VIII	IX	X	XI	XII	Year
(a)	11	9	10	8	7	7	6	6	7	8	10	11	100
(b)	9	8	9	8	7	7	6	6	7	7	8	10	92
(b)-(a)	−2	−1	−1	0	0	0	0	0	0	−1	−2	−1	−8

[1]) supply area of Essen and adjacent cities, 2) GWh = gigawatt-hours, 3) rounded.

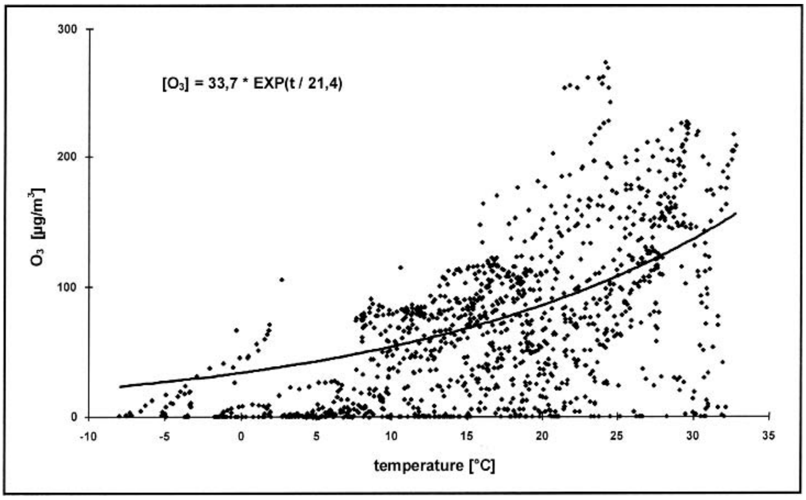

Fig. 5 Dependence of ozone concentrations [O_3] on air temperature [t] for clear days in a 70-hectare urban park in Essen (data basis: 1,231 30-minute-averages recorded from May 1995 to September 1997; after Kuttler 2001)

Air quality

Most chemical reactions are accelerated by rising temperatures, so global warming will affect the emission, transmission and concentration behaviour of atmospheric trace substances and therefore also urban air quality. For example, it is expected that there will be more emissions of certain anthropogenic and biogenic volatile organic compounds (AVOC and BVOC) which play a major role in the formation of tropospheric ozone. The most widespread urban AVOCs areas are BTX compounds (benzene, toluene and xylene) which are mainly produced by road vehicles. Apart from exhaust emissions, the collective benzene emissions caused by evaporation losses, refinery leakages, fuel production, refuelling, tank-breather and parking losses are also major factor in air pollution. In addition, motor industry projections indicate that both the number of vehicles and the distance travelled by each vehicle per year are likely to grow. As a result, further increases in pollutant output, especially in developing countries, must be expected despite the increase in use of catalytic converters. Biogenic volatile hydrocarbon emissions (mainly isoprene and monoterpenes from conifers) are positively correlated with air temperature and, in the case of isoprene, with solar radiation flux density. BVOC compounds are highly reactive and therefore represent considerable ozone formation potential even in low concentrations. In addition, it is assumed that the portion of BVOC in total global hydrocarbon emissions is of the order of 90% (Guenther et al. 1993). Although concentrations of biogenic hydrocarbons in urban areas are normally low, various studies made in urban parks and gardens confirm that they may play a considerable role in ozone formation (Benjamin et al. 1996). The expected climate changes will lead to an increase in the chemical reaction rate of ozone precursors. As a result, in combination with the rise in NO_x and carbon dioxide concentrations, ozone concentrations will reach higher levels than at present. Fig. 5 indicates the relationship between ozone concentration and air temperature in a large urban park in Essen (Kuttler & Strassburger 1999). There, ozone concentration increases by 5%/ K between 25 and 30 °C. However, air temperature is largely a function of insolation and therefore this correlation does not necessarily indicate causation.

Conclusions

The air quality in a city is chiefly determined by the thermal and air hygiene components of the bioclimatic complex.

As data from example cities show, it is likely that the number of days with high summer temperatures will increase in the temperate and subtropical zones, and that areas with thermal discomfort will become larger. On the other hand, the climatic situation in cool temperate cities may be expected to improve.

The concentration of ozone, the key component of summer smog, will increase and summer smog will spread to areas presently unaffected.

In terms of preventive or protective environmental actions, we must ask what might be done to alleviate the predicted developments. Wherever technically feasible, actions to improve air quality should be taken at the pollution sources. In addition, new buildings should be designed specifically for local climates (Barlag 1997).

Town and country planners must ensure systematic development of ventilation channels allowing fresh air from the surrounding areas to penetrate as far as possible into the city centre. In addition, it would help to increase the size of urban parks and gardens, and to increase total green area by planting both horizontal surfaces and suitable facades and roofs, using plants which have proved their resistance to urban environmental stress. More information is given by Wittig (1991). Taha (1996) says the positive effects of more vegetation in cities include:

- Reduced surface and air temperatures at ground level – This would not only alleviate thermal discomfort but also save energy as a result of shading and wind protection. Forestry authorities in the USA assume that cost reductions of the order of US \$ 4×10^9 per year through energy savings could be achieved if 100 million additional trees were correctly situated in North America. In addition, there would be positive effects on chemical reaction behaviour, in the form of reduced reaction rates.
- A reduction in biogenic hydrocarbon emissions caused by high temperatures and radiation also leads to lower ozone formation potential. However, when planting urban areas, it is important to select species with low isoprene and monoterpene emissions (no pine trees!). Otherwise, high vegetation density could be counter-productive, and might even accelerate ozone formation. In general, plants with isoprene emission rates of less than $2\,\mu g/(g \cdot h)$ and monoterpene emission rates of less than $1\,\mu g/(g \cdot h)$ are recommended. (A combined listing of 377 species found in the California South Coast Basin is ranked according to total [isoprene and monoterpenes] biogenic emission rate on hourly basis in Benjamin et al. 1996).
- An increase in exposed plant area also increases surface roughness and that in turn raises the rates at which particulate air pollutants settle out of the atmosphere (Kuttler 1991).
- The concentration of CO_2 falls because the gas is consumed by plants during photosynthesis.

Research into the potential effects of atmospheric warming on existing climatic and air hygiene conditions in conurbations is still at a very early stage. Systematic analysis is difficult because of the high complexity of urban ecosystems and the dependence of these ecosystems on the climate zone in which they are located. From the point of view of urban climatology, such work must be urgently expanded.

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Global Warming and the Urban Heat Island

Maria João Alcoforado and Henrique Andrade

Keywords: urban heat island · climate change

Introduction

The aim of this paper is not to present an exhaustive synthesis of the literature addressing global warming and the urban heat island; rather, it is to shed light on some common misperceptions and to address this issue by presenting a number of different perspectives and by highlighting some of the topics that require further research. By 2005, in the most developed countries of the world, ~74% of the population lived in urban areas. Between 2005 and 2010, the world's urban population is predicted to grow at an annual rate of 1.96%. By 2015, cities with over 10 M inhabitants ("megacities" – see also Kraas, this volume) will account for 9% of the total urban population (http://esa.un.org/unup/, [as of 10 Nov 2006]). However, urban growth incorporates two other trends: income growth and spatial growth or sprawl (Kahn 2006). The flight to the cities has been exacerbated not only by the effects of desertification, reductions in biodiversity and soil fertility in the rural areas, and the effects of globalization, especially in third-world nations (Handay et al. 1992, cited by Oke 1997), but also by income growth, especially in developed countries. As income grows, the urban population tends to sprawl (Kahn 2006).

According to Mills (2006), the new urban *utopia* is the sustainable city, whose impacts upon the environment are minimized without bringing about a reduction in the quality of life of the urban dwellers (Newman 1999; Kamp et al. 2003). Sustainable development is therefore the most important aim of the urban planning process (Barton 1996), because severe environmental problems, such as poor air and water quality, noise and thermal stresses, are especially likely in cities. Today's urban sprawl has led to worsening environmental problems, as more area is sealed over (i.e., becomes impermeable through paving or building construction), less green area remains, and water and energy consumption increase (Kahn 2006). As a result, ever more serious local, regional and even global threats have been emerging (Oke 1997; Decker et al. 2000).

Many titles in the scientific literature refer either to global warming (GW) or to urban climate, the latter including in particular the issue of urban heat islands (UHI). However, there are few papers or books that address the relationships between GW and UHI. Differences in temporal and spatial scales, research methods and data sets, as well as the lack of a comprehensive theoretical framework, are responsible. However, a better understanding of the role urban areas play in GW is an urgent concern, so that one might fight the negative consequences of GW and to take necessary measures to mitigate those consequences or to adapt to new climate conditions. On the other hand, the impact

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of GW upon urban thermal climates is also poorly understood and is often the object of many overly simplistic and erroneous ideas.

Global Warming

Climate change is defined by the Intergovernmental Panel on Climate Change (IPCC 2001) as any change in climate over time, whether due to natural variability or as a result of human activity. Ever since the formation of the Earth, there has always been considerable temperature variability over time, as is attested by both natural and documentary sources. The mean global temperature has increased by 0.6° (±0.2°C, 99% CL) during the 20th Century. Warming has been more significant over land than over the ocean, and most intense in the winter and in the Northern Hemisphere. In turn, the daily temperature range has been decreasing. Over the last millennium, colder and warmer periods have alternated, mostly due to natural forcing factors: for example, a "Medieval Warm Epoch" was followed by a "Little Ice Age" that lasted from around 1300 until the 1850s (Brázdil et al. 2005). From then onwards, the temperature has been rising, although not in a continuous and uniform manner. This temperature increase has been particularly intense during 1910–1945 (centered in the North Atlantic) and from 1976 onwards, particularly in the middle- and high-latitude areas of the Northern Hemisphere (except for the NW Atlantic areas) (Drogue et al. 2005).

According to many studies (e.g., IPCC 2001) the latter temperature rise has been exacerbated by human activity, namely through the emission of greenhouse gases. Moreover, the increase in mean temperature that occurred during the 1990s was the highest in the 20th Century. Global temperature anomalies (in relation to the 20th Century mean) in the early years of the 21st Century seem to confirm that warming is continuing – the single greatest anomaly took place in 2005 (+0.61°C; the warmest year of the 20th Century was 1998, at +0.58°C); [see at http://lwf.ncdc.noaa.gov/oa/climate/research/anomalies/anomalies.html]. According to IPCC (2001), a worldwide increase of 1.4 to 5.8°C (depending on the scenario) is expected to occur from 1990–2100. An update of the IPCC report was released in early 2007.

The Urban Thermal Effect and the Urban Heat Island

Tables summarising the climatic effects of urban areas have been presented elsewhere – e.g., Landsberg (1981), Oke (1988), Kuttler (2004) and in the present volume by Kuttler and Endlicher et al. Here, we focus on the effects of urban areas upon urban temperature, specifically upon the air temperature in the urban canopy layer.

Urban climate is a consequence of regional, local and urban geographical features (Lowry 1977). In most cases, due to the non-existence of pre-urban data, the "urban thermal effect" is calculated by comparing data from urban and nearby rural meteorological stations. The UHI is the best documented instance of inadvertent human climate modification (Oke 1987). It is exemplified by those urban areas where the surface, sub-surface or air temperatures are higher than the corresponding temperatures in surrounding rural areas. The UHI occurs most strongly during night time (see also Kuttler, this volume). Oke (1987) summarized the main modifications of the energy balance that lead to UHI formation. UHI intensity is defined as the difference between the highest air temperature recorded in the urban canopy and the lowest recorded in the surrounding rural areas. This apparently simple definition is actually not simple, and is applied in many different ways in different case-studies, due to differences in methods used in the collection of the data (such as relying on data from meteorological station[s] vs carrying out field measurements) and/or differences in the selected thermal parameter (minimum temperature, daily temperature range, maximum temperature, annual or monthly mean temperature), among other factors. These methodological differences are critically important and often render generalization difficult or impossible.

Consequences of Global Warming to the Urban Heat Island

Warmer cities have been frequently associated with global warming (the former being referred to as either a cause or a consequence of the latter). However, a large number of questions have arisen, and remain, with regard to the relationship between these two phenomena. **“**In many instances, the urban climate effects are similar to, and maybe even greater than, those changes predicted from global climate models (GCM)” (Grimmond 2006). Lombardo (pers. comm.) states that air temperature in São Paulo (Brazil) increased by ∼2°C during the last century, that is, by ∼3x the increase in mean temperature of the planet.

On the other hand, global warming may not necessarily bring about an increase in UHI intensity: urban-rural temperature differences may remain constant even in an overall warmer world (Oke 1997). UHI intensity could even decrease during global warming, due to the likely increase in vertical instability conditions and the subsequent dissipation of urban heat (Brázdil & Budíková 1999). Future UHI trends will also depend on the frequency of the synoptic conditions (e.g., anticyclones) and weather types (calm and cloudless weather) that favour high-intensity heat islands (Oke 1987; Morris & Simmonds 2000). Indeed, Beranová & Huth (2005) found that the long-term changes in Prague’s UHI are associated with variations in the synoptic circulation types. The intensity of the UHI has increased the most (from 1961 to 1990) under prevailing anticyclonic conditions in every season except for spring. That increase also depends on the prevailing airflow directions. The analysis of data stratified in accordance with the circulation conditions indicated that the intensification of the UHI is generally higher under anticyclonic conditions and a S + SW flow (+0.27°C/decade in the winter).

The London Climate Change Partnership report (LCCP 2002) predicts not only the intensity of the UHI in London (particularly in the warm season), but also the number of days during which the UHI is expected to be >4°C – both under the B2 and A2 IPCC scenarios and through the use of statistical downscaling techniques (Wilby & Wigley 1997; Patz et al. 2005). However, there are several limitations with regard to global and regional climatic modelling and the use of downscaling techniques. This is especially so whenever the downscaling concerns urban areas: additional problems arise due to the difficulty of (1) representing urban areas in regional models and (2) collecting detailed data concerning the urban features for every pixel. As argued by Lamptey et al. (2005), additional observational and modelling studies are still lacking. Furthermore, for warmer regions, if global warming causes increased air conditioning in cities, then more anthropogenic heat will be released and the UHI will increase further still (Auliciems 1997). Conversely, in colder areas there will be a reduction in energy use for heating, but “. . . early snowmelt will increase the thickness of the thawed layer in summer and threaten the structural stability of roads, buildings and pipelines” (Hinkel et al. 2003).

Moreover, the current changes in the magnitude of the urban thermal effect cannot be expressed solely in terms of UHI. In some cases, there is a decrease in UHI over time alongside an increase in the areas affected by urban warming, due to urban sprawl. In both Europe and North America, the greatest growth in both “core” urban area and population growth took place one century ago. “Much of the population shifts now underway involves movement away from concentrated urban areas and into extensive, sprawling metropolitan regions, or to small- and intermediate-size cities. (. . .) This means that the imprint of urban areas on climate is increasing” (Grimmond 2006,p. 223). Tereshchenko & Filonov (2001) predicted that UHI intensity in Guadalajara (Mexico) will decrease due to “megalopolization”, but the area affected will be larger. In New

Jersey (Rosenzweig et al. 2005), even if UHI intensity does not increase, “UHI like” conditions and temperature extremes will expand in dense suburban areas.

The Influence of Cities upon Global Warming

A Direct Impact upon Global Warming?

The increase in temperature of cities all over the world does not have a direct impact upon global warming. This is because urban areas cover $<1\%$ of the world’s land surface and urban effects upon air temperature “. . . only extend downwind for at most tens of km and dilution renders the impact on global climate negligible” (Oke 1997, p. 278). Furthermore, the energy released by human activities is only 10^{-4} that received on Earth from the Sun (Crutzen 2004).

Indirect Impact Through Increasing Emissions in Urban Areas?

Cities are, however, the most important source of greenhouse gases**.** Globally, 85% of the total anthropogenic CO_2, CFCs and tropospheric ozone are produced in or near urban areas (Oke 1997), as a result of “. . . higher living standards and higher demand for transportation, energy, water, and other resources and services” (Kahn 2006). The high density of pollutants in UHI plumes affects atmospheric chemistry and climate on a larger scale (Crutzen 2004). As the sensible heat fluxes increase, the ascending vertical atmospheric motion may intensify, leading to more unstable conditions (Lamptey et al. 2005; Makar et al. 2006) and to the carrying upward of water and pollutants into the middle and upper troposphere, with potential regional (and global?) consequences (Crutzen 2004; Sherwood 2002).

Cities are also a major source of airborne particulate material, which has a distinct climatic effect. IPCC (2001) includes computations of direct and indirect radiative forcing effects of different types of aerosols. Aerosols generally tend to slow the increasing temperature trend, with the exception of fuel organic carbon aerosols (there is also great uncertainty about mineral dust aerosols).

Urban Influences upon Global Temperature Trends?

Very different opinions can be found in the literature with respect to the urban influence upon the temperature trends. Some of those differences may reflect differences in methods and in data used. For example: (1) Several thermal parameters have been used (e.g., minimum and maximum temperature, daily temperature range, several period averages). (2) The approach varies according to the spatial scale of the research. In global or continental studies, the degree of spatial generalization is greater and therefore urban influence seems less important (Jones et al. 1999); [but note: some controversy remains (Peterson 2003) about the classification of meteorological stations as either “urban” or “rural”]. When seeking to identify the temperature trends of a city or a few cities, it is easier to collect metadata from each meteorological station in order to look for the non-climatic causes of the observed trends. Brázdil & Budíková (1999) and Beranová & Huth (2005) in Prague, and Quereda-Sala et al. (2000) in Spain are examples of research at a more detailed spatial scale. (3) Methods used to assess the urban influence may also differ. Three such methods are described below:

Method 1 – comparing series from “urban” and “rural” meteorological stations (or groups of stations). IPCC (2001) compared trends from rural stations (selected per the classification by the Global Historical Climatology Network - GHCN) with those from a larger group of stations (which included urban ones). Karl et al. (1988) and Peterson (2003) used data from several groups of US stations, the latter using 40 clusters of stations (including urban, sub-urban and rural ones), to show that the choice of different criteria to classify the cities leads to different results (see Fig. 1 in Peterson 2003). Shimoda (2005) compared rural and urban stations in Japan, whereas Brázdil & Budíková (1999) and Beranová & Huth (2005) compared Prague's urban stations with surrounding rural stations;

Method 2 – comparing the series produced by meteorological stations (which may reflect the influence of the urban effect) with the series resulting from the reanalysis carried out by the National Center for Environmental Prediction–National Center for Atmospheric Research (NCEP–NCAR) (Kalnay & Cai 2003; Zhou et al. 2004);

Method 3 – other authors used indirect criteria (Parker 2004; Parker et al. 2006). Because most studies (mostly carried out at middle-latitudes) show that the UHI increases under low wind speed and is greatest during night-time (Oke 1987), these authors compared trends of minimum temperature values from 264 urban meteorological stations under calm conditions with those under windy conditions.

Perhaps partly as a result of the differences in the methods used to assess the urban influence, in the scale of the analysis and in the thermal parameters used, these studies have yielded different conclusions, as described below.

"Large Scale Warming is not Urban"

The title of this paragraph is the same as that of a paper in Nature by Parker (2004). Parker concluded that "... urban warming has not introduced significant biases into estimates of global warming. The reality and magnitude of global-scale warming is supported by the near-equality of temperature trends on windy nights with trends based on all data".

However, uncertainties exist, for example:

(i) Parker does not state whether the relevant stations were urban or rural, nor how many of each there were;

(ii) according to Parker, "... the main impact of any urban warming is expected to be on minimum temperature"; however, the highest values of UHI typically occur a couple of hours after sunset (Oke 1987) – not at dawn, when the minimum temperature is most likely to occur. Besides, in some cities such as Shanghai (Chen et al. 2003) and Belgrade (Unkaševi¢ et al. 2005), there has been an increase in the maximum temperature that is probably due to the UHI;

(iii) as argued by Pielke & Matsui (2005), the UHI does not depend solely on the wind speed, but also on "... thermodynamic stability, aerodynamic roughness, and the vertical gradient of absolute humidity. (...) Parker's [2004] conclusions, therefore need further analysis and interpretation before they can be used to conclude whether or not there is an influence of urban warming on large-scale temperature trends";

(iv) the wind speed parameter used in that paper to select the nights in which the UHI was expected to occur was the daily mean wind speed, which is in fact an estimated (i.e., not actually measured) parameter based on the NCEP-NCAR reanalysis. Although these daily average reanalysis wind speeds are generally sufficiently correlated with the limited available observing-station wind speeds, it does not seem to make much sense to draw conclusions with regard to the UHI occurring at dawn based on the daily average wind speed;

(v) finally, there are some cities [(mostly but not exclusively in areas with irregular topography or near sea/lake shores), such as Nice (Carrega 1994), Eilat, Israel (Sofer & Potchter 2006), Lisbon Spain (Alcoforado & Andrade 2006; Alcoforado 1992, Andrade 2003), Fribourg Germany (Roten et al. 1984) and Calgary CA (Nkemdirim 1976)] where intensity of UHI does not increase linearly with reduction in wind speed from the prevailing direction.

Under very calm wind conditions, these cities are prone to thermally-induced breezes that frequently create urban cool islands or weaken the UHI.

In turn, the IPCC report (2001) concludes "... that estimates of long-term (1880 to 1998) global land-surface air temperature variations and trends are relatively little affected by whether the station distribution typically used by the four global analyses is used, or whether a special effort is made to concentrate on rural stations using elaborate criteria to identify them."

Peterson (2003) considers the urban influence to be insignificant in the USA. He compared 40 clusters of stations (191 urban, 85 rural, 18 suburban). After adjusting for biases in the data caused by the differences in altitude, latitude, time of observation, instrumentation and non-standard settings, he could not find any significant impact of urbanization across the *in situ* temperature observation network in the US. He argues that this is due to "micro and local-scale impacts dominating over the mesoscale UHI", combined with the fact that observations are less frequent in areas, such as industrial ones, that are usually warmer than rural areas. His work illustrates the need to acquire detailed knowledge of the sites and settings of the meteorological stations from which data are used.

Peterson (2003) presents Figure 6 of his paper as being a portrait of a "real" urban thermal field, but that figure in fact represents the spatial variation of surface temperature, rather than that of the air temperature – and surface temperature responds in a much more direct way to the physical properties of the substrate than does air temperature, the variation of which is much less dependent upon microclimatic factors. Besides, it seems to us that what this author calls "the ruralization of urban areas", as opposed to the "urbanization of rural areas" is a reflection of urban sprawl - a process that causes the UHI to be an outdated concept in the case of certain cities.

Based on the series produced by the National Observatory of Athens, Founda et al. (2004) found that cities are not influencing global temperature trends. They note that annual mean air temperature increased by 0.047°C/decade over the last century, most significantly during the warm season, and that the increase in the daily maximum temperature was greater than that in the daily minimum temperature. They also claim that, during the 1990s, "... a sudden temperature rise was observed during both the warm and the cold seasons ..." which attenuated the differences between the maximum and minimum temperatures and between the warm and cold seasons (p. 36). Yet, the UHI in the historical center (where the meteorological station is located) has not evolved uniformly over time, and, in particular, the urbanization rate (building density, population growth) has not undergone any recent increases, which is why these authors have concluded that this last temperature rise was not due to an increase in the urban effect. However, it is important to bear in mind that other factors may cause the UHI to increase, such as increased release of anthropogenic heat associated with rises in energy consumption.

"Large Scale Warming is Caused by Cities"

In some studies, global warming is almost ignored as the effect of the cities "encroaching upon observatories" (Quereda-Sala et al. 2000) becomes regarded as increasingly important. Two studies from Mediterranean Spain serve as examples. In their study of ten Spanish 1st order Mediterranean observatories (mostly located in cities), Quereda-Sala et al. (2000) state that "... the assumption of temperature stability in the Spanish Mediterranean should not be rejected, once systematic changes in air temperatures only occurred in highly urbanized areas". In turn, García-Barrón et al. (2004) have identified a positive trend in the minimum temperature series for Badajoz over 1864–1984. In 1984 the observatory was relocated from the historical city center to the University Campus. From 1984 onwards, the minimum temperature exhibited a negative trend until 1999, whereas the maximum temperature has exhibited a positive trend ever since.

Large Scale Warming is Also Urban

Other papers highlight the impact of the urban effect upon temperature trends while taking into account the changes over time in the conditions surrounding the climatological stations (Oke 1997), or show that the UHI intensifies as cities grow – e.g., Madrid (Yagüe et al. 1991), Seoul (Kim & Baik, 2002) and Tucson (Comrie 2000, in Beranová & Huth 2005).

Indeed, we need to somehow check whether the city under study has influenced the temperature change and to isolate that urban effect from the global temperature rise, that is, to extract the urban bias from the long-run temperature series. The papers on this subject show quite different and even contradictory results. We find that those differences are partly due to differences in the methods adopted to extract the urban bias, and partly due to the data used (number and type of stations, temperature parameter, period).

One method used to extract the urban bias, particularly when working with a large set of series, includes taking population growth into account. It is used mostly because it is easily available. According to Jones et al. (1989) "Although population growth is an indirect measure of urbanization and the urban warming effect, it is the only statistic that is readily available for most regions of the world". However, the usefulness of the demographic information is highly dependent on its reliability and "[T]he areal basis of reporting population statistics is often at an inappropriate scale" (Oke 1997). An illustration of this point comes from Portugal: the meteorological stations *Sintra/Granja* (located on a military base far from the nearest densely urbanized area) and *Lisboa/Geofísico* (in the Lisbon city center) are both presented in the NCDC database as being associated with a population of 1.1 M (i.e., the population of the Lisbon metropolitan area). How can this type of reliability problem be controlled for at the global scale? The fact is that errors such as these can lead to wrong conclusions with respect to the impact of the urban effect upon the air temperature. Moreover, the use of population size as a criterion for estimating the urban effect is also subject to controversy; the magnitude of the urban effect may be more closely associated with the level of energy consumption (e.g., cal/person/year), which depends on other socio-economic factors (Brázdil & Budíková 1999, re. Prague; Chen et al. 2003, re. Shanghai). Remember, the population of numerous cities in "developed" countries has recently tended to stagnate or even decrease. For example, the population of the seven largest French cities grew by 82% from 1955 – 1990, but is expected to increase by only 9% over 1990–2015 (http://esa.un.org/unup/). Nevertheless, this does not necessarily imply a reduction in the thermal effect. Other factors, such as "the urban area land use modifications" (Changnon 1999), and the characteristics of the urban structure and the area of the city (Oke 1973, 1987; Changnon 1999) may influence the urban effect upon temperature, regardless of whether or not there is a population increase. Changnon (1999) has used deep soil temperature data, which he claims to provide "an unbiased measure of the natural temperature trend" and has compared its observed trend with that of the adjusted air temperature series (which used the population size to extract the urban bias) presented by Karl et al. (1988). He concluded that the urban influence is much larger than estimated by Karl et al., making it clear that population is not the best parameter to use to extract the urban bias from the temperature series.

A second method consists of calculating the differences between the trends recorded in the urban and rural stations of the same area. This may yield interesting results, provided that some questions are explicitly considered. The homogeneity of the series must be checked, in order to make sure that the data are comparable. The classification of the stations as "urban" or "rural" must be very carefully made (Peterson 2003). Finally, the number of stations used, the selected temperature parameter and the choice of the periods for analysis will also influence the results. For example, Brázdil & Budíková (1999) and Beranová & Huth (2005) both studied Prague using data from the Prague/Klementinum meteorological station. However, the studies gave quite different conclusions, due to differences in the periods for which the trends were calculated and to the use of different "rural" stations. This example clearly shows that great care must be taken in interpreting the "figures" of the urban influence upon the "global" temperature variation over time. Furthermore, the use of mean values by most authors conceals the differences between the day/night trends and between trends under different synoptic conditions.

Kalnay & Cai (2003) compared the trends of actually observations from the US with data from the NCEP-NCAR reanalysis. They concluded that changes in land use patterns (mainly due to urbanization) have had a very significant influence upon air temperature trends (0.027°C/decade). Those conclusions have been disputed, because the series had not been homogenized (e.g., Parker, 2004, 2006; Peterson, 2003). Zhou et al. (2004) used an improved version of the Kalnay & Cai (2003) method (including data homogenization) to estimate the impact of the urban effect in China: ". . . The average differences in maximum and minimum temperature trends between observed and R-2 [Reanalysis 2] data are −0.016 and 0.116°C per decade, respectively".

Assessing the Impacts of Global Warming in Urban Areas

IPCC (2007) includes more recent figures and reviews forecasts with respect to global warming, as well as some downscaling references to a number of different regional areas - downscaling being particularly troublesome in urban areas. Moreover, in the elaboration of forecasts, ". . . there exists a hierarchy of causal relationships with uncertainties at each level of the hierarchy (Carter 2001), such as those associated with the assumptions concerning (i) the main socio-economic and technological drivers of environmental change; (ii) the future emissions of greenhouse gases and aerosols into the atmosphere; (iii) the future composition of the atmosphere; and (iv) the future radiation balance of the Earth, . . . each [of which] is subject to additional uncertainties attributable to imperfect data and models as well as the inherent indeterminacy of predictions" (Carter 2001, p. 48). As regards estimating future impacts of climate changes, Carter (2001) argues further that there should be ". . . more emphasis on the presentation on uncertainties [at each stage of impact assessment] – both for stakeholders and for fellow scientists" (p. 49).

Regardless of the high level of uncertainty surrounding climate change forecasts, particularly at regional and local levels, it is likely that the impacts of climate change upon human life and activities will be considerable. Although most of the foreseen consequences are not exclusive to urban areas, they will be felt most intensely in cities (Patz et al. 2005; Lindley et al. 2006), due to (1) the socioeconomic and demographic characteristics of urban areas, (2) the concentration of infrastructures and (3) the cumulative effect of global warming and urban warming that can be expected . "In central London the UHI effect could add a further 5 to 6°C to temperatures during summer nights" (LCCP 2002).

The main consequences of the temperature rise in urban areas consist of its impacts upon the human health, the ecosystems, and consumption of water and energy. Some of the main direct impacts of global warming include an increase in mortality and morbidity due to the increase in the frequency and intensity of heat waves (even allowing for enhanced acclimatization) and, on the other hand, a decrease in winter mortality, associated with warmer winters. Some authors (e.g., Kalkstein & Greene 1997), have studied several large American cities and argue that the mortality increase in the summer will be greater than its decrease in the winter. Others consider that in the case of medium- and high-latitude cities, the decrease in winter mortality due to GW will be greater than the increase in summer mortality due to the heat waves (Martens 1998, based on the analysis of several cities located in different climatic zones; Keatinge et al. 2000, for Europe; Donaldson et al. 2003, for London). Others (Dessai 2002) only mention the summer increase. In tropical cities (e.g., Singapore, Caracas), an increase in mortality is expected to occur due to GW (Martens 1997). Other indirect effects of GW include increased production of aero-allergens associated with higher temperatures (Epstein & Rogers 2004; Wayne et al. 2002; McMichael et al. 2006); increases in air pollution (particularly ozone and other photochemical pollutants: Cardelino & Chameides 1990; Stone 2005); increased vector-borne infections (Epstein & Rogers 2004; Epstein 2004; Wayne et al. 2002; McMichael et al. 2006); sea-level rise; and increases in the frequency and magnitude of extreme meteorological events (besides heat waves), including hurricanes and flash-floods, which can have particularly bad consequences in urban areas (LCCP 2002, 2006; Smith 1999; Suarez et al. 2005; Bloomfield et al. 1999). Moreover, synergies occur in urban areas between the various different hostile factors: for example, combining high temperatures, more air pollution and more aeroallergens (Epstein 2004), can leave city inhabitants in a particularly vulnerable position (Kalkstein 1997; Lindley et al. 2006).

Urban ecosystems, such as cities' green areas, will also be affected by GW: species composition may change, exotic species may be favoured (Dukes & Mooney 1999; Wilby & Perry 2006) and the phenological rhythms of plant species may be modified (e.g., Seoul, Ho et al. 2006). Certain urban ecosystems, such as the intertidal ecosystems of the Greater London area (Wilby & Perry 2006), will also be affected by sea-level rise.

Furthermore, urban temperature rises will lead to increased energy consumption for refrigeration and air conditioning in summer (Shimoda 2005; Solecki et al. 2005) and decreased energy for heating in winter. The overall balance will depend on local conditions (LCCP 2002, 2006).

Discussion and Conclusions

Throughout this text, attention has been drawn to the fact that there is a wide gap between the various types of studies that address the issue of temperature change at different spatial scales – particularly between those that address the issue of global warming and those focusing on urban warming resulting from the urban heat island.

The large number of papers addressing the issue of global warming have used a variety of data sets, computational methods and climate models to estimate temperature trends. Despite these differences, the vast majority concur: since the temperature trends of the last century cannot be explained by internal variability of the climatic system (IPCC 2001), part of the observed global warming is human-induced and largely due to the increase in the emission of greenhouse gases. Even if the level of anthropogenic influence were to be radically reduced, there would still be a climatic response to the past emissions that would project itself far into the future (Lindley et al. 2006).

In the time since Howard's works on London's climate, the methods and techniques of urban climatology have improved greatly (see Kuttler, this volume), and many papers now address this subject. Urban climatology has emerged as a new scientific field, with both theoretical and applied objectives (Oke 1987, 2006; Grimmond 2006; Mills 2006). Even though several other climatological parameters are modified in and by the cities, attention in this present writing has been restricted to temperature change. Due to increases in soil sealing, decreases in evapotranspiration, the evolving characteristics of the urban fabric, and increasing anthropogenic heat flux, among other reasons (Kuttler, this volume), urban surface and air temperatures have been on the rise. This fact is usually quantified by using UHI intensity.

However, using UHI intensity as the sole relevant parameter in assessing the magnitude of the urban thermal influence raises several questions. First of all, UHI can be computed in very different ways, as discussed above. Moreover, as urban areas continue to sprawl, UHI may not be the best parameter for showing the increasing influence of urban areas upon local, regional (and global?) climate. This is because "urban" stations are often located in old city districts, far from the present growing urbanization processes; and also, many "rural" stations are becoming increasingly integrated into sub-urban, or even fully urban, settings. In those cases, the increase in those areas' anthropogenic influence upon the temperature trend may not be visible in variations of UHI intensity over time. The methods to be used in assessing the evolution of the urban thermal influence over time should thus be the object of serious discussion by the scientific community.

Over the last few decades, an association has been drawn between warmer cities and global warming. It is true that the urban temperature rise now observed in several cities is similar to that predicted by global climate models to occur on a global scale in the next 50 or 100 years. However, a great deal of uncertainty remains with regard to the effects of global warming upon urban temperatures, on the one hand, and the influence of urban areas upon global warming, on the other. A survey of many studies indicates that the conclusions are indeed varied and that a number of problems remain to be addressed with regard to assessing and forecasting both the urban "reaction" to GW and the influence of the cities upon GW. Importantly, the results of various studies cannot be easily compared, due to the wide variety of data sets and analysis procedures used. Further discussion of these aspects by the scientific community therefore seems warranted.

Our goal was not to present a synthesis of all the literature addressing this subject; rather, it was to survey a number of relatively recent papers and thereby not only extract some of the main conclusions, but also highlight some of the problems and questions that, up until now, are yet to be given a clear and unanimous answer and which should be the object of further reflection and discussion.

Main Conclusions

In the works studied and cited for this present work, all authors agree that the energy balance in urban areas is modified due to human construction and activities, and that, as a result, the subsurface, surface, and air temperatures are higher than in surrounding areas.

Most authors agree with the following statements regarding the influence of urban warming upon regional (and global?) warming:

a) Urban anthropogenic heat fluxes do not have a direct impact upon global warming, because urban areas cover $<1\%$ of Earth's land areas and the amount of energy released by man is much less significant than the energy received by Earth from the Sun.
b) Cities are a very important source of anthropogenic greenhouse gases and thereby contribute indirectly to global warming.
c) Warming of urban atmospheres exerts a slight influence upon the computation of global warming (except in the case of those authors that use large data sets and work at the planetary or hemispheric scale).

   Most authors agree with the following statements about the influence of global warming upon urban warming:
d) The impacts of global warming (including its impacts upon human well-being and health, various ecosystems, and on levels of energy and water consumption) may be exacerbated in urban areas.
e) Depending both on their latitude and on their regional climate, cities will either be losers or winners from global warming (Oke 1997). From the point of view of the human bioclimate, the high-latitude cities will probably be the winners, and low- and mid-latitude cities, especially in the summer, will probably be losers. Warmer cities are, in general, likely to experience an increase in the levels of photo-oxidant air pollution and water consumption, so from that point of view all the cities will probably lose. As regards energy consumption, the only winners will be cities in colder climatic zones and, in the winter, those at intermediate latitudes. However, additional climate-related problems may arise in high-latitude cities as a consequence of GW.
f) The consequences of GW will exhibit considerable regional variability and will depend on the future frequencies of the various synoptic situations and weather types. For example, an increase in vertical instability associated with higher temperatures can partially offset urban warming.

Some Issues that Remain to be Addressed

The first issue concerns choosing the best methods for extracting urban bias from temperature series, bearing in mind that, during the 20th Century, urban areas have been engulfing a large number of the historic 1st order meteorological stations. One method consists of statistical modelling, seeking to correlate temperature increase with population size. This method is subject to "... some debate, since changes can occur independently of population and there are limitations to population statistics" (Voogt 2004). A second method compares series from rural and urban stations located in the same general area (e.g., Brázdil & Budíková 1999, and Beranová & Huth 2005). Although this method seems to yield the best local results, its application is very difficult at the global scale. Ultimately, the best solution may involve construction of large data sets, including appropriate metadata so that one can correct the series, and the use of standard computation methods.

Yet another problem is assessing the degree of uncertainty involved in discussing and forecasting the consequences of urban/global warming. Uncertainty is present whenever each causal relationship involved in global changes is computed and projected, as well as whenever those projections are used as inputs to assess impacts (from the global emission scenarios and the projections regarding radiative forcing to the climate projections and the analysis of regional scenarios and impacts) – and, of course, feedback mechanisms occur at and between all of these levels (Carter 2001).

To sum up, it is very important that the research in this area continues to be improved, in order for it to be possible to accurately assess and forecast urban temperature trends, and to produce better estimates and forecasts of global warming that take into account the urban data sets.

Although we cannot at this stage accurately quantify urban influences upon global temperature trends, measures must be taken to mitigate the negative consequences of both urban warming (including upon human well-being and health, and upon urban economies) and global warming, which will be felt with greatest intensity in the urban areas (the impacts of sea-level rises upon low-lying cities is an example). On the other hand, measures aimed at capturing the benefits of climate change should also be taken (Lindley et al. 2006). Special attention should be devoted to the situation of the low-latitude cities, most of which are located in developing countries where the urban population is growing at very high rates. Indeed, those are the cities that, from any perspective, will surely be the losers from urban and global warming. We can be sure that local measures taken there will have an impact upon the political measures aimed at decreasing global warming.

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A Retrospective Assessment of Mortality from the London Smog Episode of 1952: The Role of Influenza and Pollution

Micheile L. Bell, Devra L. Davis, and Tony Fletcher

Abstract The London smog of 1952 is one of history's most important air pollution episodes in terms of its impact on science, public perception of air pollution, and government regulation. The association between health and air pollution during the episode was evident as a strong rise in air pollution levels was immediately followed by sharp increases in mortality and morbidity. However, mortality in the months after the smog was also elevated above normal levels. An initial government report proposed the hypothesis that influenza was responsible for high mortality during these months. Estimates of the number of influenza deaths were generated using multiple methods, indicating that only a fraction of the deaths in the months after the smog could be attributable to influenza. Sensitivity analysis reveals that only an extremely severe influenza epidemic could account for the majority of the excess deaths for this time period. Such an epidemic would be on the order of twice the case-fatality rate and quadruple the incidence observed in a general medical practice during the winter of 1953. These results underscore the need for diligence regarding extremely high air pollution that still exists in many parts of the world.

Keywords: air pollution · influenza · London · mortality

In December 1952, a thick smog settled over London, resulting in unprecedented morbidity and mortality, bringing the relationship between air pollution and health to the attention of the general public, the government, the media, and the scientific community. The Big Smoke, a 50-year commemoration of the smog, took place in London in December 2002. The conference reviewed the events leading to the episode, the health impacts of the smog, and current air pollution conditions in London and elsewhere. This revisiting of the smog reminded us that much can still be learned from the event, in terms of both how air pollution affected Londoners at the time and how health can be affected by high levels of air pollution in much of the world today.

In an earlier study (Bell and Davis 2001), we analyzed the relationship between pollution and mortality and morbidity for the London smog of 1952. Several indicators of morbidity, such as hospital admissions, showed patterns similar to those of pollution levels. Daily mortality during the smog was also associated with daily air pollution levels. Results were not sensitive to the peak day of the pollution and were not confounded by temperature. The relationship between mortality and air pollution levels for longer periods revealed a statistically significant association between weekly ambient levels and mortality. Regression analysis showed that weekly air pollution and mortality were statistically associated, even when the week of the episode was removed from the data set,

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leaving only air pollution concentrations far below the episode (Bell and Davis 2001). Thus, air pollution was affecting mortality in London, aside from the extreme episode.

Mortality did not return to normal levels for several months after the episode. The strong immediate health response to the episode is evident in the coinciding sharp increase in mortality; however, the elevated mortality in the months after the smog requires a more detailed analysis. An initial government report (U.K. Ministry of Health 1954) proposed the hypothesis that influenza was responsible for the elevated mortality in the months that followed the episode. The weekly number of excess deaths in Greater London (the number of deaths exceeding those during the same time period the previous year) peaked at about 4,500 for the week ending 13 December 1952. Total mortality rates were about 80% higher than the previous year for December 1952 and were 50 and 40% higher, respectively, for January and February 1953. From December 1952 through March 1953, there were over 13,500 more deaths than normal. A fraction of these likely resulted from air pollution and a fraction from influenza.

The exact numbers of influenza-related and air pollution–related deaths are unknown and continue to generate debate (Stone 2002). Estimates of influenza deaths generated through multiple approaches contradict the influenza theory, indicating that far more people died from air pollution than originally believed. Estimates of the number of influenza deaths were constructed using observations from general practice of medicine in the London area and the control group of a vaccine study conducted at the time. The reconstructed estimates of influenza mortality reveal that only a portion of the excess deaths after the smog can be attributed to influenza.

Here we expand on previous analysis (Bell and Davis 2001) by providing additional estimates of the number of influenza-related deaths and by exploring how extensive an influenza epidemic would have to be to account for the excess mortality. The following describes a sensitivity analysis that explores how high influenza incidence and case-fatality rates would have to be to account for the increased mortality, indicating that such an explanation is highly unlikely.

Estimates of Influenza-Related Mortality

The number of deaths from influenza was estimated using two approaches: observations from a general medical practice and data from a vaccine trial. Observations from a general medical practice of about 6,000 persons in Greater London from 1949 through 1968 provided an estimated case-fatality rate of 0.2% (Fry 1969). For the years that had an influenza epidemic, Fry estimated that the percentage of patients who contracted influenza ranged from 3% to 17% and averaged 8% with a median of 6.5%. For the 1953 year, 6% of the patients had influenza. This supports other documents that reported a mild influenza epidemic during that time (U.K. Ministry of Health 1953, 1956).

The Medical Research Council Committee on Clinical Trials of Influenza Vaccine organized a clinical trial of an influenza vaccine with 12,710 volunteers in London and other cities (Committee on Clinical Trials of Influenza Vaccine 1953). The majority of volunteers were inoculated between 26 November and 5 December 1952, with some inoculations on 12 December 1952. Follow-up continued through 31 March 1953. The attack rate in the control group was 4.9% during the winter of 1952–1953. The original researchers divided areas into three regions based on their mortality rates because high mortality areas were assumed to have higher incidence of influenza. London was in the highest incidence category, with an attack rate of 6.61% in the control group of 1,181 volunteers. This value is similar to the incidence rate observed for the winter of 1953 for the general medical practice.

The original government report on the health effects of the episode (U.K. Ministry of Health 1954) proposed that an influenza epidemic caused the elevated mortality, although those authors recognized that some deaths in the months after the smog could be related to the air pollution from the episode. Excess deaths from this time through the end of March number more than 8,000. The report provides estimates of the number of deaths attributable to the smog using several different approaches, which do not include any deaths after 20 December 1952. The U.K. Ministry of Health estimated that 5,655 people died from influenza in the first 3 months of 1953 (U.K. Ministry of Health 1956). Another source reported 5,647 influenza deaths for the first 3 months of 1953 in England and Wales (World Health Organization 1953).

To better understand the role of influenza and pollution in the deaths that took place in the months after the extreme episode, an estimate of the number of influenza-related deaths was constructed using information from the observations of a general medical practice and the vaccine trial. The population of Greater London at this time was approximately 8.6 million, based on the 1952 census (U.K. General Register Office 1951). Because early government reports attributed no deaths after 20 December 1952 to pollution and because mortality rates remained elevated for several months, especially for January and February 1953, we calculated the number of excess deaths from 21 December 1952 through 28 February 1953. During this period, 8,275 more deaths occurred than expected based on the previous year, which did not have an influenza epidemic or extreme air pollution events.

Fig. 1 shows the fraction of deaths from 21 December 1952 through February 1953 that are related to influenza using four approaches; the original government estimate for January through March 1953 (Committee on Clinical Trials of Influenza Vaccine 1953, Fry 1969, U.K. Ministry of Health 1953) is provided for comparison. The reconstructed estimates are conservative in several ways.

All yearly influenza deaths from the observations from a general medical practice, attack rates from December through March 1953 from the vaccine trial, and the government estimate for January through March 1953 were applied to the shorter period of less than 2.5 months (Committee on Clinical Trials of Influenza Vaccine 1953, Fry 1969, U.K. Ministry of Health 1953). The government estimate was based on all deaths in England, yet in Figure 1 is compared to estimates only for Greater London. The reconstructed estimates indicate that about 1,000–1,400 influenza-related deaths took place during this period. The government estimate also leaves more than 2,600 deaths unexplained. Under each of these approaches, a significant portion of the excess mortality is unexplained by the influenza theory.

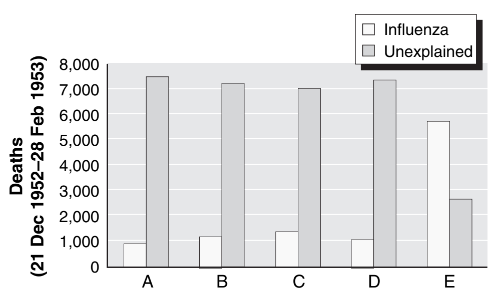

Fig. 1 Estimated number of influenza deaths in Greater London, 21 December 1952 through 28 February 1953. Methods of estimation are as follows: *A*, control group from influenza vaccine study; *B*, control group from influenza vaccine study, for areas of high mortality; *C*, observations from a general medical practice, using the average incidence rate of years with influenza epidemics; *D*, observations from a general medical practice, using the incidence rate for the 1953 winter; *E*, government estimate for January to through March 1953. Data from the Committee on Clinical Trials of Influenza Vaccine (1953), Fry (1969), and the U.K. Ministry of Health (1953)

Sensitivity Analysis

We performed a simple sensitivity analysis to explore how different values for the incidence or case-fatality rate could influence influenza mortality estimates and the fraction of unexplained deaths. This is important given that the exact number of influenza deaths is unknown and the approaches used here could conceivably underestimate or overestimate the actual number of deaths attributable to influenza. For example, the incidence rates generated through observations in a general medical practice are likely to be lower than the true incidence because some patients with influenza may not have consulted their physician. It is plausible that the underestimation is large if a significant number of patients with influenza did not consult their primary physician and he did not learn of their condition. In contrast, the case-fatality rate could be an overestimation, assuming the most ill patients sought medical care and few of the patients with influenza that did not receive medical care died. The attack rates from the vaccine control study (Committee on Clinical Trials of Influenza Vaccine 1953) could be lower than those in the general population because of the selection process of volunteers for the study.

Because of these possibilities, we performed a sensitivity analysis by increasing both the incidence and case-fatality rates. Fig. 2 provides the number of influenza deaths from 21 December 1952 through 28 February 1953 if the estimate generated using the 1953 winter rate from the general medical practice is too low by varying degrees. Results show that the estimated case-fatality and incidence rates for influenza would have to be drastically understated for influenza to account for the excess mortality after the smog. The most severe epidemic observed during the 20-year period of the general medical practice observations took place in 1957. If influenza after the pollution episode were twice that severe, many deaths still remain unexplained. If both the 1953 incidence rate and the case-fatality rate were underestimated by 100%, the corrected value of influenza deaths would be approximately 4,000. Only an extremely severe influenza epidemic could account for the majority of the excess deaths for this period. Such an epidemic would be along the order of twice the normal case-fatality rate and quad-ruple the incidence observed in 1953. These results reveal that even if the estimates of the number of influenza-related deaths, as presented in Figure 1, are substantially lower than the true values, a larger number of deaths remain unexplained by influenza.

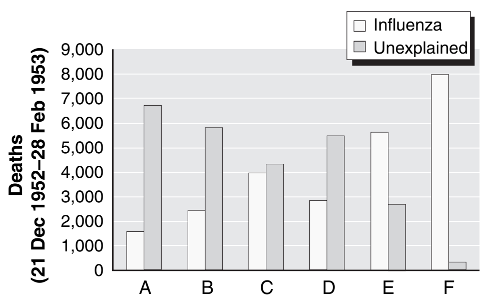

Fig. 2 Sensitivity analysis for number of influenza deaths in Greater London, 21 December 1952 through 28 February 1953. Methods of sensitivity analysis estimates are as follows: *A*, case-fatality rate and incidence from the observations of a general medical practice for the 1953 winter both increased by 25%; *B*, case-fatality rate and incidence from the observations of a general medical practice for the 1953 winter both increased by 50%; *C*, case-fatality rate and incidence from the observations of a general medical practice for the 1953 winter both increased by 100%; *D*, incidence of the most severe influenza epidemic observed for the general medical practice from 1949 through 1968 (17% for 1957 epidemic); *E*, incidence twice that of the 1957 epidemic (34%); *F*, case-fatality doubled and incidence four times higher than from the observations of a general medical practice for the 1953 winter. Data from Fry (1969)

Discussion

Our analysis shows that only a fraction of the elevated mortality in the months after the 1952 London smog can be attributed to influenza, leaving thousands of deaths otherwise unexplained. Sensitivity analysis illustrates that an influenza epidemic of enormous proportions would be required to explain the excess mortality. Such an epidemic would have to be about three times larger than the most severe epidemic recorded from 1949 to 1968.

These excess deaths could be attributable to air pollution through a delayed effect from the smog itself. They could also be related to air pollution concentrations that remained above normal levels through January 1953. Mortality in London was affected by air pollution during these years aside from the extreme episode. Regression analysis of weekly pollution and mortality levels from October 1952 through March 1953 and the corresponding weeks of the previous year (October 1951 through March 1952), controlling for temperature, found a statistically significant association, even when the week of the episode was omitted (Bell and Davis 2001). If the excess deaths in the months after the 1952 London smog are related to air pollution, the mortality count would be approximately 12,000 rather than the 3,000–4,000 generally reported for the episode. Additionally, there could be interaction between influenza and air pollution, in that people who survived the extreme episode could have been more susceptible to influenza.

A recent analysis of lung tissues from persons who died during the smog episode found soot and other particle types in the lung (Hunt et al. 2003). The particulate matter (PM) was an aggregation of ultrafine PM (PM with an aerodynamic diameter $\leq 0.1\,\mu\mathrm{m}$) and PM with a diameter $< 1\,\mu\mathrm{m}$. Carbonaceous PM was found in each compartment of the lung; however, heavy-metal–bearing particles such as lead were identified in the compartments of the lung representing recent exposure and were not found in compartments indicative of longer term exposure.

A review of limited autopsy records available from the Royal London Hospital reported twice as many deaths with chronic obstructive pulmonary disease (COPD) as a major finding from December 1952 through February 1953 as corresponding months in other years (Hunt et al. 2003). COPD mortality has been linked with exposure to air pollution (Karakatsani et al. 2003; Sunyer and Basagana 2001); thus, the autopsy reports support the hypothesis that air-pollution–related deaths continued after the peak episode.

These new lessons learned from the study of the 1952 London smog indicate that the episode had a much larger impact on health than previously reported. Conditions as extreme as the 1952 London smog are unlikely in industrialized nations today; however, London's air pollution still affects human health. For instance, deaths and hospital admissions during the week of an air pollution episode in December 1991 had higher mortality rates and hospital admissions than during the week before the episode and during previous time periods (Anderson et al. 1995). A study of air pollution in London from April 1987 through March 1992, which accounted for influenza, identified associations between daily mortality and concentrations of black smoke and ozone (Anderson et al. 1996).

Exceedingly high air pollution still exists in many parts of the world and causes a substantial health burden. Ezzati et al. (2002) estimated that almost 800,000 deaths are caused each year by urban outdoor air pollution and more than 1.6 million deaths annually from indoor air pollution. Additionally, health effects have been observed at low pollution levels (Vedal et al. 2003). The relationship between air pollution and mortality has been demonstrated in numerous locations using increasingly sophisticated statistical methodology and a variety of study designs (Katsouyanni et al. 1997; Samet et al. 2000). The analysis presented here underscores the need to address modern-day air pollution problems.

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Heat Waves, Urban Climate and Human Health

Wilfried Endlicher, Gerd Jendritzky, Joachim Fischer, Jens-Peter Redlich

Keywords: urban heat island · climate index · human health · human mortality · heat waves · climate change

1 Global Climate Change and Heat Waves

The European heat wave of 2003 was an outstanding weather event. The months of June and August have been nearly everywhere in Germany the warmest months since the start of registrations in 1901. The registered mean air summer temperature was 19, 6 °C, that is 3,4 K higher than the mean value. At August 9 and August 13, 2006, the highest maximum temperatures ever registered in Germany, 40, 2 °C, have been measured in Karlsruhe and in Freiburg. This extreme weather was caused by a blocking action of the westerly circulation due to a stationary wave forming a so-called Omega-weather type. High pressure systems with cloudless sky conditions permitted extreme sun radiation and caused repeatedly record temperatures. In southern Germany 53 hot days with maximum temperatures higher than 30 °C have been registered. This heat wave concerned not only Germany, but large regions of Western Europe with France and Great Britain. Other European countries like Switzerland, Spain, Portugal and Italy have been concerned, too.

The questions are (1), if this extreme event is in relation with the human impact on the climate system via the emission of greenhouse gases and (2), if such events will be more frequent in the future

Based on a climate change simulation the distribution of the maximum temperatures of the summer 2003 indicate that this extreme summer is expected to be a normal one by the end of this century in Central Europe (Beniston 2004, Schär et al. 2004, Meehl & Tebaldi 2004; Fig. 1)!

This is in good accordance with the IPCC (2001) statements that in the future extreme weather events are very likely and that more hot days over nearly all land areas are very likely, too.

2 Thermal Climate in Cities: The Urban Heat Island

The most important "local climate change" due to human building activities is the so-called urban heat island (UHI). Since about half a century many investigations have shown the importance of this phenomenon which is caused by the storage of short wave solar energy in the buildings during the day and its liberation by long wave radiation in the evening and night. The larger the city and the

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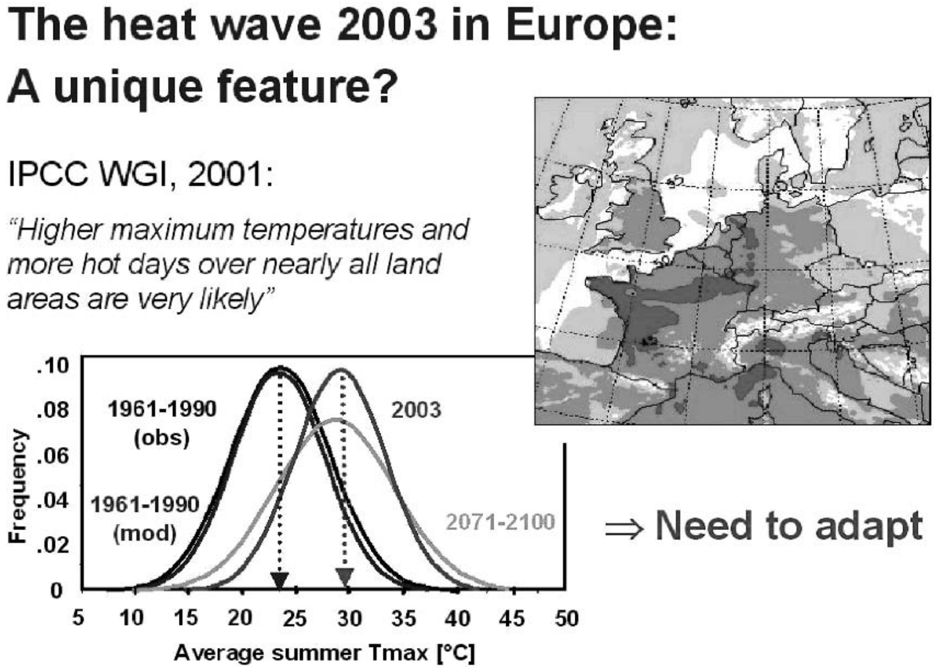

Fig. 1 The heat wave 2003 in Europe: Actually a unique feature, but a normal event in 2071–2100

denser its build-up structure the more intense is the city's heat island or heat archipelago. Maximum differences of about 10 K and more between city centers and rural areas ($\Delta(\max)T_{urban} - T_{rural}$) have already been registered in large agglomerations. Fig. 2 gives an example of the relation between the size of a city and its maximum heat island.

For sure the UHI is largest in the radiation rich seasons – e.g. the European summer months – or tropical and subtropical climates (e.g. summer dry subtropics of the Mediterranean coasts). Until this point there was only the question of global and local air temperature change. The climatic environment, however, includes much more elements which have to be taken into account, if we like to bring together climatic conditions and human health.

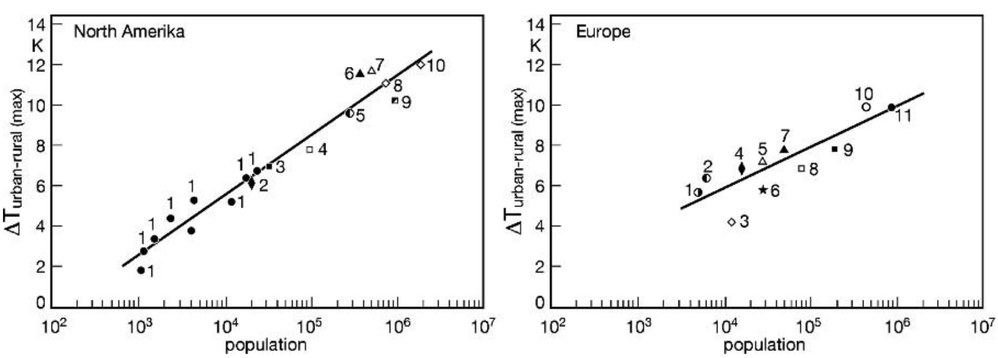

Fig. 2 European and North American Maximum Urban Heat Islands; relation between $\Delta(\max)T_{urban} - T_{rural}$ and log population (Oke 1973); a) North America – 1 Nine Quebec settlements, 2 Corvallis, 3 Palo Alto, 4 San Jose, 5 Hamilton, 6 Edmonton, 7 Winnipeg, 8 San Francisco, 9 Vancouver, 10 Montreal; b) Europe – 1 Lund, 2 Uppsala, 3 Reading, 4 Karlsruhe, 5 Utrecht, 6 Malmö, 7 Sheffield, 8 Munich, 9 Vienna, 10 Berlin, 11 London (data from literature published between 1929 and 1972)

3 The Thermal Environment

One of the fundamental issues in human biometeoriogy is the assessment and forecast of the thermal environment. This is due to the need for human beings to balance their heat budget to a state very close to his/her thermal environment in order to optimise his/her comfort, performance and health. This means to keep heat production and heat loss in a equilibrium in order to keep the body core temperature at a constant level. Heat is produced as a result of the metabolic activity required to perform activities. The body can exchange heat by convection (sensible heat flux), conduction (contact with solids), evaporation (latent heat flux), radiation (long- and short-wave), and respiration (latent and sensible).

The heat exchange between the human body and the thermal environment (Fig. 3) can be described in the form of the energy balance equation which is nothing but the application of the first fundamental law of thermodynamics:

$$M + W + Q^{*}\,(Tmrt,v) + QH\,(Ta,v) + QL\,(e,v) + QSW\,(e,v) + QRe\,(Ta,e) + S = 0 \qquad (1)$$

M	Metabolic rate (activity)
W	Mechanical power (kind of activity)
Q^{*}	Radiation budget (short wave and long wave radiation fluxes)
QH	Turbulent flux of sensible heat (convection)
QL	Turbulent flux of latent heat (diffusion water vapour)
QSW	Turbulent flux of latent heat (sweat evaporation)
QRe	Respiratory heat flux (sensible and latent)
S	Storage

The meteorological input variables include air temperature Ta, water vapour pressure e, wind velocity v, mean radiant temperature Tmrt including short- and long-wave radiation fluxes, in

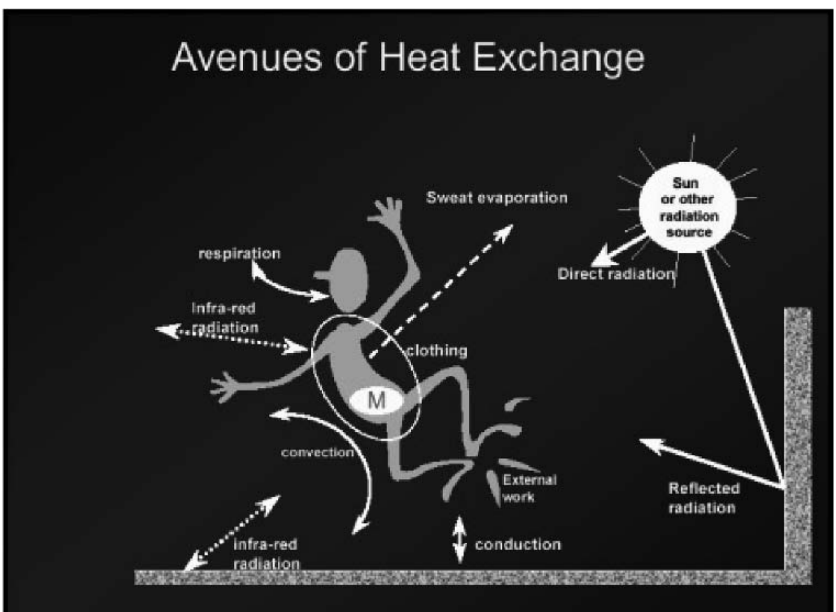

Fig. 3 The human heat budget (Havenith, 2003, in Koppe et al. 2004)

addition to metabolic rate and clothing insulation. In eq.1 the appropriate meteorological variables are attached to the relevant fluxes.

It is important to take into account all these variables for the complete description of the thermal conditions, thermal comfort or discomfort.

Thus, consequently dealing with the thermo-physiologically significant assessment of the thermal environment requires the application of a complete heat budget model that takes all mechanisms of heat exchange into account as described in eq. 1. Such models possess the essential attributes to be utilised operationally in most biometeorological applications in all climates, regions, seasons, and scales. Fanger's (1970) PMV-(Predicted Mean Vote) equation can e.g. be considered among the advanced heat budget models. This approach is the basis for the operational thermal assessment procedure Klima-Michel-model (Jendritzky et al., 1979; Jendritzky et al., 1990) of the German national weather service DWD with the output parameter "perceived temperature, PT" (Staiger et al., 1997) that considers a certain degree of adaptation by various clothing.

4 Thermo-Physiological Modeling and Urbanization

The importance of the described thermo-physiologic approach is evident. Simple indices taking into account only temperature and humidity are not any longer sufficient. An example from an urban street canyon shows it clearly (Fig. 4): Radiation received at the human body is totally different at the sunny and shadowed site of the street and so are the surface temperatures of the asphalt and the walls. Wind speed is highest in the middle of the street. Air temperature, however, is nearly the same at both sides of the street canyon. Only the modeled Perceived Temperature represent the thermal environment in a correct way (as we perceive when we cross the street from the sun to the shadow).

In Berlin, the heat load in August 2003 has been modeled comparing 3 different neighbourhoods (Fig. 5): The open esplanade Alexanderplatz in the eastern city center with only a few trees, few shaded areas and a large sky view factor (a), the densely built-up Potsdamer Platz, in the city center as well, but with a smaller sky view factor and larger shaded surfaces (b), and the garden suburb

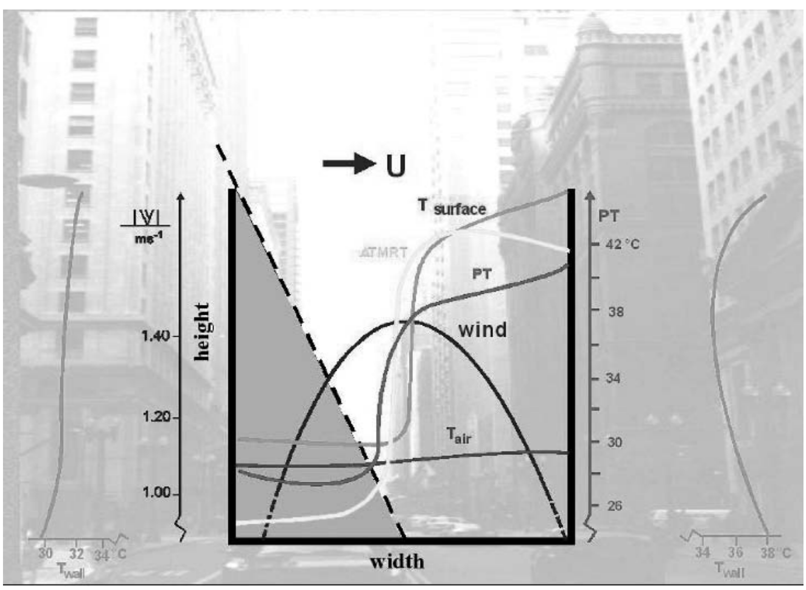

Fig. 4 Meteorological and biometeorological conditions in a cross section of a street

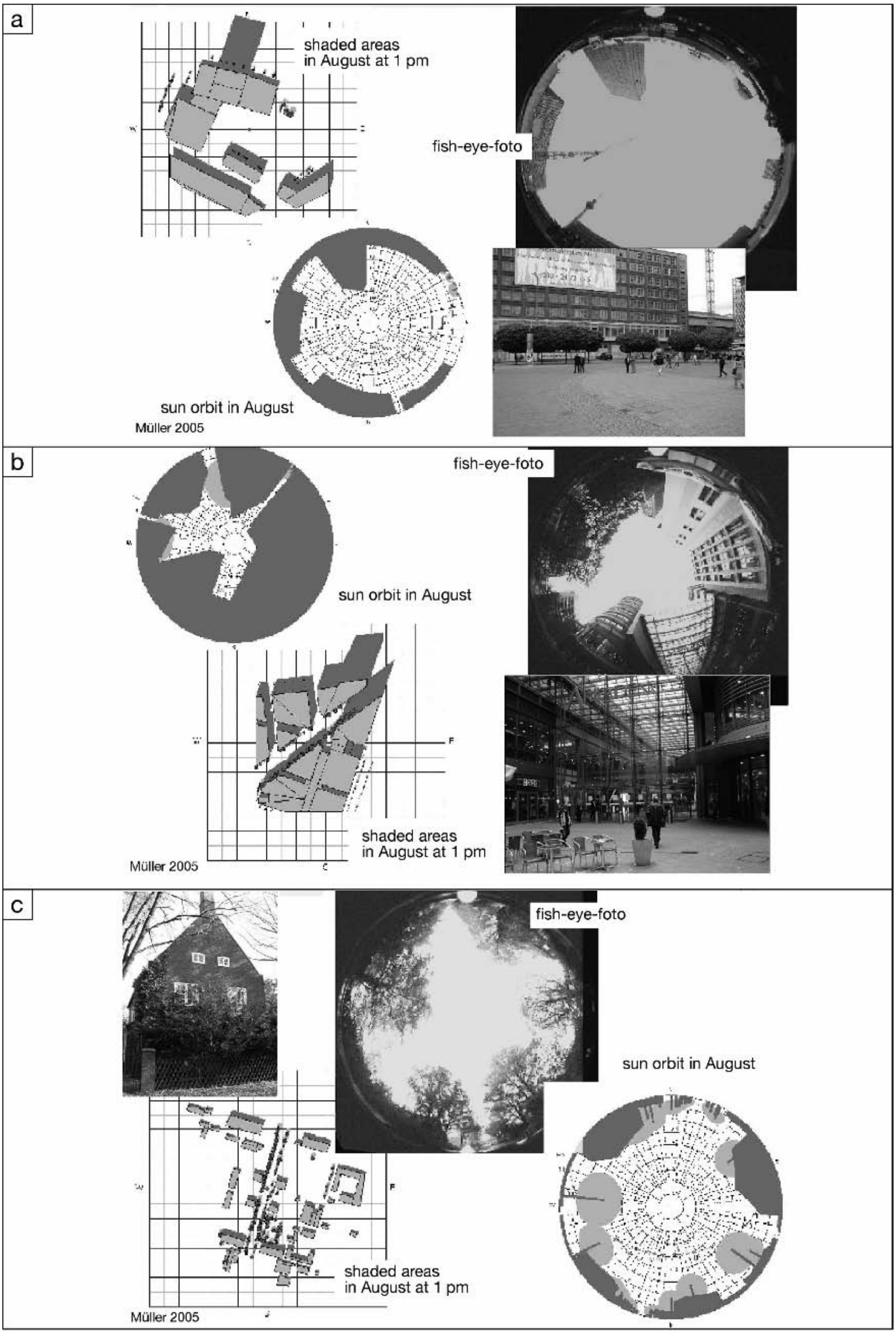

Fig. 5 Heat load in different neighbourhoods of Berlin, August 2003; a) City center – Alexanderplatz, b) City center – Potsdamer Platz, c) Garden suburb Dahlem, d) frequency of different heat load in comparison of the 3 neighbourhoods

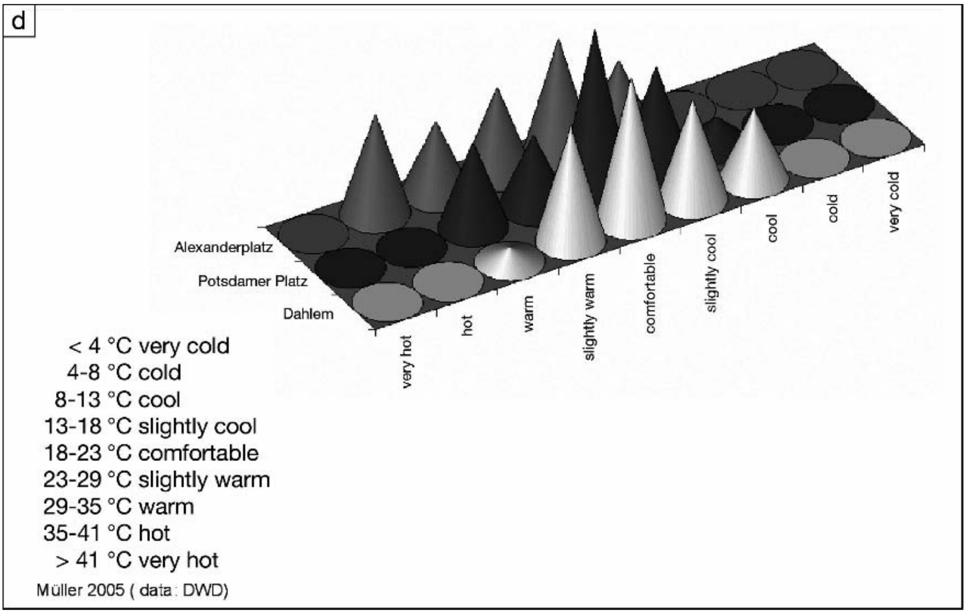

Fig. 5 (continued)

Dahlem with a reduced sky view factor due to numerous trees (c). The model results (d) show, that the Alexanderplatz was the neighbourhood with the highest heat load, followed by the Potsdamer Platz and Dahlem. Between August 1 – 15, 2003, 12 days with a moderate to high heat load have been registered at the Alexanderplatz. The Potsdamer Platz showed a lower heat load, because of its reduced sky view factor, but a higher one than Dahlem. So, densely built-up neighbourhoods show high heat loads, even if the direct radiation is reduced, due to the heat storage in walls and sealed surfaces which are missing nearly completely in the garden suburb.

5 Heat Waves and Human Health: the Future has Already Started

There are numerous epidemiological studies published which impressively show worldwide the health impact of extreme thermal conditions such as heat waves. During the hot summer 2003 in Europe, in particular in August, between 35.000 and 55.000 heat related extra deaths occurred. It can be assumed that the urban heat island effect (UHI) has intensified the regional heat load. If the calculations are correct – and there are no doubts about, that heat waves like the European one in 2003 will be in the future more frequent and perhaps even more intense, then we have to develop different ways of coping with this heat vulnerability. An impressive example of the already now enhanced mortality during heat waves in Southern Germany is given in fig. 6. The figure shows the deviation of the total mortality rate during the days before and after the average of nine extreme heat waves (1968-1997). The mortality curve follows closely the modelled “Perceived Temperature” (Jendritzky et al. www.utci.de/documents/Perceived_Temperature).

5.1 The Universal Thermal Climate Index UTCI

The International Society on Biometeorology ISB recognised the issue presented above some years ago and established a Commission “On the development of a Universal Thermal Climate Index

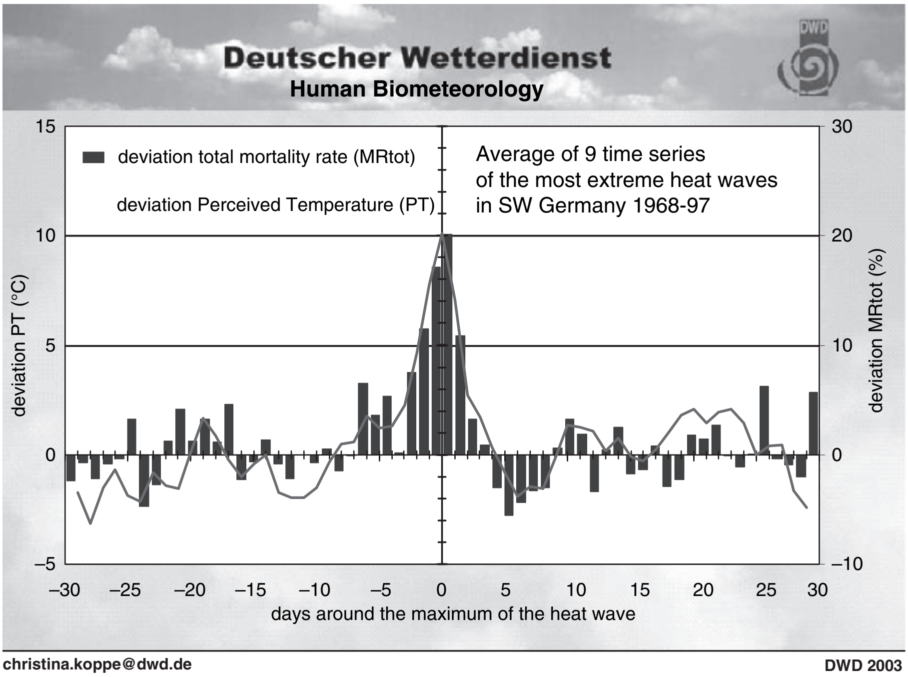

Fig. 6 Deviation of the total mortality rate during the days around the maximum of a heat wave; average of nine time series of the most extreme heat waves in Southwest Germany (1968-1997); the curve of the modeled "Perceived Temperature" follows closely the mortality deviation (Koppe et al. 2004)

UTCI" (Jendritzky et al., 2002; http://www.utci.de). Since 2005 the COST Action 730 (Cooperation in Science and Technical Development) of the European Science Foundation ESF provides the basis that scientists can join together on a regular basis in order to achieve significant progress in deriving such an index. Aim is an international standard based on scientific progress in human response related thermo-physiological modelling of the last 30 years (Fiala et al., 2001; Kusch et al. 2004) including the acclimatisation issue. This work will finally be made available in a WMO "Guideline on the Thermal Environment" so that every national health service can then easily apply the state-of-the-art procedure for its specific purposes. The guideline will provide numerous examples for applications and solutions for handling meteorological input data.

5.2 The Need for Adaption

The need for adaptation to the growing problems of the thermal environment is evident. In addition demographic factors like the growing number of senior citizens have to taken into account, because elderly people have more difficulties to deal with thermal stress. Therefore the development of suitable measures for adaption is necessary. Two levels have to be distinguished in order to reduce the urban thermal stress indoors and outdoors:
Long-term and short term adaption to future enhanced heat stress.

5.2.1 Long-Term Adaption

Long-term measures need a longer implementation time and apply especially to *urban planning* and *building design.*

Outdoors:

- Creation of green and open spaces, especially with trees
- Ventilation and air flow
- Enhancement of albedo (less heat storage by absorption of short wave radiation)
- Reduction of anthropogenic heat production

Indoors:

- Thermal capacity of buildings
- Position of apartments
- Control of solar irradiation
- Passive cooling

5.2.2 Short-Term Adaption and Heat Health Warning Systems

Short-term adaption measures are possible to realize in a short run and apply especially to the set-up of *Heat Health Warning Systems (HHWS)* which take into account the actual weather and the forecast of the next few days. Lives would have been saved if adequate Heat Health Warning Systems would have been in action, as promoted by the WMO/WHO/UNEP showcase projects in Rome and Shanghai. Such systems are based on biometeorological forecasts expecting exceeding of an agreed threshold (heat load forecast). The following interventions (a locally adjusted emergency response plan) belong then to the responsibility of the public health service. HHWSs must be prepared in advance with complete descriptions of all processes (Kovats & Jendritzky 2006). In addition self-organizing networks will allow to take into account as well outdoor as indoor conditions.

Sensor systems (e.g. HHWS) require a communication infrastructure capable of transporting captured sensor data (temperature, humidity, air pollution, etc.) to a central data collection and processing center. This communication infrastructure must be low-cost, real-time, dense, and quickly adaptable. No existing technology satisfies/optimizes all requirements at the same time. Hence, solutions are needed that are tailored to specific usage scenarios.

The Berlin RoofNet project [www.berlinroofnet.de] has demonstrated that it is possible to build an autonomous wireless information network in the city of Berlin, at a moderate budget. Main points in its design are: The network evolves spontaneously (ad-hoc), without explicit prior planning. It does not require a central administrative authority, like an operator. Nodes (access points) communicate with each other and with client stations (laptops, sensor stations). The network is self-organizing: nodes create network structures automatically and determine configuration parameters without human intervention. Inexpensive Commercial Off The Shelf (COTS) hardware is used, such as IEEE 802.11 g WLAN, that operates in the unlicensed 2.4 GHz ISM band. Communication is close to real-time (delay $\sim$ 0.5-1 seconds), robust (mesh-structure with redundant paths) and based on the Internet Protocol, which allows for easy integration with existing applications and with the public Internet (where available).

The option of reducing cost even further and simultaneously increasing the coverage area by a large factor is available, at the cost of significantly increased delay times. An experimental technology called DTN (Delay Tolerant Networks) uses mobile objects (busses, cars, even people) as carriers of information: Sensors capture data and aggregate them locally - using small-scale wireless networks. Selected busses are equipped with access stations that, if they enter the communication range of the sensor station/network, retrieve all data within seconds, store it, carry it, and finally deliver it to a central data collection station. Dependent on bus frequency and reliability of bus schedules, delay times range between few hours and 1 day. In addition to carrying sensor data, the DTN infrastructure could also carry messages of people living in the covered areas (similar to E-mail). This additional use of the proposed communication infrastructure may not be attractive to people with always-online broadband Internet access, but for poor people it may be the only means for tele-communication available to them.

For a later integration with already existing applications of the National Meteorological Services it is of strategic importance that the communication infrastructure is IP-based. An evaluation of the behaviour of the network infrastructures is almost impossible without accompanying model investigations. By experimental adjustment of the net topology and the protocol software iterative improvements in the behaviour and thus positive effects on the quality of the planned communication infrastructure can be gained. The expenditure, which has be to be paid repeatedly for each concrete configuration and installation of a monitoring system, should be strongly reduced by simulations of the time behaviour of such systems with consideration of the dynamic load and the changing environmental influences to each communication computer of the monitoring system.

In order to be able to realize this complex task, a library of parameterisable model components should be developed, which enables the efficient configuration of the monitoring system models with appropriate experimentation and evaluation support. An established approach for that is an adoption of the OdemX library (Fischer & Ahrens 1996; Gerstenberger 2003), developed in C++ for the modelling and simulation of time-discrete and time-continuous processes.

6 Conclusion

The superposition of five factors: Heat waves, thermal stress, rapid urbanization, growing number of elderly people and global climate change exclude simple solutions. Nevertheless adaptation measures are possible (Kirch et al. 2006). Intelligent short-adaption measures like Heat Health Warning Systems can be installed already nowadays. Long-adaption measures need more time to be developed and introduced; however, global climate change and rapid urbanization, especially the growing number of megacities, will complicate this issue in the future even more.

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Section IV
The Biosphere

The study of ecological systems in urbanizing landscapes has expanded dramatically in the last decade. Our selection of papers draws heavily from this period, but places modern research into a historical context by including a translation of an early summary work on general ecological conditions by Sukopp (1973). This and others by Professor Sukopp throughout the volume (e.g., Sukopp & Hejny (1990), emphasize the important foundations he and his colleagues in Berlin laid for the field of Urban Ecology. We selected reviews of biotic factors (McDonnell 1997, Kowarik 1995, Marzluff 2005) and included papers that represent how humans directly mold ecological composition (Hope et al. 2003) and how a variety of taxa (plants, birds, fishes, bats, and spiders) respond to human actions. The rapid evolutionary response generally described by Palumbi (2001) in Section I is now grounded in urban systems by the pioneering work of Johnston & Selander (1969). This collection provides a good entry into the variety of ways plants and animals respond to urbanization. It also serves to emphasize the fact, that understanding how and why humans interact with the natural system and explaining the interactions between the socioeconomic and the natural system, as Hope et al. (2003) begin to explore, is a novel and crucial point of today‘s urban ecology.

Regardless of the taxa or scale of study, several important generalizations emerge from our selection of papers. Cities are typically warmer, drier, nutrient–laden, and floristically enriched by human activity. The enhanced roughness of the urban morphology (topography), the sealing of the soils and the high heat storage capacity of the built-up environment described in Section III change the characteristics of the surrounding regional climate and soil conditions on a local scale with consequences for plants, animals, and human beings as well. The lush and diverse vegetation found in portions of cities, as well as natural and man-made waterways often attracts a wide array of birds, mammals, insects, and fish. But nutrient enrichment and extensive land conversion often allow a few tolerant species to attain extremely dense populations. The importance of plant diversity to birds, bats, and insects is consistent and suggests that where people allow structurally complex vegetation to occur, invertebrate and vertebrate diversity will prosper.

Human facilitation of invasions by exotic species is probably the greatest concern to the biological diversity of cities. Global climate change and the nearly unlimited exchange of persons and goods, typical in a globalized world, intensify today the pressure on the endemic flora and fauna. Globalized lifestyles of city dwellers reduce regional and local identity of the urban systems, too. The presence of some exotic species can harmlessly increase local urban biodiversity and provide city dwellers with cultural, health, and aesthetic benefits. However, at global scales, people often homogenize the faunas and floras of cities so that in effect each city, worldwide, may come to harbor the same few birds and plants (as reviewed by Blair 2001). This leads us to suggest two general keys to maintaining biological diversity in urbanizing areas. First, promote structurally diverse green spaces to increase local diversity within the city. Second, seek to increase diversity among green spaces within and especially between cities. Doing the same thing everywhere may increase local diversity at the expense of regional, national, and global diversity.

Appreciating the changes in ecosystem function and composition in urban areas causes us to recognize that three strategies will be needed to maintain global biological diversity. (1) Some reserves will be needed to maintain sensitive species that are intolerant of people. This strategy will be rarely possible in highly urban areas, because large spaces are required. (2) Some restoration of damaged habitats will also be needed to increase the functioning of ecological systems that are impacted by people. This may be possible, to limited effect, in urban areas. Reduction of air pollution, restoration of streams, lakes, rivers, and wetlands may be especially important in urban areas as clean water directly benefits people and allows native plant and fish communities to persist in urban areas. Air and water are the two most important elements for life. (3) Lastly, reconciliation (shaping human domination of place in such a way as to allow other species tolerant of our presence to live with us; Rosenzweig 2004) is necessary, and probably the most important strategy to embrace in highly urbanized systems. Reconciling to conserve those species that tolerate human presence allows managers to favor a diversity of species in urban areas. Reconciling the needs of people with conservation of biological diversity allows us to celebrate the plants and animals that tolerate us, even some nonnative ones that are valued for aesthetic, health, and cultural reasons. Generating a positive association between people and nature in urbanizing areas is an important goal of urban reconciliation ecology that may build human compassion, values, and political will to reserve and restore ecological systems elsewhere. Reconciliation will favor tolerant species and passively discourage intolerant species in urban systems.

A review of *The Biosphere* helps resolve some of the issues unveiled in the previous section. We learn that community and population studies contribute important insights into urban ecosystem function. Studies *in* urban ecosystems inform studies *of* urban ecosystems. Studies of urban ecosystems are not fundamentally different from studies in urban ecosystems, but they are often done at larger spatial scales and focus on ecosystem processes (energy flow, nutrient cycling) rather than population and community dynamics or individual adaptation. Future studies that fully investigate ecological hierarchy, from individuals to ecosystems, may be able to truly study the ecology of urban ecosystems. At present we have important and compelling studies of worldwide ecology at a variety of points in the hierarchy, but not an integrated Urban Ecology. Future scholars have much to build on, but much left to do.

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The City as a Subject for Ecological Research

Herbert Sukopp

Keywords: plant diversity · exotic species · native species · human population · biosphere · urban climate · ecological characteristics of city · plant succession · growing season

Development and Importance of Ecological Investigations of the City as a Habitat

The city, in today's meaning for Central Europe, may be considered in the context of the development of modern technology and new energy sources. However, historically, cities may be considered in a narrower context, associated with the erratic increase in the world population. During the 1960s, the percentage of the population living in urban areas (i.e., areas with more than 20,000 inhabitants) was estimated to be 30% world wide (with the highest rates in North America 46%, Northwest Europe 54%, and Australia and New Zealand 65%). Thus, it is understandable that the most recent ecology has been focussed on the most densely populated regions (Aschenbrenner et al., 1970, 1972, 1974a,b: Dansereau 1970; Müller 1972, Fitter 1946, Kieran 1959, Miyawaki et al., 1971, Peters 1954, Rublowsky 1967).

The often repeated statement that each city is generally hostile to life, seems to be disproved in several ways. It was surprising to find that the first investigations of urban locations, showed that, with existing complications, purely anthropogenic biotopes can offer suitable habitats with characteristic species combinations. The species combinations of such habitats vary between industrial facilities, railways, ports, rubbish dumps, and so on, and may be different from those known from other habitats.

The flora of economically important species have been carefully researched in only three German cities: in Stuttgart (e.g. Kreh 1951), Leipzig (e.g. Gutte 1971), and in Berlin. The fauna has been researched in Hamburg, Kiel, Dortmund and Berlin (Erz 1964, Mulsow 1968, Weidner 1952, Wendland 1971). Ecology is now stronger and more systematic than in previous years; human influences in the conurbations have been studied, and research programs have been developed.

In recent years, ecological research projects have been initiated in Berlin (Kunick 1973, Runge 1973, Sukopp 1966, Zacharias 1972), and the preliminary results will be reported here.

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Characteristics of the City Habitat

At first it may appear appropriate to consider the density of the human population as the decisive characteristic of a city, (whereas nearly every single "artificial" location has its analogy in nature; Strawinski 1966). However, classification of the city areas according to the residential density would result in a distorted picture, because in each city, one can observe a tendency for the depopulation of the city centre, with a decline from the most densely structured city centre to the garden suburbs. The number of people who actually use the area (such as workers and those traversing the area) is more difficult to determine.

An additional characteristic is the number and magnitude of anthropogenic interferences which exclude organisms (this occurs because the influences are arythmic; Erz 1964, Schweiger 1960). Furthermore, eutrophication is characteristic for human settlements in general, and is especially so in cities. Historically, the city of today can be seen as the end stage of a development, which develops from the village, to a small, and then middle-sized town.

Nevertheless, the total area does not appear to be a suitable criterion by which to understand the city. It is more of a mosaic habitat, formed from many diverse smaller biotopes. Certainly, the amount of technically fallow land is high (Hamburg and Bremen each 40%, Federal Republic of Germany on average 10%).

For biogeographical characteristics of a city, one can consider the original habitat of the apophytes, the number, immigration route, and geographical origin of the hemerochores, as well as the decline in plants. The composition of the synanthropic flora of the cities, reflects the influence of the adjacent environment, while under the apophytes, partly wood and bushes, and partly meadow plants, prevail (Krawiecowa & Rostanski 1971). In cities situated on bigger rivers, water and bank plants comprise approximately 10–17% of the apophytes.

In addition, the degree of development of a city (urbanisation, industrialisation, development of commerce and traffic), affects the spectrum of the synanthropic flora in the flowing characteristic ways:

Number of Non-Native Species

Falinksi (1971) compared different settlements from villages and towns (Table 1). This shows many difficulties in comparing material which originate from different settlements, with different borders, and reported by different authors. One would not expect a linear relationship between the number of inhabitants and the number of non-native species, if for example, important data on trade and traffic are missing. However, these data nevertheless do provide important clues. A map of the species number of non-native species in 47 places in Finland published by Erkamo (1959), states only the absolute species numbers of the non-native species, and does not place this in perspective with the total species number of the respective flora.

Among native plants on ruderal sites, perennials predominate, whereas amongst the non-natives, annual and biennial plants dominate (for example Misiewicz 1971 in Falinski 1971).

Table 1 Composition of the synanthropogenic flora of some villages and cities in Poland (including Ephemerophytes; in %)

	Native	Non-native
forest settlements	70–80	20–30
villages	70	30
towns	60–65	35–40
cities	50–60	40–50
large cities	30–50	50–70

Immigration of Non-Native Species

From the immigration of the introduced non-native species, one can distinguish between those that were intentionally introduced and cultivated by people, (and then spread beyond the cultivation area), and accidental introductions. A study of the species from 47 towns in Finland summarized by Erkamo (1959) showed that:

The accidental introductions (naturalised) form the largest group in most places. The accidental introductions are the largest group in only some (not all) large cities (Helsinki, Turku, Vaasa, Oulo, and Viipuri). For those that escaped from cultivation, generally the opposite applies: the accidental species are mostly (in 36 settlements out of 47 (69%)) more richly represented than are the naturalised ones. This is always so where the species pool is rich. The increase in species results from transportation and commerce.

The Geographic Origins of the Non-Native Species

Most the species which prefer old settlement areas originally had a southern distribution (Saarisalo-Taubert 1963). Some 3/5 of the archaeophytes and neophytes of the Berlin flora (on ruderal sites), most of which are annuals, are resident in, and spread from, warmer regions. Their settlement and establishment in moderate climatic regions of Europe, has been made possible through the climate of human settlements. The climatic effects of compact industrial areas and large cities result in a warmer and drier urban climate (Scholz 1960).

Reduction of Species Number

The change in habitat conditions results in a strong reduction in the species number. The flora in the surroundings of large cities and industrial areas shows a strong decline (Table 2). Because of the different methods of data acquisition in the individual areas, the table represents only an approximate picture of the loss of plant species (between 6 and 13% of the stock one hundred years ago).

Since 1859, some 114 species of ferns and flowering plants are presumed extinct or become extinct in Berlin. These represent a loss of about 12% of the native and archaeophytic species in one hundred years. Strong declines have been shown particularly by the plants of the pond weed family (Potamogetonaceae), with a loss of about 41% of their species number in one hundred years, and the orchid family (Orchidaceae).

In areas intensively changed by people, we do not know of a single case in which a plant would have disappeared due to natural causes. Under anthropogenic influences, habitat changes have caused more losses than direct collecting of plants. Stricker (1962) had emphasised the meaning of the change in habitats, by noting that the plants are not initially in decline, but rather it is a decline in habitats at which the plants can grow. Important factors accounting for the decline in

Table 2 Loss of species of ferns and flowering plants in the surroundings of some cities. a = archaeophytes; I = introduced; n = native; t = naturalized

Area	(km^2)	# of species	% extinct or missing
Paderborn	1,250	684 n	6
Stuttgart	1,000	1,080	4 n + 2 i
Berlin	884	965 n + a	12
Aargau	1,404	~1,300	16
South Lancashire	3,100	839 n + t	7.9

Table 3 Decline in plant communities in Berlin, illustrated by the number of extinct and presumed extinct ferns and flowering plants (modified after Sukopp, 1966)

Formation or Habitat	Total number of species	Species extinct or missing %	#	Change of surface, last 100 years
weeds of arable fields	90	17	15	–
bogs	61	16	10	–
waters	176	14	25	–
dry grassland	148	14	21	–
moist grassland	91	13	12	–
forest, hedges, skirt vegetation	236	10	23	
ruderal vegetation	121	7	8	+
meadows	42	0	0	–
Region of Berlin	965		114	

species of vascular plants are a lowering of groundwater, eutrophication, and water contamination. Subsequently, mechanical factors such as building, digging, deposits, and recultivation of fallow land follow. In future, the use of herbicides will play a significant role in the decrease in species. The decrease in plant communities in Berlin can be quantified using the number of extinct and presumed extinct species (Table 3).

The strongest decline is shown by vegetation of the fields, bogs, waters and the acidic grasslands, with some 14-17% of the species lost, then follows moist meadows and woodland. The extinction of species in Berlin freshwaters is particularly marked. The extent of the decline in water plants is clearly seen if one considers that in the period under consideration, the surface areas of waters did not change, (with the exception of ponds). Despite this, there has still been a strong decline in number of species. Also for fields, the decline of coverage is not alone decisive. The ruderal vegetation finally shows a stronger increase of area – today nearly half of the city area – nevertheless a marked decline of native and archaeophytic species. Clearly, this loss (of species number) is by far outweighed by the arrival of neophytic ruderal plants.

Methods for Ecological Research with the Aim of Classification of Large City Areas

In contrast to the ecological approaches described later in which the condition of the city – or a part of a city – is characterised at a particular point in time (a "snap shot"), are the historical and historical – ecological methods in which the past events in a particular place are represented. From the summary of the observed data from the past, (flora and fauna lists, measurements, maps and so on), one can reconstruct the history of the flora, fauna, climate and soil (that is to say, the history of the landscape), and one can determine statistical characteristics (such as species numbers, percentage of neophytes and so on, and mean and range values for climate).

Historical Classification

When the available information permits a comparison with the former species pool, historical representations of changes in flora and fauna are very informative. Causes of the changes are introductions of species via trade and traffic and important changes in habitats. Gusev (1968) represented the changes in ruderaflora in the Leningrad area (now St. Petersberg) in the last 200 years. The recent ruderal flora comprises more than 350 species (excluding those found only rarely (that is to say 1–3 times)). Nearly half of the ruderal flora belongs to only three plant families: Compositae, Cruciferae, and Gramineae. Some 75% of the species are introduced. More than 130 species were first introduced, or brought in, in the last 200 years, including (e.g.) *Matricaria metricarioides* and *Convolvulus arvensis*, which are common today. In contrast, some ruderal plants became rare or extinct (for example *Bromus arvensis*).

A description of the changes in the synanthropic flora Posens between 1950 and 1970 was given by Zukowski (in Falinski 1971), analysing an increasing "continentalization" of the Flora Posens.

According to Scholz (1960), the number of naturalised neophytic ruderal plant species in Berlin were as follows: 20 in 1787, 51 in 1884, and 79 in 1959. One can determine that this amount is directly related to the human population size. In 1860, the dramatic increase in human population began, which corresponded to a sharp increase in the number of ruderal species (Fig. 1). The increase in urban weeds corresponds to a stronger decline of native and archaeophytic wild plants (Table 3).

Historical records can be provided not only by organisms, but also by changes in ecological factors. In the city of Berlin, investigations of temperature and groundwater were undertaken by Scherhag (1963) and Denner (1958) respectively. When earlier studies permit comparisons (for example in Berlin the maps of the phreatic level from 1870, 1916, 1929 and later, Denner), one can establish not only a temporal, but also a spatial structure of the city region.

Historical – Ecological Approach

Under the assumption that the oldest quarters of a city, today exemplify the oldest urban ecosystems, one can determine relationships between the different ages of settlement areas and their biotic dominant communities. A study of the flora of three small cities in Finland (Saarisalo-Taubert 1963) showed that the current distribution of the flora accompanying old settlements is determined by favourable edaphic and microclimatic conditions of the old settlements. Where these conditions are missing, these species are only present occasionally or rarely. The more fastidious the particular species are, the more suitable are the conditions in the older settlements. The so-called "friends of old settlements" could be absolutely new arrivals, or they could belong to the oldest species. Figure 2

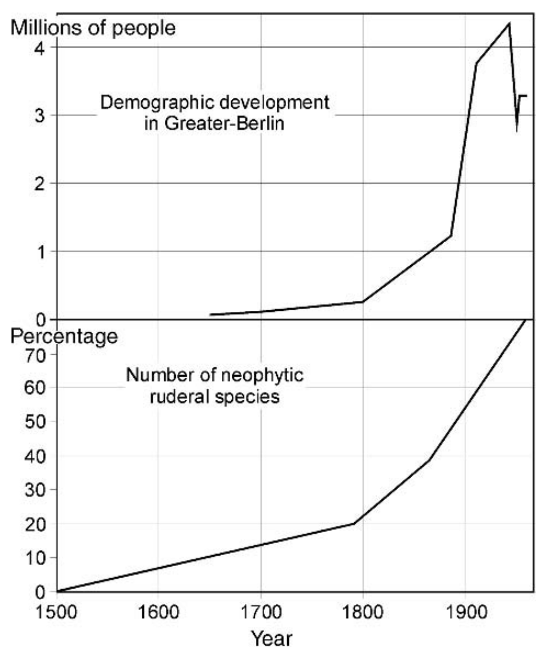

Fig. 1 Population growth in Berlin, and the number of neophytic ruderal species

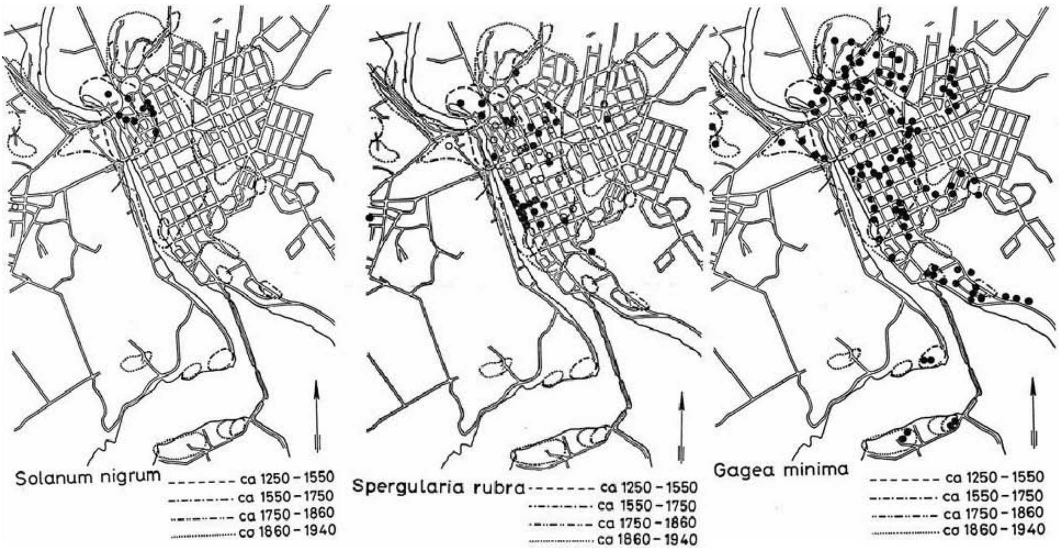

Fig. 2 Distribution of species in different areas of old settlements in the south Finnish city of Porvoo (after Saarisalo-Taubert 1963)

shows the distribution of species in different areas of old settlement in Porvoo. The distribution of species is similar in each of the three Finnish cities. However, one cannot necessarily carry over the knowledge of the indicator value of a single species, to other cities.

In general, the predominantly southern origin of indicator species has been proven. Zajac (in Falinski 1971) had interpreted the distribution maps of wall and garden plants in Bielsko-Biala (southwest Crakow), in connection with the city development, in relation to the distribution of other species, and the responsible critical recent climatic and edaphic factors.

In the studied biotopes, this floristic process corresponds to the phosphate mapping (Arrhenius 1931). Here, the appearance of sites with high phosphate contents in the soil corresponds to the appearance of settlements.

It is desirable, that similar studies be undertaken in central European cities that were not destroyed in the war. For many towns, and especially for large cities, the historical-ecological approach is not suitable. On one hand, due to war damage and more recent restoration, the oldest districts have been completely destroyed; and on the other hand, buildings are new.

Ecological Approaches

Classification According to Land Use

With studies of the flora, one can ascertain which species characteristically accompany certain land uses. Although the concept that, for example, meadow plants only grow on meadows, in the public spirit such relationships are not yet generally accepted for city and industrial biotopes. Almquist (1957) and Militzer (1961) have presented maps showing the spread of plants along railway lines. In Berlin for example, the distribution of a certain evening primrose (*Oenothera coronifera*), is linked to the railway lines. Sukopp and Scholz (1965) have reported about specific “canal plants” in Berlin. Such Berlin canal plants include the willow-leaved dock (*Rumex triangulivalvis*), and the garden angelica (*Angelica archangelica*). Other examples for the characteristic large city biotopes in Berlin are: tree of heaven (*Ailanthus altissima*) – spreading in industrial areas, butterfly bush (*Buddleja davidii*) – spreading in the inner city (where housing developed in the post-war years (Kunick 1970)), alien species from bird seed for the built-up areas, and bulbous meadowgrass *Poa bulbosa* (Sukopp & Scholz 1968) for bathing beaches and camps. A comparison of the areas with the same use, from the outskirts to the city centre, leads to formulation of ecological series**.** An example is given for recreational areas in Berlin (Fig. 3).

Nearly all the species groups decline with decreasing surface area and increasing recreational use, this disappearance may take place gradually, in other cases it may happen unevenly and abruptly. Only a few species are commonly encountered in the urban green spaces (*Euphorbia peplus*, *Galinsoga ciliata*, *Sisymbrium loeselii*, *Urtica urens*).

Spatial Classification

The natural classification can be of significant assistance in our understanding of the development of artificial landscape. However, today, natural borders often lie within settlements and are difficult to distinguish, and thus they alone are insufficient as classification characters.

With the classification of an urban agglomeration according to topographical units, it becomes evident that in addition to the present land use, additional regional influences operate to allow the combination of smaller mosaic sites into a larger whole. From the outskirts toward the city centre, there are clear gradients. “When the mosaic character of the biotopes remains, differences in the settlements populations can be determined, along with the proportion of a whole area occupied by single biotopes, especially those which are indicative of urbanisation” (Kühnelt 1955, pg. 35). In many cities, a clear zonation of the vegetation with epiphytic lichens and mosses has been determined (Ando & Taoda 1967; Barkman 1969). Generally, in the centre of a city only a few species are found, whereas in the periphery, the species show greater vitality, and more species can thrive. Observations which show a gradient away from and towards a city, include: climate data (Kratzer 1956, Scherhag 1963, Schlaak 1963), phenological data (Hoffmann, dissertation, Zacharias 1972), air pollution (Bracht 1960), frost damage (Joachim 1957), species which decrease in the city (summarised for mosses and lichens by Barkman 1969), and species which are only present in the city (“city plants” Gutte 1971, Sukopp 1971). A simple model of a city and the changes of its biosphere is given in Fig. 4, below.

Through urban building and economic activity, the city becomes divided into zones of densely built-up areas, row and edge buildings, and loosely built-up areas. The outskirts are characterised by allotment gardens and parks, as well as refuse dumps, rubble heaps, and sewage farms. In the surrounding countryside, fields and forests dominate. The urban building and economic activity result in pollution and warming of the air, changes in groundwater level, and large deposits. The volume of imported building materials, raw materials for manufacturing, and food, is greater than that exported, and this results in the ground level rising over time (numbers given by Peters 1954, Fels 1967). Due to the magnitude of the artificial (cultural) layers, eutrophication at many locations is combined with the artificial (cultural) layer, as well as condensation or sealing of the soil within the settlement. The eutrophication affects not only urban rubbish, refuse tips, and sewage farms, but also nearly all waters. In general, there is little space for vegetation within the city. However, exceptions include numerous street trees, and extended areas of ruderal vegetation on war damaged areas in the middle of the city Similarly, Kühnelt (1955) and Schweiger (1969) have delineated city zones based on their land use.

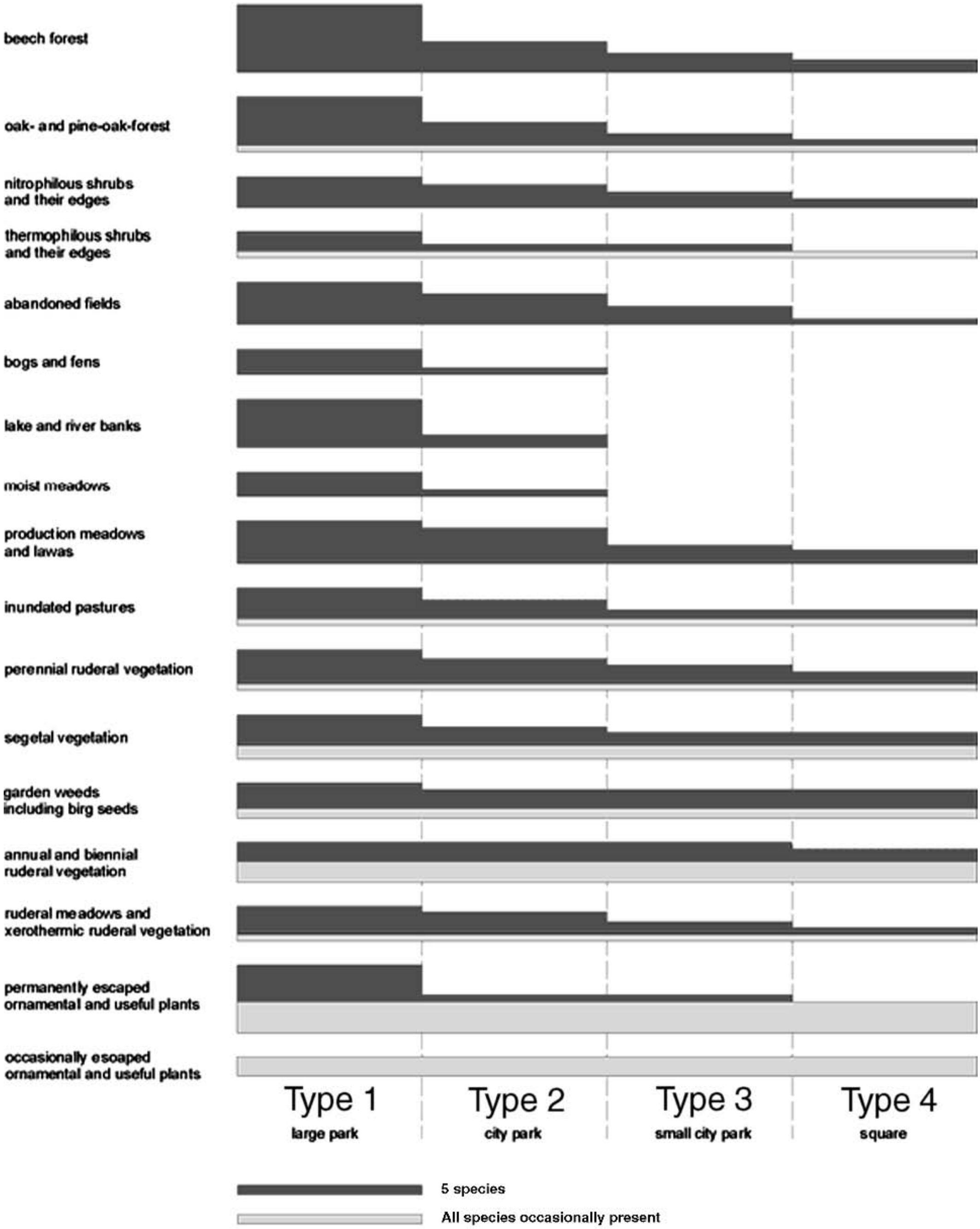

Fig. 3 Occurrence of species groups in different types of recreational areas in Berlin (from Kunick 1970). Large parks ranged from 60–140 ha and contained 250–450 species. City parks ranged from 10–25 ha and contained 120–150 species. Small city parks were 1 ha in size and held 60–140 species. Squares were 1 ha in size and held 40–120 species

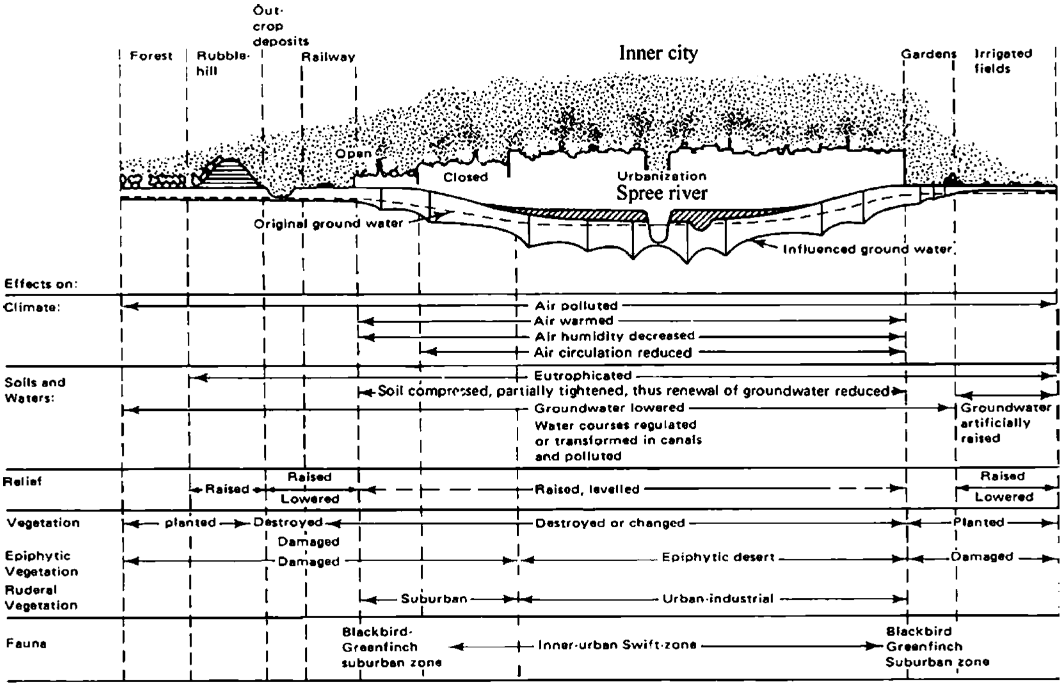

Fig. 4 Systematic portrayal of important changes in a city's biosphere

Classification with Berlin as an Example: Ecological Characteristics of the Berlin City Region.

Macro-Climate

According to the climate atlas of the GDR (German Democratic Republic), and the atlas of German environment, Berlin has an east German inland climate, with warm summers, relatively cold winters, and modest precipitation. The representative weather station for Berlin in Dahlem (at the edge of the densely built-up city centre), reported the following mean data for the period 1891–1951: July $+18.3\,^{\circ}$C, January $-0.7\,^{\circ}$C. The mean annual precipitation in the Berlin city area lies between 540 and 600 mm.

Urban Climate

The effects of the city on the climate have been summarised in a monograph by Kratzer (1956). The impact of the individual climate of such an extensive city as Berlin, for the naturalization of southern ruderal species has been demonstrated in detail by Scholz (1960). The following aspects detail the essential features of the city climate:

Higher Temperatures in Berlin Compared to the Surroundings

The inner districts of Berlin are, in all seasons, $1\,^{\circ}$C warmer than the surroundings. The simultaneous temperatures during measurements by car are approximately ten times larger than with the mean temperature. According to the amplitude, the temperatures in the region in summer and winter are not significantly different, giving characteristic seasonal shifting in thermal differences. In winter, the inner city shows a slight condensation of the isolines. In summer, between 2/3 and 3/4 of the temperature amplitude is outside the built-up area. This shows seasonal separation of both main factors of the thermal differentiation: in summer the radiation balance dominates, in winter (when this amount is approximately 1/10 of the summer value), the artificial heat sources of the city (heat emitted from buildings) dominate.

Longer Growing Season in the City.

According to the climate atlas of the GDR, the mean frost-free time in the inner city of Berlin (for the period 1891–1930) was 206 days, in contrast to that in the outlying districts with means of 168 in Blankenburg, 169 in Spandau, and 184 in Dahlem on the borders of the city.

From phenological observations of trees (Günther 1959), one learns of an early start of the vegetation period in the city, which is in contrast to the situation in the surroundings (Kleinmachnow, south of Berlin), for example *Robinia pseudo-acacia* leaf shoots appeared five to six days earlier, and full flowering three to five days earlier (data for 1955–1958).

Detailed observations of blooming phases of street trees can be mapped (Zacharias 1972). In the centre of the city, the first blossoming of *Tilia euchlera* occurs eight days earlier than it does at the outskirts of the city. When the corresponding temperature differences are considered, there is a shifting of the blossoming date by about one week for each one degree Celsius. This corresponds well with what is known for larger scale phenological distributions of stronger temperature-determined phases, from for example central Europe. Trees in cold hollows, in forest clearings, and on forest edges blossom about two or more days later compared to the open sites. The strongest gradient (density of the phenological isochrones) occurs at the border between forest land and buildings, or between open sites and buildings. The relief of the land, especially in the forested areas, has less effect on the phenological data. This characteristic, is akin to that which we already know from the mean temperature data. An analysis of the phenological maps as mean temperature maps is possible.

Review of the research shows that the representation by profiles is well suited for characterisation of the city climate. The profiles are represented in Fig. 5, with the outskirts on the left, and the inner city on the right: dashed curves = temperature (distribution on summer radiation nights), continuous curves = phenological data in days (before or after representation station in the surroundings).

In considering the degree of correspondence between both curves, one should consider that for the phenological data the mean temperature of the preceding months is essential, but the temperature values depend on extreme nightly weather conditions. Thus, it is not surprising, that strong gradients do not always coincide. In both profiles, the phenological data show a smaller influence of the city than do the temperature measurements during radiation nights.

The connection of the thermic and phenological data with the different structural types of the city surface can be determined from the numbers on the lower edge of both representations (Fig. 5). The structural types are as follows: 1 open space such as fields, allotments, fallow land, green areas essentially without trees; 2 forests and similar growths; 3 forest settlements; 4–9 settlements and industrial land of increasing complexity, with density of building ranging from spread-out detached houses, to closely-packed 5 story high developments; 10–11 signify special locations (such as particular relief, or adjacent to water), which reflect the thermic and phenological characteristics.

The influence of the building construction is dominant everywhere. Furthermore, the cooling effect of even small open spaces is obvious, especially that if the Tiergarten park in the city centre. The different structures of the inner area are less clearly represented by the temperature distribution on radiation nights, than portrayed through the phenological data. In contrast, one can recognise a remote effect of neighbouring structures much better with phenological data than with the night temperatures. This addresses the expectations that the plants are also exposed to the temperatures during advection periods.

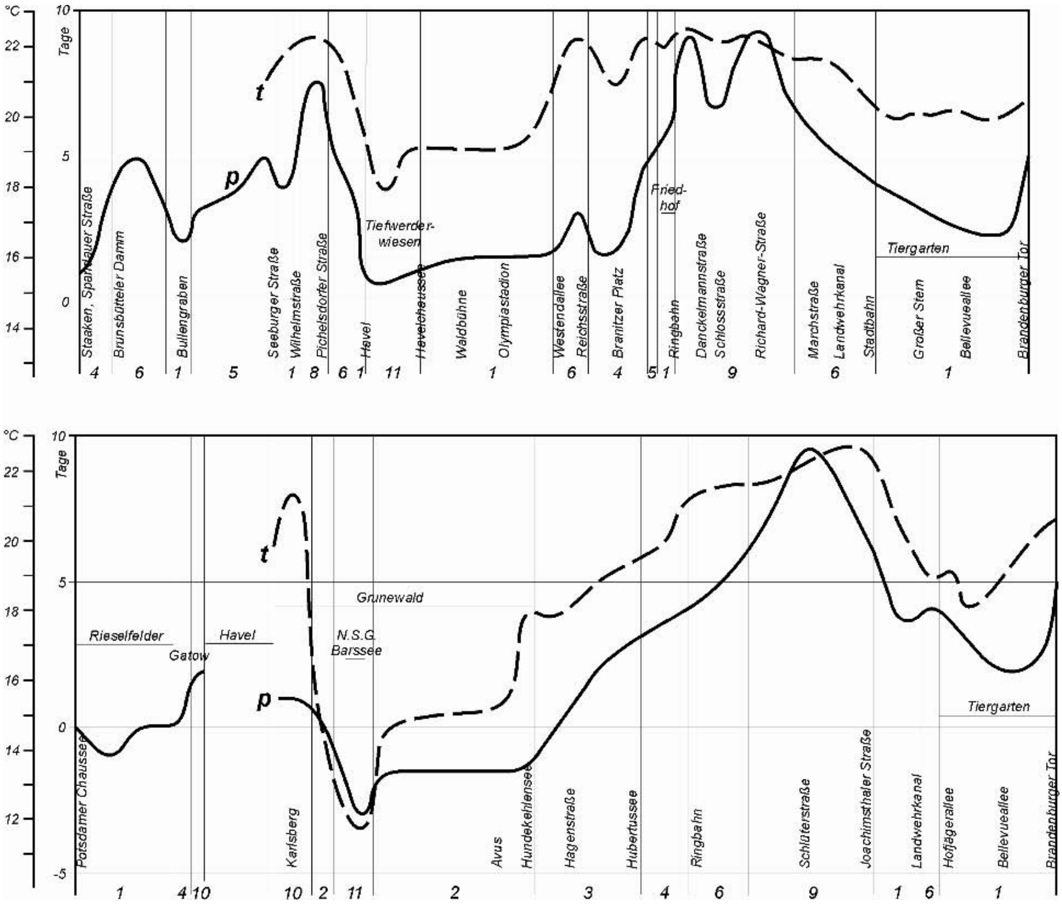

Fig. 5 Thermic and phenological profiles through western Berlin (from Zacharias 1972). Explanation in the text

These documents also facilitate a thermal estimate for the green areas. The cooling effect of large green areas also influences the surrounding built up areas. One should consider that in the means, the stronger wind periods are included. However, during these stronger wind periods, the high temperature of the city should hardly concern us. The overheating of the residential areas, especially during low wind radiation weather, may be very disturbing. This is often the case from the later afternoon into the night. The maps show that the larger green spaces (such as the park or zoo) can be about 4 °C colder at night than the surrounding area. However, only a small strip of the surrounding area is involved in the cooling down effect of the green space. The lower the wind velocity, the smaller is the effect of the green spaces on cooling their surroundings.

Relevant temperature effects of larger green spaces on the surrounding settlement can be expected only within a border of at least 100 m width. Such thermal edge effects – on the other hand – also arise from relatively small green spaces. With some caution, one can conclude that few large green spaces are not so suitable for obtaining favourable thermic effects, as are small or rather small green areas with the same surface area. The question of within which subdivision level the optimum for air purity actually lies, must still be determined. For urban planning decisions, the thermic effect of open spaces may carry relatively little weight. It is however to be noted, that the temperature distribution also permits conclusions to be drawn about the air exchange, and with it also related variables (such as degree of air pollution).

Soils

The soils of the city area are remarkably heterogeneous in composition, fine earth, humic content, and storage. There are all variations from fine earth and humus-poor rubble soils to humic garden and sandy soils with a small proportion of rubble.

From the work of Runge (1973), we can learn about wasteground soils. As in all young soils, the characteristics of the wasteground soils are strongly influenced by the materials from which they derive. The raw material is building debris, which was initially sorted by hand, and then by machine, after the World War II. During the sorting process, large brick pieces were separated from adhering mortar, and were used for reconstruction. This process still continues today "Enttrümmerung", with the debris being sorted on shaking riddles with a mesh size of less than 5 cm, and the bigger material being removed. By both hand and mechanical sorting, the material that remains is mostly brick and mortal debris, about 5 cm in diameter. This may be used for levelling at the same site, or elsewhere. There are frequently also considerable amounts of natural material mixed in with the manmade material. The places may remain unused for some considerable time, until they are newly built upon.

The levelled wastelands have subsequently been exposed to the influence of the urban environment in the 0–25 years since their initial formation, (which is in sharp contrast to the situation in a natural landscape). The environment influences the substrate, and its qualities are changed. Easily soluble materials are lost, and relatively insoluble materials become, both relatively, and absolutely, more concentrated. These processes lead to profile differentiation, and with that, to soil formation. The rapidity with which this occurs is surprising, and profile differentiation can be seen in rubble areas after 5–25 years. This is expressed in the fine earth fraction by the increase in calcium carbonate content in deeper horizons of 6 to 10%, by a pH value of 7.0 to 7.5, and a decrease of organic carbon of 3 to 0.5% (depending on the substrate). The short term or long term available nutrient content of the fine earth lies more often above, than below, the average values for natural soils. The high stone content ($\sim$ 25–50%) in the wasteground soils is more effective for the water supply than for the nutrient supply. This forms the limiting factor, because the stones not only decrease storage volume, but also effect a faster water discharge in fissures and pore volume.

Flora and Vegetation

The results of phenological research correspond to floristic-vegetation research on classification of the urban area based on building usage. The density building conspicuously forms and classifies the physiognomy of the city. Statistical and city planning data are based on this, and it appears meaningful to use this data for the basis of the classification of a city. Four zones of Berlin can be identified (see Fig. 4).

Zone 1 – Closed build-up area - Characteristic for this zone are the rental tenement buildings dating from 1914, with poorly exposed backyards, and a minimum size of $5.3\,\mathrm{m} \times 5.3\,\mathrm{m}$ according to the building regulations of 1853 (at a building height of 22 meters!). The boundary of this zone is often arbitrary, and breaks down, where the building activity was interrupted by the outbreak of the World War I. However, in reality, the actual city formed "a closed unit" within the boundary formed by the circular railway, until the destruction of World War II. Within this area, the city reached its greatest dimensions, introducing a reorganisation of development schemes.

Zone 2 – Open built-up area - Within this zone are ribbon and margin development, and open residential buildings. A further characteristic is a higher proportion of ground that is covered by vegetation (higher than in zone 1). The population density drops sharply to an average of 150–200 inhabitants/ha. In recent years, the older residential suburbs have become increasingly mixed with new residential areas, uniform new housing developments, in which there is a high proportion of lawns, giving rise to the impression of a decorated quilt.

Zone 3 – Outskirts - This region is marked by the presence of rubbish dumps (rubbish tips some 117 m high, rubble heaps and so on), and in contrast by recreational facilities (parks, allotment gardens, summerhouse colonies, and sports fields).

Zone 4 – Surrounding countryside used for farm and forestry - West-Berlin has only a small portion of this zone, namely the Grunewald, which with its inshore waters, has become the most important local recreation area for the population. The areas of arable land in the west Berlin area have been sharply reduced in recent years, as expanded housing developments encroached and took over.

Floristic Classifications

For the current determination of species composition (both qualitative and quantitative) in individual city zones, the following method has been chosen (Kunick 1973):

For sample areas of the same size (1 km2), in single zones and their transition areas, as well as for representative land uses, the entire species composition of ferns and flowering plants which were not intentionally planted at those locations, were recorded. In order to obtain more exact information about the situation for an individual species, not only presence or absence was noted, but also the frequency per 25 patches 4 acres in size (equivalent to 1 block), and a shorthand outline for the location was assigned. In addition, the occurrence of a number of chosen species was mapped regularly in a chessboard fashion for surfaces distributed regularly over the city area.

When the species richness of flowering plants is considered in relation to the use of the land, the following pertain per km^2 (based on investigation of 18 areas):

Closely built-up - 400 species
Loosely built-up - 434 species
Outskirts - 440 species
Surrounding countryside - 250 species

There is a sharp increase in species richness from the surrounding countryside to the outskirts, of about 200 species per sampling area (km^2). Towards the city centre, the species number then decreases.

Similar results were found from research on city birds and the insect fauna (Erz, Mulsow, Schweiger), where – among all studied biotope types - the highest species number and the maximum population density, occurred in the highly structured residential areas. However, in the city centre, the results diverge, with extreme poverty of species in the fauna combined with high population densities, yet this does not apply to the flora in Berlin. Nevertheless, the species number per unit area in the closely built-up zone, is still higher than in the partly natural surrounding countryside. This is explained by the high percentage of immigrant species in the city centre.

For several cities, data are available for the species number and its changes, and a high proportion of neophytes are usually emphasised. This proportion of neophytes is usually between 30 and 50%, and can be used to indicate the intensity of the industrial-metropolitan influence (Sukopp 1969). The numbers calculated for Berlin are:

Zone 1	18%
Zone 2	12–18%
Zone 3	5–12%
Zone 4	$<$ 5%

However, these numbers for Berlin need to be confirmed, and compared with other examples, before they are generalised. The rising percentage of neophytes (based on historical data), is related to differences in structure and "degree of naturalness" of the sites, and the intensity of cultural influences in the individual city zones, (as are differences in total number of species per unit area) (see Hemerobie, Table 4).

Table 4 Stages of cultural influence on ecosystems (degree of hemeroby)

	Criteria for Classification					
Hemerobiograde (terminology- after Julas)	Example	cultural influence	Substrates, soils, waters	Vegetation	Proportion of vascular plants that are neophytes	Loss of native vascular plants (per 1000 km ^{2})
metahemerobic	poisoned or biozide treated ecosystems; intact buildings and their interiors	very strong and one-sides; all organisms tend to get destroyed (whether intentionally or not)	characterized through defective or excessive organic substances, toxic substances, or extreme physical effects	only specialized species, or resistant phases; species numbers approach 0	–	–
polyhemerobic	pioneer communities with low competition, many short-lived ruderal communities	exist in short term and non-periodic formation and destruction of sites, new combination or extreme concentrations of ecological factors	greatly changed ruderal sites such as mortar soils, neopedon, substrates such as casting slag, mine heaps	greatly simplified communicty structure and destabilisation of the vegetation; extermination of less tolerant species	21 – (80)%	
euhemerobic	numerous perennial ruderal communities, field and garden weeds, lawns, forests of non-native trees	continuously strong	Changed (cultivated soils, rigosols)	vegetation and flora human conditioned (not "left to nature")	>6%	13–20%
mesohemerobic	meadows, pastures, forests of trees from different habitats, heathland, dry grassland	weaker or periodic	not totally changed or returns to the natural condition (e.g. deposits)	vegetation physiognomy human conditioned ("far from nature")	5–12 %	1–5%
oligohemerobic	forest with weak thinning or graded pasturing, salt meadows, growing dunes, growing bogs and fens,	not stronger than showing pristine features of the natural habitat	nearly no changes	actual vegetation corresponding to natural vegetation (near nature)	< 5 %	< 1 %

Table 4 (continued)

Criteria for Classification						
Hemerobiograde (terminology-after Julas)	Example	cultural influence	Substrates, soils, waters	Vegetation	Proportion of vascular plants that are neophytes	Loss of native vascular plants (per 1000 km ^{2})
	some water plant communities					
ahemerobic	water, fen and rock vegetation in some parts of Europe, in Central Europe only in parts of the alpine vegetation	does not exist		vegetation not touched by people (natural vegetation)	0 %	0 %

Classification According to the Distribution of Individual Species

Mapping of individual species in West Berlin shows that there are numerous species which have a clear concentration in the city and or in outlying districts, in contrast to other species which occur more or less everywhere with similar frequency. This may be accounted for by the fact that certain species are bound to particular land uses (such as lawns and gardens with walls), and when the land use changes, the plants no longer thrive. Alternatively, it is conceivable that, independent of the present land use, comprehensive city influences enhance or impede the distribution of higher plants, as has been demonstrated several times for lichens.

In Berlin, one can speak of a *Tussilago*-*Chenopodium botrys*-*Sisymbrium loeselii* city centre, which is subdivided into parts with the most intense war damage (characterised by intensified occurrence of *Diplotaxis tenuiflora* and *Buddleja davidii*), which are distinguished from those quarters with tenement buildings that were not seriously damaged (with *Commelina communis*, *Parietaria pennsylvanica* and wild *Ailanthus* (Contributions to ecology of *Chenopodium botrys* I–VI, Kunick 1970, Sukopp & Scholz 1964).

The open built-up zone is characterised by, for example *Atriplex oblongifolia*, *Lamium purpureum* and *Veronica* spec. div.. Within this zone, *Clematis vitalba* is associated with boulder clay, and occurs less frequently on sand. The surrounding countryside is distinguished from the other zones by the appearance of *Pinus silvestris*, *Prunus serotina* and other forest plants. For a comparison with the distribution pattern of an animal species, the collared dove can be considered: it is absent from the city centre, at a maximum at the edges of the built-up areas with street trees, common near grainfields, and uncommon near closed forest areas (Löschau & Lenz 1967).

Classification According to Biotic Communities (Thus far Only for Special Taxa)

A classification according to bird communities exists for the urban area of Hamburg, and will probably be valid for other northwest German cities (Mulsow 1968). Mulsow subdivides the house sparrow – blackbird city landscape as follows:

A. House sparrow – swift – city centre

1 Industrial – commercial zone black redstart – field landscape
2 Swift – old buildings quarters
3 Crested lark – new building quarters

B. Blackbird – greenfinch – outskirts

1 redstart – residential quarter
2 tit – dunnock – park landscape
3 lesser white throat – garden landscape

For the urban area of Berlin, a temporary classification, according to plant communities can be described:

Zone 1: *Chenopodium botrys* – community, Chenopodietum ruderale, Sisymbrietum altisimi as pioneer community, with a tendency for the development of ruderal semi-dry grassland (Poa – Tussilaginetum) and false acacia bushes (succession scheme Fig. 6).
A vegetation map of a characteristic part of the destroyed inner city of Berlin, with levelled wastelands is described by Sukopp (1971).
Zone 2: Hordeetum murini , predominant *Acer-Ulmus* young growth, with a tendency for development of a Alno-Padion forest community.
Zone 3: Urtico-Malvetum, Leonuro-Ballotetum
Zone 4: Sand – dry grassland, Pino-Querceten changed by forestry (with *Prunus serotina*).

Finally, we may inquire as to the purpose and utility of such an analysis of the biotic communities and their locations in a city, and the following observations can be made: due to the rapid increase in land development in Berlin, and the increasing urbanization of the world, the effects on organisms – plants, animals, and humans – must be observed and monitored. Most investigations about such questions usually begin only after the first bad consequences are manifest. Today, landscape conservation and urban green space management usually limp behind city development. In the present study, foundations are created to facilitate evaluation of the changes in landscape and biotic communities. If it succeeds in uniting individual research results into a comprehensive statement about the quality of life of the city biotopes, there will then be – in future – a basis for rational planning decisions; a necessity that is already urgent today.

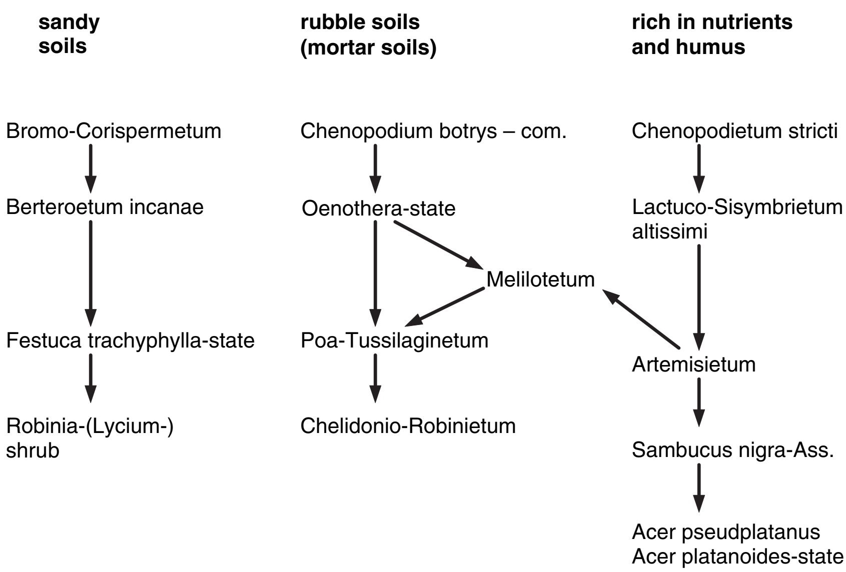

Fig. 6 Vegetation development at locations in the inner city of Berlin

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Ecosystem Processes Along an Urban-to-Rural Gradient

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Abstract In order to understand the effect of urban development on the functioning of forest ecosystems, during the past decade we have been studying red oak stands located on similar soil along an urban-rural gradient running from New York City ro rural Litchfield County, Connecticut. This paper summarizes the results of this work. Field measurements, controlled laboratory experiments, and reciprocal transplants documented soil pollution, soil hydrophobicity, litter decomposition rates, total soil carbon, potential nitrogen mineralization, nitrification, fungal biomass, and earthworm populations in forests along the $140 \times 20\,\mathrm{km}$ study transect. The results revealed a complex urban-rural environmental gradient. The urban forests exhibit unique ecosystem structure and function in relation to the suburban and rural forest stands; these are likely linked to stresses of the urban environment such as air pollution, which has also resulted in elevated levels of heavy metals in the soil, the positive effects of the heat island phenomenon, and the presence of earthworms. The data suggest a working model to guide mechanistic work on the ecology of forests along urban-to-rural gradients, and for comparison of different metropolitan areas.

Keywords: urban · rural · forests · gradients · ecosystems

Introduction

The conversion of natural or agricultural landscapes throughout the world to highly modified urban landscapes is expected to continue and many urbanized areas are expected to become even more highly modified (Alig and Healy, 1987; Richards, 1990; Douglas, 1994). The need for comprehensive studies of the ecological impacts of urbanization is great (Brown and Roughgarden, 1989; Rogers, 1994; Penner, 1994). In 1989, 74% of the U.S. population (203 million people) resided in urban areas and that number is expected to increase to more than 80% by the year 2025 (Fox, 1987; Alig and Healy, 1987; Haub and Kent, 1989). Between 1960 and 1980, urban land in the United States increased by 22 million acres (Frey, 1984) resulting in the conversion of cropland, pastures, and forests into urban and suburban environments.

Ecologists in North America, however, have historically been reluctant to study ecological systems in urban environments because they are perceived as ‘unnatural’ and contain such problems unknown disturbance histories, multiple stresses, and lack of dedicated research sites (Cairns, 1987, 1988; Ludwig, 1989; McDonnell and Pickett, 1990; Botkin, 1990). Thus, at a time

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when legislators, managers, scientists modeling global change, and the general public critically need ecological information from urban areas there is relatively little available.

Because ecologists have not historically worked in areas populated by humans, important terminology and concepts related to urban environments have been developed by geographers, social scientists, anthropologists, and economists and may not always be ecologically operative. Therefore, commonly used terms as 'urban' and 'rural' have multiple meanings and relate to a variety of conditions such as land cover, population density, the amount of impermeable surfaces, and cultural practices. For the purpose of this paper, we use the geographer's definition of urban based on the number of humans per hectare (ha). urban areas are defined as having more than 6.2 people per ha whereas rural areas have 1 to <1 person per ha (U.S. Bureau of Census, 1980; Bourne and Simmons, 1982). Areas described as suburban typically have human population densities between the urban and rural levels. The high density of humans in urban areas typically results in large-scale modification of the environment and a tremendous concentration of food, water, energy, materials, sewage, pollution, and garbage which we collectively categorize as urban land use (Godron and Forman, 1983; Roodman, 1996). At the other end of the land-use spectrum are rural environments that are sparsely populated with humans and consequently exhibit less built-on land and lower concentrations of energy, materials, water, and waste products (Godron and Forman, 1983).

The conversion of rural land use to urban land use is referred to as urbanization and occurs over time. This paper does not address the ecological consequences of urbanization over time, but instead focuses on the effects of urbanization in a spatial context. We are specifically interested in quantifying changes in the structure and function of forests embedded in a range of land-use types from existing urban to rural landscapes. Our measure of urbanization in space is the distance from the urban core which is supported by our land use analysis of the study area. The forests closest to the New York City end of the transect are considered to be more urbanized than those at the rural end.

This paper is a synthesis of more than a decade of research on forest ecosystems along an urban-rural land use gradient running from densely populated New York City north to rural Litchfield County, Connecticut. We feel that the use of standard gradient analysis techniques provides a new approach for addressing both basic ecological questions and practical environmental problems facing urban areas (McDonnell *et al.*, 1993). By quantifying changes in ecosystem structure and function in relationship to varying levels of urbanization, we can obtain a greater understanding of the nature of urban impacts on natural ecosystems which will assist us in developing important new research questions and management strategies.

Ecologists have effectively studied natural gradients of soil moisture, elevation, and salinity to understand the relationship between environmental variation and the structure and function of ecological systems, including populations, communities, and ecosystems (Whittaker, 1967; Siccama, 1974; Pickett and Bazzaz, 1976; Peet and Loucks, 1977; Austin, 1987; Keddy, 1989; Vitousek and Matson, 1990). The gradient paradigm also applies to urban environments (McDonnell and Pickett, 1990; McDonnell *et al.*, *1993*). Urban areas in the United States typically have a highly modified and densely populated urban core surrounded by asymmetric rings of diminishing landscape modification (Dickinson, 1966; Forman and Godron, 1986; Berry, 1990). The resulting array of natural and human-modified ecosystems within a metropolitan area can be conceived of as a readily measurable gradient of land use and a more complex gradient of urban effects (McDonnell and Pickett, 1990; McDonnell *et al.*, 1993; Medley *et al.*, 1995). Thus, the gradient paradigm is a useful organizing tool for research on the ecological consequences of urbanization. Since the concept of urban-rural gradients was introduced (McDonnell and Pickett, 1990) it has been effectively used to study a variety of ecological issues in urbanized areas throughout the world, including avian diversity (Blair, 1996) and richness (Jokimaki and Suhonen, 1993), air pollution effects on moss-dwelling animals (Steiner, 1994), heavy metal accumulation by mosses (Gupta, 1995), and lepidoptera population diversity and variability (Wolda *et al.*, 1994).

A Model for Assessing Anthropogenic Causes and Ecological Effects

The structure of metropolitan areas and their fringes consists of a variety of components, ranging from totally built environments to 'natural' or seminatural areas (Stearns and Montag, 1974). Ecological studies of urban areas can focus on several scales including the metropolitan area as a whole, the city core, or a natural area within a city. Natural areas are defined as ecosystems which persist primarily because of natural processes of plant establishment, water availability, nutrient cycling, and plant-animal interactions with minimal human manipulation (e.g., wooded natural areas in New York City parks, lakes, ponds, streams, etc.) (McDonnell, 1988). To facilitate the study of the ecology of urbanization, the individual components (e.g., structures, physical and chemical environments, populations, communities, and ecosystems) must be quantified, and correlations among them assessed. By doing so, the ecologically important impacts of urban development and change on natural areas can be determined. McDonnell and Pickett (1990) proposed a model of the effects of urbanization on ecological phenomena that includes: a) aspects of urbanization, b) biotic and environmental effects of urbanization, and c) ecosystem effects (Fig. 1). We will use this model to illustrate the utility of the gradient paradigm in studying the structure and function of forests (i.e., natural areas) along urban-rural gradients.

The Study Site

As previously stated, we have focused our research on urban-to-rural gradients on a 140-km transect running from highly urbanized Bronx County, New York, to rural Litchfield County, Connecticut. (McDonnell *et al.*, 1993) (Fig. 2). Manhattan and the study area to the north constitute

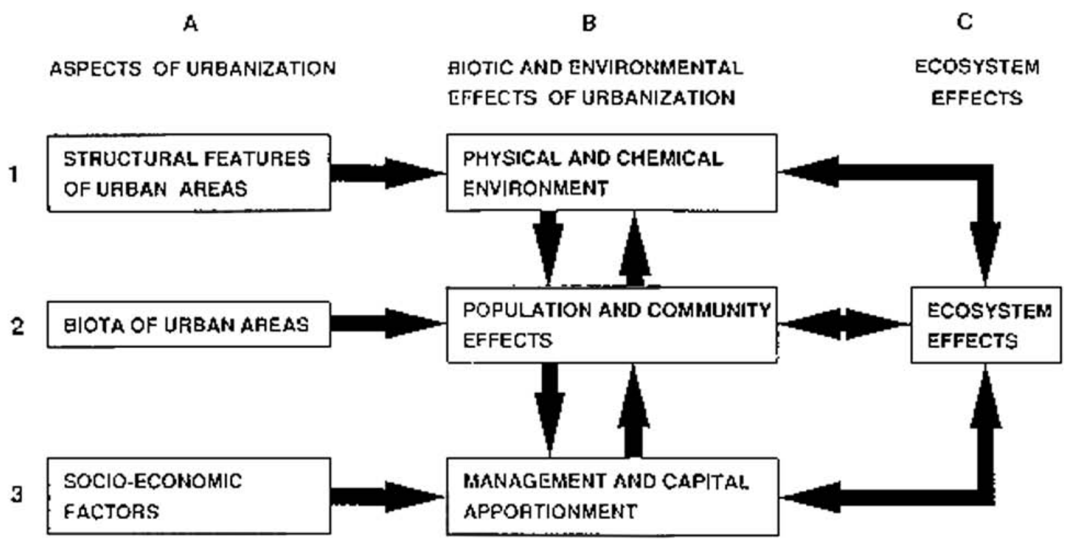

Fig. 1 A composite model of the effects of urbanization on ecological phenomena. The three columns represent relevant components of urban-rural gradients. The arrows indicate causal linkages between the features of urban areas (column A) as inputs, and the ecological phenomena (columns B, C) as results. Ecological research is focused primarily on the phenomena represented by rows 1 and 2, whereas the results would be helpful in decisions concerning societal phenomena represented by row 3. Feedbacks from columns B and C to A would be useful in developing strategies to reduce the environmental impact of urbanization. (Modified from McDonnell and Pickett, 1990)

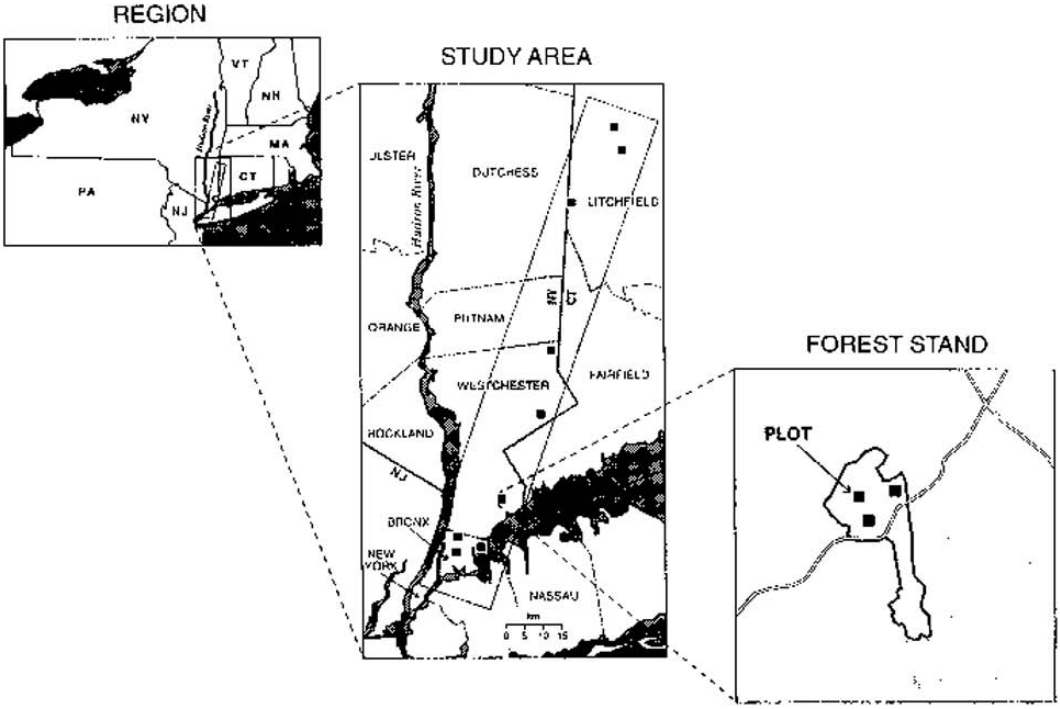

Fig. 2 A map of the New York City Metropolitan Area showing the location of the 20 km wide by 140 km long transect used to study the structure and function of forests along the urban-rural gradient. Ecological studies can focus on a variety of scales ranging from the metropolitan region, landscape unit to forest site. (Modified from McDonnell and Pickett, 1990)

the southern portion of the Northeastern Upland Province (Broughton *et al.*, 1966). The study transect was centered on that part of the landscape with bedrock consisting of metamorphosed and dissected crystalline rocks which were separated into various formations classified by their composition of schist, granite, and gneiss (Schuberth, 1968). Upland soils in the study area are classified as Typic or Lithic Dystrochrepts, loamy, mixed, mesic subgroups (Gonick *et al.*, 1970; Hill *et al.*, 1980).

To control for potentially confounding factors, such as differences in parent material, soil moisture regimes, stand age, species composition, and soil texture, only forests exhibiting the following conditions were considered for study: 1) similar topography and soil type to at least the U.S. Department of Agriculture (USDA) subgroup classification category (Soil Survey Staff, 1975); 2) oak dominated community (including at least one of the following species *Quercus rubra, Quercus veluiina, Quercus alba*, and *Quercus coccinea*) with similar species composition across sites; 3) minimum stand age of 60 years; 4) a closed canopy; and 5) no evidence of recent severe anthropogenic disturbances such as soil excavation or tree cutting.

Nine forest study sites were established along the transect (Fig. 2). They were stratified to have three forests representing each of the urban, suburban, and rural regions of the gradient. Within each forest study site three $20\,\mathrm{m} \times 20\,\mathrm{m}$ plots were randomly established at least 50 m from the edge of the forest patch giving a total of 27 plots along the transect (Fig. 2).

Aspects of Urbanization

Landscape structure and human population densities

We have coarse-resolution data to characterize the overall patterns of change along the $20 \times 140\,\mathrm{km}$ transect. Population characteristics, summarized by county subdivisions, have been plotted along a line positioned in the middle of the urban-rural belt transect. State and interstate highways, with traffic counts averaged for major road sections, were mapped at 1:250,000. From this map we computed a mean of annual daily traffic averages and a measure of connectivity (β = no. road segments/no intersections; see Haggett *et al.*, 1977) for the highways within $20 \times 5\,\mathrm{km}$ sections along the transect. The U.S. Geological Survey (USGS) land use-land cover digital data were imported into IDRISI (a grid-based geographic analysis system produced by the Clark University Graduate School of Geography) and rasterized into a grid-cell map. The area of the grid cells, 17.6 ha, closely approximates the original scale of generalization used in the original classification of rural localities ($\sim 16\,\mathrm{ha}$; see U.S. Geological Survey, 1986). Land use and landscape fragmentation were compiled for ten $\sim 77\,\mathrm{km}^2$ land units randomly selected from the rasterized map. We used multiple sampling techniques to determine a most effective approach, considering the variety of spatial data applicable to the project (Medley *et al.*, 1995).

Preliminary analyses document a complex urban-to-rural gradient. Human population density, traffic volume, and the percentage of built-up land (i.e., urban and residential) decline, whereas the percentage of forest land and the mean size of forest patches increase, in a linear or logarithmic trend away from the urban New York City core (Medley *et al.*, 1995). A sharp transition from an urban matrix to a forest matrix occurs along the urban-rural transect in this agriculturally unproductive region. Conversely, 1960–1990 population growth, highway connectivity, and the structural heterogeneity of the landscape, show quadratic relationships to urban-rural distance (Medley *et al.*, (1995). These findings suggest that urban exposure parallels the transect, but the changes or disturbances associated with urbanization show a complex spatial pattern not clearly related to urban-rural distance alone (Medley *et al.*, 1995).

Biotic and Environmental Effects of Urbanization

Physical and chemical environment

The climate of the region is characterized by warm, humid summers and cold winters with average annual air temperatures ranging from $12.5^{\circ}\mathrm{C}$ in New York City to $8.5^{\circ}\mathrm{C}$ in northwestern Connecticut (National Oceanic and Atmospheric Administration [NOAA], 1985). A close examination of the temperatures recorded from 1985 to 1991 at NOAA/National Weather Service sites located at 0, 25, 57, 112, and 140 km along the transect revealed that the mean monthly temperatures at the 0-km site, which is the urban core, are typically 2–$3^{\circ}\mathrm{C}$ warmer than any of the other locations (McDonnell *et al.*, 1993). This reflects, in part, the well documented heat island phenomenon occurring in urban areas throughout the world (Bornstein, 1968; Berry, 1990).

Precipitation ranges from an average annual total of 108 cm in New York City to 103 cm in north-western Connecticut (NOAA, 1985). Precipitation is evenly distributed throughout the year (McDonnell *et al.*, 1993). Air quality is poor at the urban end of the gradient and improves in rural areas as illustrated by the decline in particulate sulfate and total particulate levels with increasing distance from the urban core (New York State Department of Environmental Conservation, 1989; Gradedel and Crutzen 1989).

Although many features of the forest soils along this urban-rural gradient are similar, we discovered elevated levels of lead, copper, and nickel in forests at the urban end of the gradient, declining to background levels in rural sites (Pouyat and McDonnell, 1991; Pouyat *et al.*, 1995). We also found soil hydrophobicity (i.e., water repellency) as measured by contact angle of a water droplet to be highest in forests at the urban end of the gradient (White and McDonnell, unpubl. data). Finally, analysis of the forest floor leaf litter along the transect indicated that mean depth, mass, and density of the leaf litter layer increased with increasing distance from the urban core (Kostel-Hughes, 1995; Kostel-Hughes *et al.*, 1996).

Population and Community Attributes

Forests in the urban end of the transect have lower stem densities, depauperate understories, and contain an increasing proportion of non-native species in the sapling and seedling size classes than similar forests in rural areas (Rudnicky and McDonnell, 1989; McDonnell *et al.*, 1990; McDonnell and Roy, 1996; McDonnell, unpubl. data). Studies of the soil seed bank in urban forests also reveal the presence of non-native woody plant species, but at relatively low densities (Kostel-Hughes, 1995).

Pouyat *et al.* (1994a) found that soil microinvertebrates, of which they included the taxonomic groups Mesostigmata, Orbatida, Collembola, and other microinsects, abundances were higher in the rural forests than in either the urban or suburban sites during the fall season, but exhibited no significant differences during the spring. Red oak litter placed on the forest floor of the study sites along the transect by Pouyat *et al.* (1994a) also exhibited a measurable difference in fungal activity. Rural forests had the highest total fungal hyphal length (i.e., abundance) after 36 weeks of exposure as compared with the suburban and urban sites. Between leaf drop and mid-winter, fungi appeared to have grown more rapidly on litter in the rural sites than either the suburban or urban forests. Both fungivous microinvertebrates and litter fungi were inversely correlated with soil heavy metal concentrations (Pouyat *et al.*, 1994a).

An assessment of earthworm populations in forests along the transects by Steinberg *et al.* (1996, in press) revealed high numbers and biomass of earthworms in the urban forests with relatively few in forests at the rural end of the transect. Steinberg *et al.* (in press) report that urban forests had 25.1 worms m^{-2} and 2.16 g of worms m^{-2} whereas the rural forests had only 2.1 worms m^{-2} and 0.05 g of worms m^{-2}. The reduced leaf litter depth, mass and density of the forest floor (O_2 horizon) (Kostel-Hughes, 1995), and increased organic matter levels at depth (Pouyat *et al.*, 1995) provide further support for increased earthworm activity in the urban forests along the study transect.

These studies are not suggesting that rural forests with earthworms do not exist along the transect or that every urban forest has abundant earthworm populations. However, the forests on our study transect are in previously glaciated areas. No native earthworm species occur in these areas, and the prevalence of non-native species may depend largely on the degree of human activity (Sam James, Maharishi International University, pers. comm.; Steinberg *et al.*, 1996, in press). Therefore, our observations of low earthworm densities in the rural sites are not unreasonable, and for our gradient it appears that the absence of earthworms in the rural sites and high densities in the urban sites is a real dichotomy that can explain, in part, some of the differences in nutrient cycling parameters described below.

Ecosystem Effects

We chose to focus this area of research on two key components of forest ecosystems that are likely to vary in response to the complex gradients that extend from a dense urban core to the surrounding countryside: plant litter dynamics and nitrogen cycling. Foliar litter decomposition is an ideal feature of ecosystems to examine because it integrates many features of the abiotic and biotic environments. Litter decomposition affects plant community regeneration (Facelli and Pickett, 1990), is a critical bottleneck in determining ecosystem nutrient flow and therefore availability of resources for higher plants in a community (Coleman, 1986; Monk and Day, 1988), is an important site of heavy metals incorporation into ecosystems (Van Hook and Shults, 1977; Tyler, 1978), and provides both a habitat and resource for fungi, bacteria, and invertebrates (Choudhury, 1988; Seastedt and Crossley, 1983). Litter decomposition, consequently integrates the effects of resource quality, environmental factors and activities of decomposer organisms on nutrient cycling, thereby serving as an easily measured indicator of the impact of urbanization on an important ecosystem function.

Forest nitrogen dynamics have been studied extensively in the Northeastern United States and there is a large body of literature that suggests that increased anthropogenic inputs of nitrogen in urban environments should have complex non-linear effects on plants, microbes, and soil chemistry (Friedland *et al.*, 1984; Nihlgard, 1985; Agren and Bosatta, 1988; Aber *et al.*, 1989). Aber *et al.* (1989) made a number of specific predictions of ecosystem-level responses to constant, elevated nitrogen inputs continued over many years. They emphasized the pivotal nature of the onset of significant nitrification, when more nitrogen is mineralized than can be taken up by plants and microbes. Nitrification could precipitate decreases in fine root biomass and increases in NO_3 leaching below the rooting zone (Aber *et al.*, 1989). McNulty *et al.* (1990) documented a pattern of increased nitrification (when expressed as a fraction of nitrogen mineralization) across a transect from southern Maine to northern New York, a gradient characterized by an almost twofold increase in nitrogen wet deposition. McColl and Bush (1978) have shown increased nitrification and NO_3 leaching in forests attributable to nitrogen deposition from the urban area downwind of San Francisco and Johnson *et al.* (1991) have reported the same phenomena in high-elevation spruce forests of North Carolina, which receive elevated levels of nitrogen deposition from rain, dry deposition, and cloud water.

Our initial hypothesis concerning decomposition and nitrogen-mineralization rates along the transect was based primarily on the soil characteristics showing high levels of heavy metals in urban forest soils and the biotic measurements which indicated reduced soil fungi and microinvertebrates in the forests at the urban end of the transect. This would be consistent with the widely held view that forests in urban environments are under stress and should exhibit reduced ecosystem function (Goudie, 1990). Thus, our initial hypothesis of ecosystem processes along our transect predicted that decomposition rates, nitrogen-mineralization rates, and nitrification rates would be lower in forests in the urban environment.

Litter decomposition and carbon dynamics

Litter decomposition rates in forests along the transect were determined using standard litter bag techniques (Bocock, 1964; Freedman and Hutchinson, 1979). The initial experiment used a reference litter of sugar maple (*Acer saccharum*) collected from a rural site and, contrary to expectations, revealed that decomposition rates were higher in forests at the urban end of the transect (Pouyat, 1992; McDonnell *et al.*, 1993). To test whether decomposition rates were controlled by site environment or litter quality, reciprocal red oak (*Q. rubra*) litter transplants were performed between urban and rural sites on the transect. The results of this study indicated that both urban and rural litter decomposed faster in urban vs. rural sites (Pouyat *et al.*, 1996, in press). The rural litter, however, consistently decomposed faster than the urban litter in all sites. This suggests that the rural litter is of higher quality than the urban litter.

The differences in decomposition rates and litter quality between the urban and rural forest stands just described, coupled with the variations in the physical, chemical, and biotic environments along the study transect presented in previous sections, suggested that the flow of carbon between plants and microbial communities may be quite different in urban vs. rural forests. Groffman *et al.* (1995) examined the carbon dynamics of forests along the study transect in detail, separating the soil carbon into four pools: 1) readily mineralizable carbon with a turnover time of days to weeks; 2) labile carbon with a turnover time of weeks to months; 3) potentially mineralizable carbon with a turnover time of months to years; and 4) passive carbon which is very recalcitrant with a turnover time of years to decades to centuries. They discovered that the urban forests had lower labile carbon and higher total passive carbon than the rural forests (Groffman *et al.*, 1995). Rural forests also had higher pools of both readily mineralizable and potentially mineralizable carbon (Groffman *et al.*, 1995). Groffman *et al.* proposed that high total passive carbon in the urban forest soils are most likely due to three factors: 1) high decomposition rates in urban forest which would deplete the labile carbon pool; 2) air pollution, especially ozone damage, which may lead to the creation of more passive carbon; and 3) the presence of earthworms, which would expedite the retention of organic matter in soil aggregates. They concluded from this study that, over a long period of time, urban forests along the transect have a potential for sequestering and storing more carbon than the rural forests, but the lower levels of labile carbon in the urban stands could result in lower rates of microbial activity in the urban forest soils.

Nitrogen dynamics

Several studies have been conducted to determine the nature of the nitrogen dynamics of forests along the transect (White and McDonnell, 1988; Pouyat, 1992; Pouyat *et al.*, 1994b; Pouyat *et al.*, 1996, in press; Goldman *et al.*, 1995; Steinberg *et al.*, 1996, in press). Measurements of nitrogen-mineralization and nitrification rates were made both in the laboratory and *in situ* in buried bags during the growing season (April–November). The laboratory incubations allowed us to control moisture and temperature which we knew varied along the transect. During the past 5 years nitrogen-mineralization and nitrification rates have been obtained for the A horizon, and for the forest floor and mineral soil combined in forests along the transect (O plus A Horizons).

Net nitrogen-mineralization and nitrification rates of the A horizon, as measured in the laboratory incubations (i.e., net potential nitrogen mineralization) and *in situ* in buried bags, were highest in the urban forests in comparison to the rural forests (Pouyat, 1992; Pouyat *et al.*, 1994b; Pouyat *et al.*, 1996, in press). In addition, nitrification accounted for close to 50% of the total nitrogen-mineralization measured in urban forest soils as compared to only 20% for the rural soils (Pouyat, 1992).

In subsequent studies of mineralization and nitrification rates of forest soils along the transect the humus layer was included in the soil samples (O plus A Horizon). These studies revealed results contradictory from those mentioned above. Net potential mineralization rates were higher in the rural forest stands as opposed to the urban stands (Goldman *et al.*, 1995; Pouyat *et al.*, unpubl. data; Groffman *et al.*, unpubl. data). Nitrification rates in urban forest soils, on the other hand, were still higher than in the rural forest soils (Goldman *et al.*, 1995, Pouyat *et al.*, unpubl. data, Groffman *et al.*, unpubl. data). These high nitrification rates that occur in urban soils even when nitrogen-mineralization rates are low, appear to be the result of a history of earthworm activity (Bohlen *et al.*, 1996; Pouyat *et al.*, 1996). The discrepancies between nitrogen-mineralization and nitrification rates from A horizon samples vs. O plus A horizon samples in urban and rural forest soil samples can be explained simply by the amount of organic matter available for microbial degradation. Urban forest A horizon samples have higher organic matter mixed throughout the sample due to the presence of large numbers of earthworms in comparison to the rural forest soils (Steinberg *et al.*, 1996, in press). Those soil samples with higher organic matter (i.e., urban forest soils) would exhibit higher nitrogen-mineralization and nitrification rates due to the increased availability of nitrogen and carbon. Following this reasoning, soil samples that contain both the O and A horizons would have higher nitrogen-mineralization and nitrification rates than samples that contain only the A horizon. Because the rural forests have a more developed humus layer as compared to the urban forests, we would expect that nitrogen-mineralization and nitrification rates would be higher in the rural forest stands (Pouyat *et al.*, 1995).

The low available nitrogen, low labile carbon pools, and high passive carbon pools in the urban forest soils (O plus A horizon) in relation to the rural forest appear to also have an effect on trace gas fluxes. Laboratory and field measurements of methane (CH_4) consumption rates in urban, suburban and rural forest soils along the study transect indicated that urban forest soils consumed 30% less CH_4 than either the suburban or rural stands (Goldman *et al.*, 1995). The authors suggest that the low amount of labile carbon in the urban forest soils adversely influences the soil microbial community that is the primary cause of CH_4 consumption in forest soils.

The Structure and Function of Oak Forest Ecosystems Along an Urban-rural Gradient

The application of the urban-rural gradient concept to determine the influence of urbanization on the structure and function of oak forest ecosystems has been successful in obtaining a new understanding of the effects of urbanization on ecological phenomena. Using our initial model of anthropogenic causes and ecological effects along urban-rural gradients (Fig. 1), especially levels 1 and 2, we obtained somewhat unexpected results that revealed unique interactions between the three major components of the model: 'aspects of urbanization,' 'biotic and environmental effects of urbanization,' and 'ecosystem effects' (Fig. 3). The structural features of the landscape (e.g., land-use patterns, roads, etc.), varied significantly along the study transect with the urban end having more built structures and surfaces such as roads in comparison to rural Litchfield County. There were also measurable differences in the physical and chemical environment along the study transect with the urban end exhibiting high levels of air pollution and soil heavy metals as well as increased temperatures in comparison with the rural end of the transect. These results confirm that there is an underlying complex environmental gradient that runs from the New York City urban core to rural Litchfield County, Connecticut (McDonnell *et al.*, 1993).

The literature on the detrimental effects of poor air quality (e.g., high levels of SO_2, sulfate, ozone, etc.), elevated levels of soil and forest floor heavy metals, and low water availability such as those caused by hydrophobic soils, support the commonly held ecological belief that forest ecosystems in urban environments would have low species diversity, low productivity, slow decomposition rates and low nitrogen-mineralization rates (Tyler, 1975; Lepp, 1981; Smith, 1990; Treshow, 1984; Hutton, 1984; Goudie, 1990; Findlay and Jones, 1990).

In our study area, the heavy metal levels in urban forest stands approach or exceed those levels found to affect soil invertebrates (Bengtsson and Tranvik, 1989), macrofungi (Freedman, 1989), and soil microbial processes (Baath, 1989). Our studies of the fungal and microbial communities in oak forests along the study transect found that the urban forests exhibited reduced fungal biomass and microarthropod densities in relation to the rural stands. These results support the commonly held belief that urban forests have depauperate communities because of anthropogenic stress (Goudie, 1990).

Unlike the population and community effects, the ecosystem effects along the urban-rural gradient proved to be more complex and contrary to commonly held ecological beliefs. Litter decomposition and nitrification rates were higher in urban forests than in rural forests even though they had poorer leaf litter quality. The lower quality litter, in part, explains the lower nitrogen-mineralization rates, lower methane consumption, lower labile soil carbon pools, and higher total passive soil

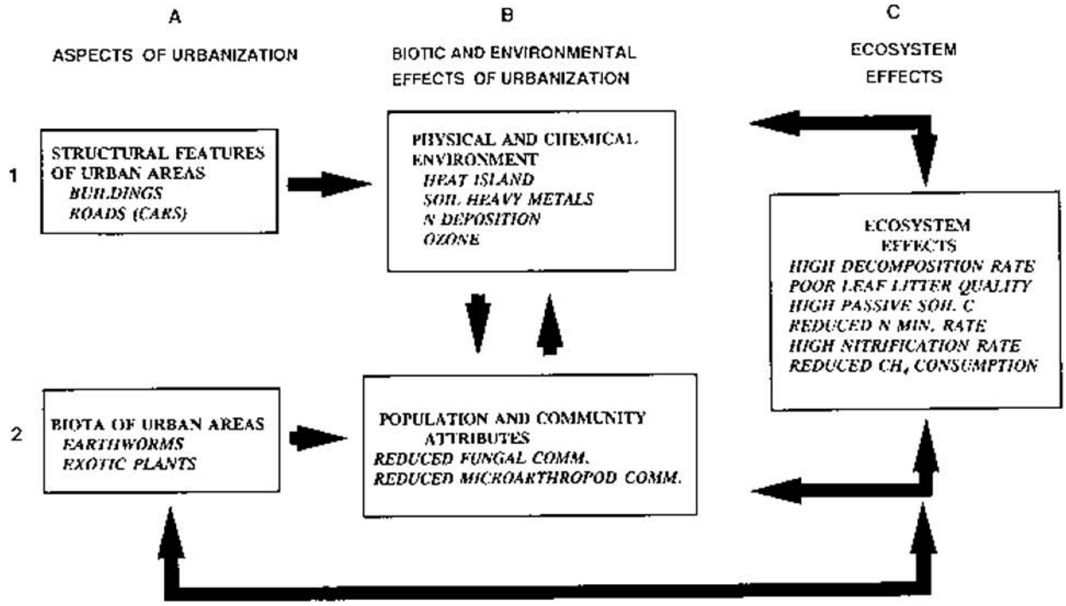

Fig. 3 The composite model of the effects of urbanization in ecological phenomena listing critical factors for each component determined in the study of a 140×20 km urban-rural gradient running from the New York City urban core to rural Litchfield County, Connecticut. Note the addition of the strong interaction between the biota of urban areas and ecosystem effects components of the model due to the activities of earthworms in the urban forests

carbon pools in urban forests in relation to the rural forests. The higher average temperature at the urban end of the transect may, in part, explain the increased rates in ecosystem processes in the urban forests. The key to understanding nutrient cycling in the urban forests proved to be the large densities and biomass of non-native earthworms at the urban end of the gradient which falls in the 'aspects of urbanization' component of our model (Fig. 3).

It has long been recognized that earthworms can greatly increase the loss of surface litter (Raw, 1962; Lee, 1985; Lavelle, 1988). Satchell (1976) concluded that the amount of organic matter consumed by earthworms was limited by the amount available rather than by their capacity to ingest it. In no-tillage agroecosystems in Georgia, Parmelee *et al.* (1990) reported that earthworms increased organic matter loss by 45%, and on a rich mull site in Sweden, earthworms were associated with higher decomposition rates of beech litter compared with less fertile soils (Staaf, 1987).

Earthworms can also directly affect available nitrogen and nitrogen-mineralization rates. Direct nitrogen flux, as estimated by secondary production, can be considerable. For example, Parmelee and Crossley (1988) calculated an annual nitrogen flux of 63 kg/ha in a no-tillage agroecosystem. Earthworms ingest organic material of relatively wide C:N ratios and convert it to earthworm tissue of lower C:N ratio (Syers and Springett, 1984). Nitrogen is then returned to the soil in dead tissue, urine, and mucoproteins. Satchell (1967) estimated that a minimum of 70% of the nitrogen in dead earthworm tissue was mineralized in 10–20 days. Annual urine excretion for lumbricoid populations has been estimated to be 18.5 kg/ha (Lee, 1982) and similar inputs have been estimated for mucoproteins (Dash and Patra, 1979). Whether or not earthworms increase system loss of nitrogen is not clear. Earthworms increased the amount of ammonia and nitrate in leachate from forest litter (Anderson and Ineson, 1984) and the percentage of inorganic nitrogen in leachate from grassland soils (James and Seastedt, 1986).

In addition to direct effects of earthworms on decomposition and nitrogen-mineralization, earthworms may also affect these processes by altering the structure and function of the decomposer food web. This research is largely based on studies of earthworm casts. Casts generally have higher moisture, carbon and nitrogen contents, and often a higher C:N ratio than surrounding soil, and thus provide a favorable habitat for increased microbial activity (Parle, 1963; Lee, 1985; Shaw and Pawluk, 1986; Scheu, 1987). Classic mull vs. mor comparisons have found mull soils to be dominated by earthworm activity, lower fungal and higher bacterial densities, and lower microfaunal populations. Macroinvertebrates may also shift the bacterial:fungal ratio to favor bacteria (Anderson, 1988). It has been hypothesized that shifting to a bacterially based food web can lead to greater decomposition and nitrogen-mineralization rates (Hendrix *et al.*, 1986). Earthworms have also been implicated in altering the structure of collembolan communities (Marinissen and Bok, 1988) and decreasing nematode populations (Yeates, 1981). These observations are consistent with the lower fungal hyphae and nematode and microarthropod populations we have observed in our urban forests.

Conclusion

Our studies suggest a potential cause and effect relationship between the physical and chemical environment along the gradient and changes in oak forest community structure and ecosystem processes. Forests at the urban end of the gradient exhibit reduced fungal and microarthropod populations and poorer leaf litter quality than the more rural forests. But, the potential negative effect of these conditions on ecosystem processes such as decomposition and nitrogen-cycling in the urban forests appears to be ameliorated by two other anthropogenic causes: increased average temperatures caused by the heat island effect and the introduction and successful colonization of earthworms in the urban forests. Forests at the urban end of the transect, in fact, exhibit faster litter decomposition and nitrification rates than the rural forests. Our current and planned research has focused on experimental studies designed to test the hypotheses generated by the results of our work. All future studies of urban-rural gradients should be coupled whenever possible with experimental studies to test the hypotheses suggested by gradient analyses.

The documentation of environmental changes and the community and ecosystem response of forests along an urban-rural gradient provide a new template for ecological research. North American ecologists have only rarely studied the structure and function of populations, communities, and ecosystems under varying and known degrees of human population densities and their associated effects. The study of ecological systems along urban-rural gradients could readily expand to discover new patterns and interactions at genetic, physiological, population, community, ecosystem, and landscape levels. Furthermore, the linkages among interactions at various levels might well be sensitive to position along the gradient. Because the combination of stresses and disturbances is new and often extreme at certain ranges of the gradient, the potential to exploit the urban rural gradient as an unplanned, but profound experimental treatment is great. It provides a new, but widespread substrate for ecological study, and a potential interface for ecology with the study of human societies. In the face of global environmental change, it is critical to understand such an interface in order to develop predictions about how these important areas may change in the future.

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House Sparrows: Rapid Evolution of Races in North America

Richard F. Johnston, Robert K. Selander

Abstract Conspicuous adaptive differentiation in color and size has occurred in the house sparrow *(Passer domesticus)* in North America and the Hawaiian Islands since its introduction in the middle of the 19th century. Patterns of geographic variation in North America parallel those shown by native polytypic species, in conformity with Gloger's and Bergmann's ecogeographic rules. Racial differentiation of house sparrow populations may require no more than 50 years.

Keywords: House Sparrow · rapid evolution · body size · coloration · geographic variation · morphology

A number of workers have attempted to demonstrate evolutionary changes in the house (English) sparrow (*Passer domesticus*) in North America since its introduction from England and Germany in 1852 [1]. Several early studies based on small samples of specimens produced results [2] which were negative or statistically unreliable. In an investigation which has been widely cited [3] as evidence for slow rates of evolution of avian races, Lack [4] found no unequivocal evidence of divergence in bill and wing dimensions of the North American and Hawaiian populations from the Old World stock. However, Calhoun [5], using larger samples and employing refined methods of analysis, was able to show that average wing length in populations of the eastern and central United States increased slightly more than 1 mm between the time of introduction and 1930. He also demonstrated geographic variation in average length of wing, femur, and humerus correlated with regional differences in duration and severity of freezing temperatures in the United States. Recently, the possibility that New World populations exhibit regional color differences has been suggested by Keve [6].

To assess the full extent of variation in characters of color and size, series of 100 to 250 specimens of house sparrows in fresh plumage were taken by us in October and November, 1962 and 1963, at various localities in North America and in the Hawaiian Islands, Bermuda, England, and Germany (Fig. 1). This extensive material clearly demonstrates the existence of pervasive geographic variation in a large number of characters in the North American and Hawaiian populations. Each New World population sampled has differentiated to greater or lesser degree from any other and from the Old World stock. Preparation and analysis of this material is still in process, but the preliminary findings presented here provide a general indication of the surprising extent to which selection has produced morphologic differentiation in a small number of generations. We have not as yet undertaken studies of the developmental basis of the morphologic characters of the house sparrow populations, but, in view of extensive evidence for comparable racial characters in other species of birds and in mammals [7], we are safe in assuming that the geographically variable characters of color, pattern, size, and body proportions are in fact genetically controlled and are either directly adaptive in themselves or represent selectively neutral or weakly non-adaptive correlates of other adaptive characters.

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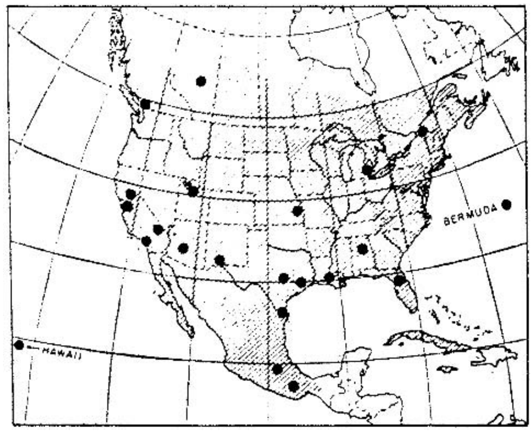

Fig. 1 Map of North America showing distribution of house sparrow (shaded area) and localities where specimens were taken (dots)

In analyzing individual and geographic variation in color, as well as in size, we have found it useful to segregate our specimens into adult and first-year age groups, since many of the characters studied exhibit significantly different means and variances in the two age groups.

In general, geographic variation is more pronounced in characters of color than in those of size. Specimens from northern and Pacific coastal localities and from the Valley of Mexico (Mexico City) are darkly pigmented, and those from Vancouver, British Columbia, are especially dark. Sparrows from collecting stations in the arid southwestern United States from southern California east to southern and central Texas are relatively pale in color, with extremes of pallor being achieved in samples from Death Valley, California, and Phoenix, Arizona. Samples from Salt Lake City, Utah, Lawrence, Kansas, and other localities in North America can be categorized broadly as intermediate in color. Specimens from Zachary, Louisiana, and Oaxaca City, Mexico, have a conspicuous yellow wash on the posterior under parts which is absent or only weakly indicated in birds from other North American localities.

Geographic variation in color of the breast in female house sparrows from Honolulu, Hawaii, and several localities in North America is shown in Fig. 2, which presents spectral reflectance curves [8] for five specimens from each locality.

The overall geographic pattern of color variation in North American house sparrows conforms with Gloger's ecogeographic rule, which relates color to regional variation in temperature and humidity [9]. Students of geographic variation in warm-blooded vertebrates a priori expect native North American species to be darker along the northwestern coast and paler in the arid southwest.

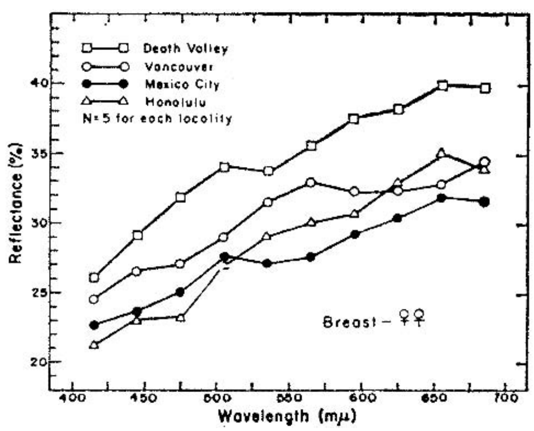

Fig. 2 Spectral reflectance curves for the breast of female house sparrows from Honolulu, Hawaii, and several North American localities

The fact that house sparrows manifest this pattern of variation is evidence for the selective action of the same environmental factors that are assumed to be significant for native species.

Sparrows from Oahu, Hawaiian Islands, are very distinctive in color, being unlike specimens from English, German, and North American localities. They are characterized by a reduced value of the dark markings of the plumage, a general absence of fine streaks on the under parts, and an overall rufous-buff color which is especially intense on the breast and flanks. The legs and feet tend strongly to be pale buff in color rather than dark brown as in continental birds. The unusually strong differentiation of the sparrows of the Hawaiian Islands is not surprising in view of their geographic isolation and the fact that they have had an evolutionary history apart from North American populations. Sparrows were introduced to the islands in 1870 or 1871 from a New Zealand stock, which in turn had been brought to New Zealand from England in the years 1866–1868 [10].

We emphasize the fact that geographic variation in color in New World house sparrows does not consist merely of subtle average differences among the samples, with broadly over-lapping ranges of variation. On the contrary, in many cases the color differences between samples are both marked and consistent, permitting 100 percent separation of specimens from the two localities. For example, we have observed no overlap in color of the pileum (top of the head) in females between samples from Oakland, California, and Progreso, Texas, or between those from Death Valley, California, and Vancouver, British Columbia. Again, specimens of either sex from the Hawaiian Islands and any of the North American localities are consistently separable on the basis of color.

Geographic variation occurs as regularly in size as in color, and for the most part parallels trends which are generally characteristic of indigenous species. The pattern of variation in size is largely clinal: in the United States and Canada the largest individuals are from the more northerly localities sampled, the smallest are from the desert southwest, and birds from other stations are of intermediate sizes. Some indication of the degree of geographic variation in wing length in North America is provided by data for adult males of three populations shown in Fig. 3. Variation in bill length in four representative populations is shown in Fig. 4. Note that the bill averages longer in the sample from Honolulu than in the continental populations. This was previously suggested by Lack's data [4], which had been considered equivocal because of uncertainties concerning seasonal variation in his material [11].

For reasons not important to the exposition here, wing length in house sparrows tends to vary independently of body size. Body weight is a good index of size, provided weights are taken from specimens having similar relative amounts of body fat, a character known to vary seasonally with the gonadal condition of the individual [12]. Our samples are strictly comparable, since all were taken when the birds had just completed the annual molt and were in the same condition gonadally and physiologically; all specimens show uniformly moderate degrees of subcutaneous lipid deposition.

In Fig. 5, mean body weights of adult males from 17 localities in North America are plotted against isophanes of the localities. Isophanes are calculated from latitude, longitude, and altitude of

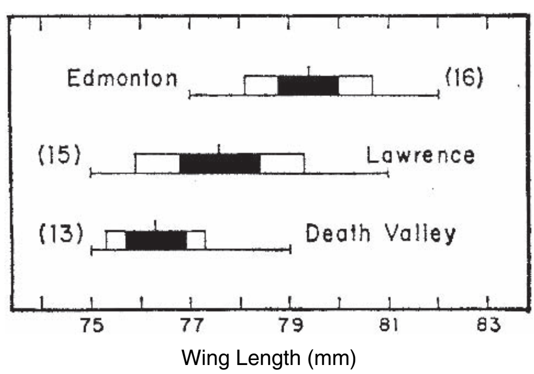

Fig. 3 Individual and geographic variation in wing length in adult male house sparrows from three localities. Vertical line: mean; horizontal line: observed range; solid rectangle: 2 standard errors on either side of mean; open rectangle: one standard deviation on either side of mean

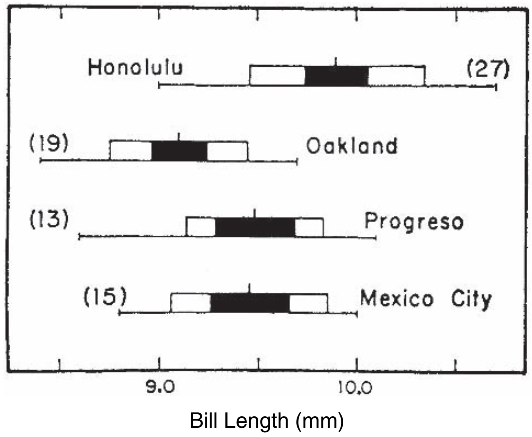

Fig. 4 Individual and geographic variation in bill length (from nostril) in adult male house sparrows from four localities. See Fig. 3 for an explanation of the graph

the localities and hence reflect gross climatic features [13]. For localities north of southern Texas a simple relationship is evident, and a straight line fitted to the points by the method of least squares has the equation $Y = 24.1 + 0.12X$. The regression coefficient is highly significant (99.9 percent confidence interval = 0.02 to 0.21), and 93 percent of the variability is attributable to the linear regression effect. The observed relationship, which was predictable on the basis of Calhoun's geographically more restricted study [5] of linear dimensions, exemplifies the ecogeographic rule of Bergmann, which describes adaptive trends in body size as they relate to problems of heat flow and temperature regulation [9]. Birds of larger body size occur at localities having high isophane numbers, reflecting boreal climates with severe winter cold; and those of smaller body size are from stations with low isophane numbers, reflecting mild or austral climates, occasionally with severe summer heat. A similar relationship between body size and climate is found in many native species of birds.

South of latitude 28°N in North America, other selective factors tend to override the effects of selection for body size as described by Bergmann's ecogeographic rule. Although mean body weight in the sample from Oaxaca City does not fall far from an expected position along the regression line

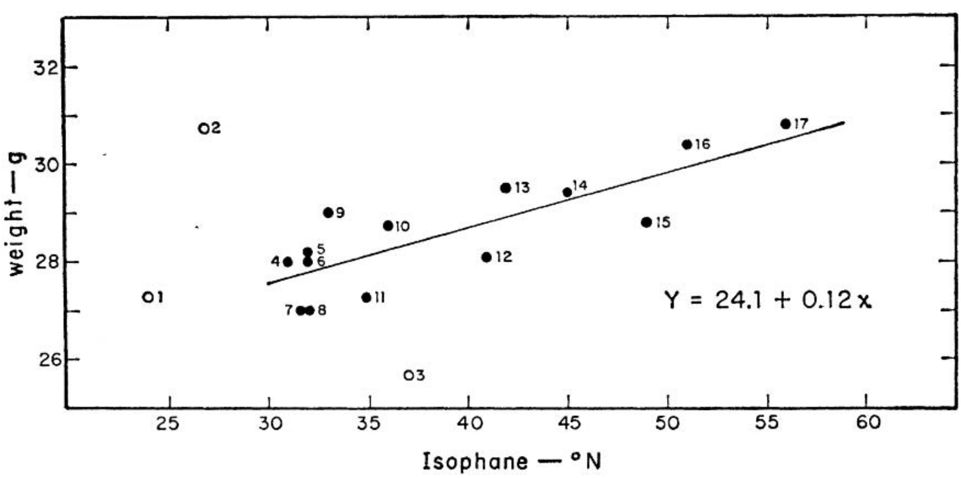

Fig. 5 Mean body weights of adult male house sparrows plotted against isophanes (see text for explanation). Localities: 1, Oaxaca City, Mexico; 2, Progreso, Tex.; 3, Mexico City, Mexico; 4, Houston, Tex.; 5, Los Angeles, Calif.; 6, Austin, Tex.; 7, Death Valley, Calif.; 8, Phoenix, Ariz.; 9, Baton Rouge, La.; 10, Sacramento, Calif.; 11, Oakland, Calif.; 12, Las Cruces, N.M.; 13, Lawrence, Kan.; 14, Vancouver, B.C.; 15, Salt Lake City, Utah; 16, Montreal, Quebec; 17, Edmonton, Alb. The regression line is based on data from localities 4 to 17

based on data from samples taken in the United States and Canada, birds from Mexico City are surprisingly light in weight and those from Progreso, Texas, are unexpectedly heavy. That these differences reflect real variation in body size and not merely nongenetic variation in level of fat deposition is indicated not only by examination of the fat condition of the specimens but also by data on the length of the tarsus, which in house sparrows is closely correlated with body weight.

Current taxonomic practice gives formal nomenclatural recognition, at the subspecific level, to morphologically definable geographic segments of species populations. And it is obvious that the levels of differentiation achieved by the introduced house sparrow in the Hawaiian Islands and in a number of areas in North America are fully equivalent to those shown by many polytypic native species. Although application of subspecific trinomials to certain New World populations of sparrows would be fully warranted, we are not convinced that nomenclatural stasis is desirable for a patently dynamic system. Nomenclatural considerations aside, the evolutionary implications of our findings are apparent. Current estimates of the minimum time normally required for the evolution of races in birds range upward from about 4000 years [14], and nowhere is there a suggestion that such conspicuous and consistent patterns of adaptive evolutionary response to environments as we have found in New World house sparrows are to be expected within a period covering not more than 111 generations. Actually, much of the differentiation in North American populations must have occurred in the present century, since sparrows did not reach Mexico City until 1933 [15], and they were not present in Death Valley before 1914, or in Vancouver before 1900. Our findings are consistent with recent evidence of evolutionary changes in some other groups of animals, including mammals and insects [16], within historical times. Clearly, our thinking must not exclude the possibility of animals attaining to extremely rapid rates of evolution at the racial level.

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On the Role of Alien Species in Urban Flora and Vegetation

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Abstract The role of alien species in urban vegetation is reviewed. Representation of aliens is compared with that of native species by taking into account both historical and spatial aspects. On a long-term time scale (more than a century), the importance of aliens, especially neophytes, is increasing. The representation of alien species, if expressed in quantitative terms, shows a close relationship to the spatial structure of the city, and decreases from the city center towards the outskirts. In many cases, this trend is still valid at the level of particular species, as exemplified by an alien tree *Ailanthus altissima*. The relationship between species richness and the level of man-induced disturbance supports the intermediate disturbance theory only if the native species are considered; aliens, both archaeophytes and neophytes, are encouraged in sites subjected to high disturbance levels. At the level of phytosociological alliances, the proportion of alien species was closely correlated with the disturbance level, although the relationship was rather variable between particular alliances. As regards the role of alien species in succession on urban waste land, it appears that alien species may persist as dominants in succession for a long period. The following factors promoting the success of alien species in urban environment are discussed: availability of specific urban niches, high level of disturbance in urban environment, and the minor isolation of seed sources.

Keywords: plants · urban vegetation · invasive species · native species · exotic species · succession · plant diversity · gradient analysis

Introduction

In one of the first studies of rural-urban gradients, the Finnish botanist Linkola (1916) proved with quantitative data that the number of alien species increases from natural forests through meadows to arable land and human settlements. Later, comparisons between some European settlements have shown a close relationship between the presence of alien species and the size of settlements (Falinski 1971; Sukopp and Werner 1982; Pyšek 1989). This is usually explained by the considerable habitat heterogeneity, the role of big cities as centres of species' immigration and the better adaptation of alien species to man-made perturbations (Sukopp 1981; Sukopp and Trepl 1987; Kowarik 1990a, 1991; Pyšek 1993).

In this paper, the role of alien, compared to native, species in the urban flora and vegetation will be illustrated by addressing the following questions:

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1. What is the proportion of alien compared to native species in the urban flora and do both groups differ in frequency patterns?
2. Do special patterns exist in the spatial distribution of alien *versus* native species?
3. Are alien species better adapted to man-made disturbance than natives?
4. What role do alien species play during the course of succession in urban vegetation?

Data sources and definitions

This paper refers mainly to Berlin, Germany, as a city with an extensive background of research in urban ecology during the last 40 years (Sukopp 1990, and papers in Sukopp *et al.* 1990). In order to elucidate some general trends in the performance of alien species in the urban environment, two extensive data sets are considered: the work of Kunick (1974, 1982a) who analysed spatial patterns in the occurence of species, and a dataset with 5136 vegetation relevés including most of the plant communities currently existing in Berlin (Kowarik 1988). Additionally, quantitative analysis on alien species in the woody vegetation of derelict sites (Kowarik 1992a) are used. Taxonomy and species' status (native/alien) is in accordance with the list in Sukopp *et al.* 1982.

Because of the long tradition of differently used terms some definitions are necessary: 'Alien' (= non-native, non-indigenous) species are those occurring in an area in which they have not evolved since the last Ice Age and whose introduction or immigration was assisted deliberately or involuntarily by human activities. This definition of alien species is in the tradition of Thellung (1912, 1918/19, see Trepl 1990a). It also follows the definition by Roy (1990), but enlarges it by two additions. First, the reference to the last Ice Age must be made; second by the connection to the role of humans assisting the introduction of alien species. The former is necessary in order to exclude species as natives which had formerly evolved in the area but became extinct during the colder periods. Re-introduced they should be treated as aliens because if they occurred in the area before or during the last Ice Age, it was not under present conditions as the climate was different from today (Webb 1985), and the chance of coevolution with other organisms was limited. The latter reference is necessary because most native species did not evolve *in situ* either, but arrived in the period since the last Ice Age, as Egler (1961) stated. Compared with alien species, however, the appearance of natives has not necessarily been supported by humans (see also Pyšek 1995).

Refering to Thellung, but accepting the terminological changes by Schroeder (1969), in the central European tradition alien species are usually divided in two groups according to the time of their introduction or immigration: archaeophytes brought in up to 1500 AD, and neophytes brought in after this date.

For studying the urban flora and vegetation, it is advantageous to consider the total area within the political borders of a city (Klotz 1990; Kowarik 1992b). Although this border is not defined ecologically, this approach is quite appropriate to the transition character of various habitats on the urban-rural gradient (*e.g.*, urban forests - 'natural' forests, urban settlements - rural settlements) which may be discussed in terms of the ecotone concept (Pyšek 1992) and which offers 'unexploited opportunities' for ecological research (McDonnel and Pickett 1990).

Species' responses to increasing disturbance have been studied by using the hemeroby-approach (Jalas 1955; Sukopp 1972, 1976; Kowarik 1988), which refers explicitly to the man-made components of disturbance (details in Kowarik 1990a).

Alien species in the urban flora

Species numbers and frequency

Dividing Berlin's present day flora into native and alien species reveals a large proportion of aliens: 593 alien species (41%), which include 167 archaeophytes and 426 neophytes, as opposed to 839

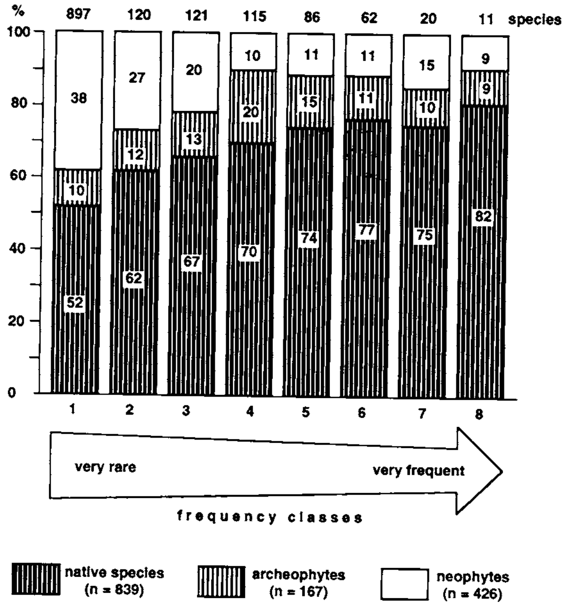

Fig. 1 Division of Berlin's flora according to frequency classes (based on the occurrence in 5136 phytosociological relevés made in the territory of the city). Proportion of native and alien species (the latter divided into archaeophytes and neophytes) is shown and absolute species numbers in each category are indicated for each frequency class on the top of the bar (after Kowarik 1988)

native species (59%) (Kowarik 1988). This differs from Falinski (1971) who assumed a proportion of aliens between 50–70% in big cities. A limit of roughly 50% aliens seems to exist in central Europe. Even from highly industrialized cities in Northrhine-Westphalia, only between 42 and 55% aliens are reported in the urban flora (Reidl and Dettmar 1993).

In Fig. 1, native and alien species of Berlin's flora are grouped in 8 classes according to their current frequency. In contrast to urban floras from the 19th century, most species are currently rare. The proportion between native and alien species changes directionally with increasing frequency: 48% of the rarest species are aliens, but only 18% of the most common species are aliens.

Some historical analyses have shown that, mainly due to industrialisation, a wave of newly introduced species reached central Europe in the second part of the 19th century (Scholz 1960; Sukopp 1976; Jäger 1977, but see Kowarik 1995 on effects of the climatic warming). Beyond changes in the species number, shifts in species' frequency have to be considered. A comparison of species' current frequency in Berlin with data from the 19th century (Ascherson 1864) reveals evident differences between native species and both groups of alien species (Table 1).

Two thirds of the species which became more frequent during the last 120 years are neophytes. In contrast, most of the decreasing species are natives (78%) which are mainly listed in Red Data

Table 1 Changes in frequency of native and alien species in the flora of Berlin during the last 120 years. Aliens are divided in archaeophytes and neophytes. Species with constant frequency are not shown. From Kowarik 1992b after Kutschkau 1982 (*a, b*) and Böcker *et al.* 1991 (*c*). Frequency values in % are given

	Native	Archaeophytes	Neophytes
a. Species with increasing frequency (100% = 361 species)	26.9	8.0	65.8
b. Species with decreasing frequency (100% = 821 species)	78.0	13.8	8.3
c. Extinct and endangered species (100% = 525 species)	80.2	11.2	8.6

Books: in Berlin, 51% of the native species, 37% of the archaeophytes, but only 20% of the neophytes are endangered (Böcker *et al.* 1991). Obviously, unlike the neophytes, the archaeophytes which are mostly confined to agricultural disturbance regimes, are not encouraged by the effects of urbanization. Similar trends have been found in Halle, Warsaw, and Poznan (Klotz 1984; Sudnik-Wojcikowska 1987; Jackowiak 1989).

Table 2, showing the most common alien species in Berlin, also indicates conspicuous shifts in species' frequency. Half of the 20 most frequent neophytes have had a shift through more than three frequency classes during the last 120 years. In contrast, most of the common archaeophytes were already frequent, or more frequent, during the last century. The same is true for most native species. From these data, the increasing importance of alien species, mainly neophytes, in the urban environment can be predicted.

Spatial patterns

Abiotic environmental conditions usually change on rural-urban gradients from the centre to the outskirts of cities (Kuttler 1988; McDonnel and Pickett 1990; Bullock and Gregory 1991; Pyšek 1992). Due to the polycentric structure of many European cities, these changes are spatially diversified.

Table 2 The currently most frequent alien species (archaeophytes and neophytes) in the flora of Berlin (frequency classes decreasing from 8 to 5, see Fig. 1, Kowarik 1988). Striking changes in frequency compared with Ascherson's flora from 1864 (after Kutschkau 1982) are indicated. Species with frequency having increased by +: 2–3 frequency classes; ++: 4–5 frequency classes; + + +: 6 frequency classes. Species with frequency having decreased by –: 2–3 frequency classes; –: 4–5 frequency classes; — 6 frequency classes. No symbol is given for species with more or less constant frequency. For native species, only the number of species belonging to a given frequency class is shown

Archaeophytes	Neophytes		Native
Frequency class 8			
Poa annua	+ + +	*Solidago canadensis*	9 species
Frequency class 7			
Capsella bursa-pastoris		*Conyza canadensis*	
+ *Plantago major*	+	*Impatiens parviflora*	
	++	*Prunus serotina*	15 species
Frequency class 6			
– *Apera spica-venti*	+	*Arrhenatherum elatius*	
Berteroa incana	++	*Bidens frondosa*	
Convolvulus arvensis		*Galinsoga parviflora*	
Fallopia convolvulus		*Oenothera biennis*	
Plantago lanceolata	+	*Poa compressa*	
Tripleurospermum inodorum	+ + +	*Rumex thyrsiflorus*	
Veronica arvensis	+ + +	*Sisymbrium loeselii*	48 species
Frequency class 5			
– *Ballota nigra*	++	*Acer negundo*	
Bromus tectorum	++	*Clematis vitalba*	
+ *Crepis capillaris*	–	*Euphorbia cyparissias*	
Euphorbia peplus	++	*Matricaria discoidea*	
– *Hordeum murinum*	+	*Oenothera chicaginensis*	
– *Lamium purpureum*	?	*Oxalis fontana*	
– *Polygonum lapathifolium*	++	*Parietaria pensylvanica*	
P. persicaria	++	*Robinia pseudoacacia*	
– *Setaria viridis*	?	*Solidago gigantea*	
Solanum nigrum			
Sonchus oleraceus			
Urtica urens			
Viola arvensis			64 species

Table 3 Percentage of native and alien species (divided in archaeophytes and neophytes) on an urban-rural gradient. Shown are data from Berlin (city zonation according to Kunick 1982), from a grid mapping of *Atlanthus altissima* (Böcker and kowarik 1982) and data from surrounding rural districts of Brandenburg (Klemm 1975). Data in %

	Natives	Archaeophytes	Neophytes	Atlanthus altissima*
Berlin				
Zone 1 (centre)	50.2	15.2	34.6	92.2
Zone 2	53.1	14.1	32.8	46.1
Zone 3	56.6	14.5	28.9	24.8
Zone 4 (outskirts)	71.5	10.2	18.3	3.2
Brandenburg districts				
Spremberg	74.8	8.1	17.1	–
Ruppiner Land	75.7	8.9	15.4	–
Priegnitz	77.5	9.1	13.4	–
Dahme	78.4	10.6	11.0	–
Spreewald	79.3	10.4	10.3	–

*Percentage of occupied squares mapped in a grid in zones 1-4

There is a considerable evidence that the distribution patterns of plants and animals correspond significantly with the spatial structure of cities (for a review see Wittig *et al.* 1993).

In Berlin, Kunick (1974, 1982a) worked out a city zonation using floristic data. Table 3 shows that the proportion of natives and aliens changes directionally on the urban-rural gradient. In the city centre (zone 1), the proportion of natives is about 50%, increasing to more than 70% in the outskirts (zone 4). Data from surrounding rural districts of Brandenburg show that in less urbanized areas, the proportion of natives may increase to about 80% of the flora. A higher percentage of archaeophytes compared to neophytes is typical of rural landscapes (Pyšek and Pyšek 1988, 1990). City centres, however, may be characterized mainly by neophytes. One of them is *Ailanthus altissima*, native to China, which is virtually confined to urban-industrial areas in most parts of central Europe (Kowarik and Böcker 1984; Gutte *et al.* 1987; Müller 1987; Landolt 1991). Results of a grid mapping incorporated in Table 3 show a close relationship between the distribution of this woody species, and Kunick's city zonation of Berlin. Similar results have been found in Vienna (Kugler cited by Punz 1993).

Apart from the general trend on the urban-rural-gradient (increasing numbers of aliens from the outskirts to the centre), variations due to the specific habitat structure have to be considered. High numbers of aliens are usually found on industrial and railway sites as well as in built-up areas (Brandes 1983; Kowarik 1986; Rebele 1986; Wittig *et al.* 1989; Aey 1990; Dettmar 1991; Reidl and Dettmar 1993; Reidl 1993). The presence of safe sites for the germination and establishment of alien species may be due to the history of a city. Presumably, closely built-up city centres are less rich in alien species than cities including ruins and derelict areas. Differences between particular cities can be explained either by varying abiotic, mainly climatic, conditions or by methodological problems concerning the selected areas and species groups (Kunick 1982b; Brandes 1989; Wittig 1989; Pyšek 1993).

Responses to man-made disturbance

General pattern

The hypothesis that disturbance promotes the establishment and spread of alien species is accepted in most studies on biological invasions (*e.g.*, Trepl 1983, 1990b; Orians 1986; Crawley 1987; Sukopp and Trepl 1987; Hobbs 1989; Rejmánek 1989). No ecosystem is absolutely free of disturbance (Pickett and White 1985), but in the urban environment, it is mainly the man-made components of disturbance which have to be considered as driving forces affecting the species composition.

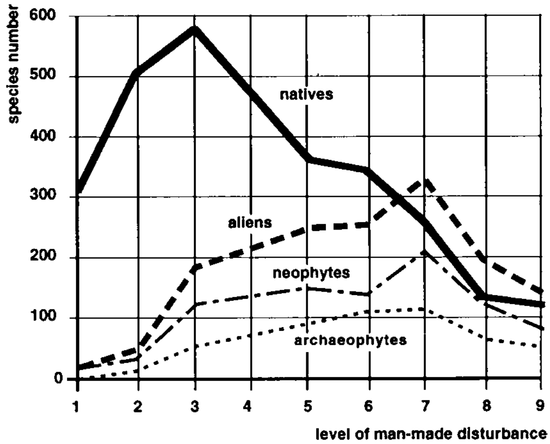

Fig. 2 The relationship between species richness and the level of man-made disturbance. Each of 5136 vegetation relevés made in the Berlin vegetation was classified using a 9-degree hemeroby scale (expressing the level of man-induced disturbance, see Kowarik 1990a). The total number of native and alien species (the latter divided into archaeophytes and neophytes) was then calculated for relevés belonging to the same hemeroby class

Referring to a data set of 5136 vegetation relevés which had each been assigned to one of nine levels of man-made disturbance (hemeroby, see Kowarik 1988, 1990a), the relationship between increasing disturbance and species richness in natives, archaeophytes and neophytes has been analysed. At first glance, the results shown in Fig. 2 confirm the intermediate disturbance hypothesis which means that species richness is highest at an intermediate level of disturbance and lowest under conditions of both high and low disturbance (Grime 1979; Connell 1979). But this is only true for native species. Regarding the alien species, an opposite trend is revealed: archaeophytes and neophytes are generally enhanced on sites subjected to a higher level of disturbance. These results may help to refine the intermediate disturbance hypothesis by differentiating according to species' origin. Furthermore, they support the theory on the positive relationship between disturbance and the promotion of alien species.

Differences among vegetation types

By grouping the 5136 relevés in 52 phytosociological alliances of the Braun-Blanquet system, which represent virtually the total vegetation of Berlin (according to the list of Sukopp 1979), the relationship between the average proportion of alien species and the average disturbance level which had been calculated for each of these 52 vegetation units, can be tested (data in Fig. 4). For each vegetation relevé the proportion of alien species (% of total species number) and the level of man-made disturbance was calculated, the latter by using species' indicator values for hemeroby from Kowarik (1988). In Fig. 3 average data for 45 vegetation units are shown. There is a significant correlation ($r = 0.95$, $P < 0.001$) between the level of man-made disturbance and the proportion of alien species. Because of the varying ratio between archaeophytes and neophytes in several vegetation units (see Fig. 4), the correlation between disturbance and these groups is less strong ($r = 0.89$ and 0.68, respectively) but still significant ($P < 0.001$).

Analysis of communities belonging to the same vegetation unit (at the level of phytosociological alliances) but subjected to different levels of human impact, reveals less clearcut trends (Table 4). When the relationship between the proportion of aliens and level of man-made disturbance was tested (calculation based on single vegetation relevés available for a given alliance), a positive sig-

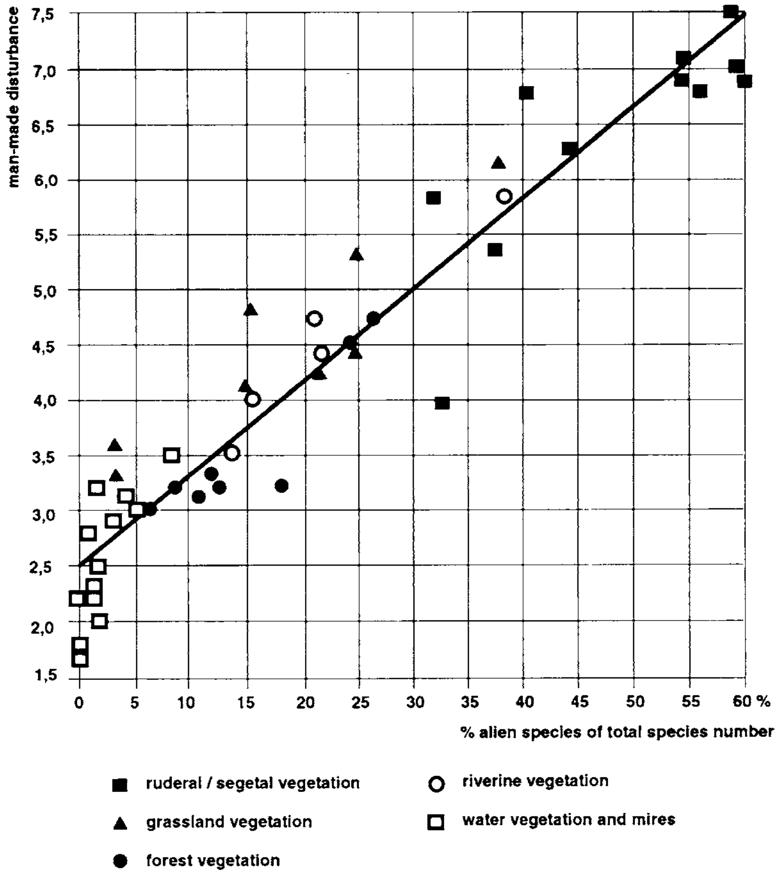

Fig. 3 The relationship between the level of man-made disturbance and the proportion of alien species in various vegetation units (phytosociological alliances representing virtually the total vegetation of Berlin). The average proportion of alien species and average disturbance level (expressed by a 9-degree hemeroby scale, see Kowarik 1990a) were calculated for each vegetation unit. Particular vegetation types are distinguished by using different symbols

nificant correlation between both variables was found in 32 alliances (65.3%) whereas in 17 alliances (34.7%) the relationship was not significant. A similar result was obtained for archaeophytes treated separately (significant correlation between both variables in 33 alliances, *i.e.*, 67.3%); for neophytes the respective figure was 19 (38.7%).

In detail, the following conclusions may be drawn from these results (Table 4):

a. Archaeophytes show a closer relationship to disturbance than neophytes: in 25 alliances, the significance level of the correlation coefficient was higher for archaeophytes than neophytes whilst in only 3 cases, neophytes were more closely correlated with the level of disturbance than archaeophytes.
b. In most vegetation units of wet sites (water and mire vegetation), no significant correlation between the representation of alien species and the level of disturbance was found.
c. The relationship between the representation of alien species and the level of disturbance was not significant in several units of segetal pioneer vegetation. This may be due to the constantly high proportion of aliens in this vegetation type.

d. With few exceptions, there was a close relationship between the proportion of aliens and the level of disturbance in vegetation units dominated by perennial grasses or herbs. This holds true for dry, mesic, and wet sites.

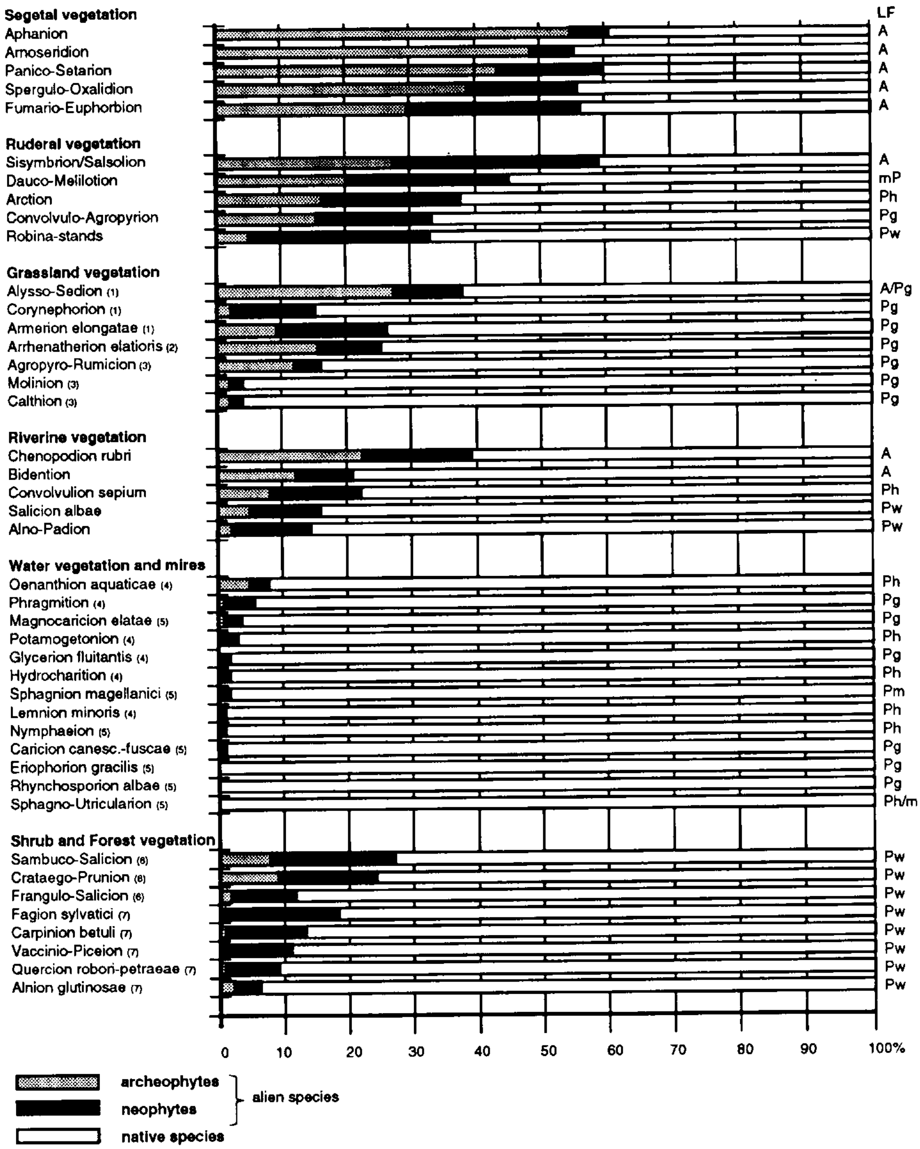

Fig. 4 The representation of native and alien species (the latter divided into archaeophytes and neophytes) in the vegetation of Berlin. The percentage contribution of each category to the total number of species is shown for particular phytosociological alliances (after Kowarik 1990a). The vegetation units are roughly arranged according to successional age and additional information on the vegetation type is indicated in brackets following the name of the alliance: 1: dry grassland; 2: mesic grassland; 3: wet grassland; 4: water vegetation; 5: mire vegetation; 6: shrub vegetation; 7: forest vegetation. Dominant life form (LF) is indicated: A: annual; mP: monocarpic perennial; Ph: perennial, mainly herbs; Pg: perennial, mainly grasses; Pm: perennial, mainly mosses; Pw: perennial, mainly woody species

Table 4 Relationship between the proportion of archaeophytes (A), neophytes (N) and alien species in total (A + N) and the level of man-made disturbance in the vegetation of Berlin. Calculations were based on vegetation relevés assigned to particular phytosociological alliances (after Kowarik 1988). The significance level of the correlation coefficient (t-test) is shown: $^{*}P < 0.05$, $^{**}P < 0.01$, $^{***}P < 0.001$, NS: non significant; if not given, the respective unit was free of alien species. The following groups of vegetation are indicated: Pr: ruderal pioneer vegetation; Ps: segetal pioneer vegetation; R: riverine vegetation (RF: riverine forests); H: vegetation dominated by perennial herbs; G: grassland vegetation (Gd: dry, Gm: mesic, Gw: wet); W: water vegetation and mires; F: forest vegetation (Fr: ruderal forests, RF: riverine forests)

Alliance	A + N	A	N	Vegetation
Dauco-Melilotion	***	NS	***	Pr
Sisymbrion	***	***	***	Pr
Arnoseridion	NS	NS	NS	Ps
Spergulo-Oxalidion	NS	NS	NS	Ps
Panico-Setarion	***	**	NS	Ps
Fumario-Euphorbion	***	***	NS	Ps
Aphanion	**	NS	*	Ps
Chenopodion rubri	*	**	NS	P/R
Bidention	***	***	*	R
Convolvulion	***	***	*	R
Agropyro-Rumicion	***	***	NS	R
Geo-Alliarion	***	***	**	H
Arction	***	***	NS	H
Aegopodion	*	***	NS	H
Convolvulo-Agropyrion	***	***	NS	H
Trifolion medii	NS	NS	NS	H
Molinion	***	**	***	Gw
Calthion	***	***	NS	Gw
Arrhenatherion	***	***	***	Gm
Polygonion avicularis	***	***	***	Gm
Koelerion	**	NS	**	Gd
Alysso-Sedion	***	***	NS	Gd
Corynephorion	**	***	*	Gd
Armerion	***	***	NS	Gd
Oenanthion	***	***	NS	W
Glycerion	NS	***	NS	W
Nanocyperion	*	**	NS	W
Phragmition	***	***	NS	W
Filipendulion	NS	**	NS	W
Magnocaricion	***	***	*	W
Caricion canescenti-fuscae	NS	NS	NS	W
Sphagnion magellanici	NS	NS	NS	W
Lemnion	NS	NS	NS	W
Hydrocharition	NS	NS	NS	W
Potamogetonion	NS	NS	NS	W
Nymphaeion	NS	NS	NS	W
Eriophorion	–	–	–	W
Sphagno-Utricularion	–	–	–	W
Alno-Padion	NS	***	**	RF
Salicion albae	NS	***	NS	RF
Crataego-Prunion	***	***	***	F
Frangulo-Salicion	**	***	**	F
Alnion glutinosae	***	***	***	F
Rubo-Salicion	*	*	NS	F
Robinia stands	NS	***	NS	Fr
Quercion robori-petraeae	***	***	***	F
Fagion	***	NS	***	F
Carpinion	**	**	*	F
Vaccinio-Piceion	NS	NS	NS	F

Alien species in the urban vegetation

Representation of alien species in particular vegetation types

From urban environments, plant communities dominated by alien species were mainly reported from railway areas and from heavily disturbed urban-industrial sites. In Essen, for example, Reidl and Dettmar (1993) found 55 communities (of a total of 262) which are characterized by alien species, among them 27 belonging to the annual and perennial ruderal vegetation (*e.g.*, communities with *Inula graveolens* and *Chenopodium botrys*, Dettmar and Sukopp 1991).

Considering the data set representing the vegetation of Berlin, comparable information can be derived on the role of alien species in various urban, seminatural and natural vegetation units. 5136 vegetation relevés were grouped in phytosociological alliances, and average representation of alien species was calculated for each alliance (expressed as the percentage of alien species per relevé) (Kowarik 1988, 1990a).

In Fig. 4, the vegetation units are grouped as follows: segetal vegetation, ruderal vegetation, grassland vegetation, riverine vegetation, water and mire vegetation, shrub and forest vegetation. For each vegetation unit, the dominant life form is shown. From the pattern of alien species which are divided in archaeophytes and neophytes, the following conclusions can be derived:

1. In contrast to the early stages of old field succession in non-urban areas, including less than 15% aliens of the total species number (examples in Rejmánek 1989), the proportion of alien species is much higher in the segetal and ruderal vegetation (up to 60% in the Aphanion or Sisymbrion).
2. The highest percentages of alien species are found in vegetation units with annuals as the dominating life form, *i.e.*, all units of segetal vegetation and the pioneer stages of ruderal vegetation (Sisymbrion/Salsolion).
3. Comparing archaeophytes and neophytes as early and later immigrants shows distinct differences: archaeophytes prevail in the vegetation of arable fields, neophytes in ruderal vegetation units. High percentages of archaeophytes are only found in vegetation units characterized by annuals.
4. As expected, there are two trends concerning the moisture gradient and the stage of succession. Linkola (1916) found less alien species in wet compared to mesic meadows. The data in Fig. 4 support the generally accepted relevance of the moisture gradient for the establishment of alien species (Rejmánek 1989). The vegetation of mires is virtually free of alien species, and there are only a few in aquatic vegetation. In grassland vegetation, the trend is clear: starting from 25% in the Armerion elongatae on dry sites, the percentage of alien species decreases to only 4% on wet sites (meadows of the Calthion and Molinion).
5. In the course of succession the percentage of alien species decreases conspicuously: In the ruderal vegetation, the annual phase starts with 59% aliens. The Dauco-Melilotion which is already dominated by monocarpic perennials, still includes 44% of alien species, but in the perennial ruderal vegetation (dominated by both grasses or herbs) the average percentage of aliens decreases to about one third. Grassland vegetation generally includes less than 30% of aliens. In the Quercion robori-petraeae as the most frequent woody vegetation in the fringe of the city (dominated by oaks and Scotch pine), even less than 10% of all species are aliens. (These 10% include the North American *Prunus serotina*, a species which was able to establish dense populations after having been planted by foresters, Starfinger 1991).
6. Nevertheless, on a much higher level, compared to areas beyond the impact of urbanization, the percentage of alien species seems to decrease during succession as expected. However, woody vegetation on urban sites may include a surprisingly high amount of aliens: 33% of total species number have been found in urban stands of *Robinia pseudoacacia*. These stands (which have not

been planted) represent the oldest stages of old field succession on urban sites which developed in the centre of Berlin after the WW II. Their age varies between 25 and 35 years.

May alien species persist in the urban woody vegetation?

In most studies on alien species as dominants or co-dominants in urban woody vegetation, their association with other species has been analysed (*e.g.*, Kohler and Sukopp 1964a,b; Kowarik and Böcker 1984; Becher and Brandes 1985; Kunick 1990; Diesing and Gödde 1989; Schmitz 1991). Although most of the studied stands represent later stages of successional series, only a little information is available on the role of alien shrub and tree species through succession. The results shown in Fig. 4 confirm a decreasing proportion of alien species during succession as a general trend. However, the existence of more than 30-year-old successional stages which are both rich in and dominated by alien species (such as the urban stands of *Robinia pseudoacacia* in Berlin) does not fit this theory. Obviously, alien species may persist longer than expected in urban old field successional series.

The succession on urban wastelands in Berlin may serve as an unintended long-term experiment on the performance of alien compared to native species over more than four decades of succession. Vast sites were destroyed during and after the WW II, and due to the political situation in Berlin, many of them in the western part of the city have remained abandoned until today. Because of the total destruction, both native and alien species had to immigrate in order to colonize the open sites. After more than four decades of succession the woody vegetation in five wastelands (totally 98.7 ha) has been analysed (see Kowarik 1992a,c for details).

On average, 45% of the forests are dominated by aliens, 55% by native species. *Robinia pseudoacacia*, native to North America, is the most successful alien tree with *Betula pendula* as its native counterpart. However, there are extreme differences between single areas: alien woody species dominate between 5% and 74% of the woody vegetation, even in areas with comparable site conditions (such as derelict railway areas).

The vegetation of one railway area of about 20 ha had been mapped twice in 10 years. The analyses of the vegetation maps revealed that the area covered by woody vegetation virtually doubled in ten years from 37% to 69%. However, the proportion of alien and native trees as dominants of forest vegetation remained almost unchanged, with 37:63 in 1991 compared to 38:62 in 1981. Obviously, both *Robinia* and *Betula* may persist and enlarge the area occupied over time.

But how long may *Robinia* dominate ruderal forests? Demographic studies show urban *Robinia*-stands being far from resistent to the invasion by other woody species (Kowarik 1990b): a strikingly high number of 38 tree, 35 shrub, and 4 species of woody climbers have been found in *Robinia*-stands, and about half of these species are aliens in Berlin. Compared to native *Betula pendula*, the stands of *Robinia* harbour more individuals of, mainly shade tolerant, tree and shrub species. Focusing upon the performance of shade tolerant and tall growing trees as potential competitors of *Robinia*, indicates future succession trends. On average, shade tolerant trees (mainly *Acer platanoides, A. pseudoplatanus*) already represent 83% of all tree individuums in the herb layer (<0.9 m), 57% in the shrub layer (<5 m), but only 9% in the tree layer (>5 m). Most of the potential competitors are still confined to height classes up to 7 m, and in some stands, resulting from the distance to seed sources, competitors are completely missing in the shrub or tree layers.

These results indicate a strikingly slow change in the dominance structure of urban *Robinia*-stands in Berlin which contrasts clearly with the performance of *Robinia* in native American forests. In the Appalachians, it is replaced almost totally by other tree species within 20–30 years (Boring and Swank 1984). The immigration success of native species as potential competitors, however, indicates future changes in the dominance of tree species but this is presumably not a question of years, but of decades.

Discussion

Both historical and spatial analyses of the occurence of alien species in the urban environment have shown that non-native species are strongly promoted by urbanization in central European cities. In rural floras of Brandenburg, for example, there are less than 20% aliens, but from the outskirts to the city centre of Berlin, the percentage of alien species increases from about 30 to 50% of all species (Table 3). Comparisons with floristic data from the 19th century revealed increasing numbers of alien species (Scholz 1960; Klotz 1984; Sudnik-Wojcikowska 1987; Jackowiak 1989), and increasing frequencies of alien species. It is mainly neophytes as the later immigrants (since 1500 AD) that are promoted by the effects of urbanization (Tables 1, 2).

Even if there is no doubt on the general trend, it has to be questioned which of the diverse factors usually summarized under the term ‘urbanization’ may be responsible for the enhancement of alien species. I will discuss the role of three of them:

a. the availability of specific ‘urban’ niches,
b. the usually high level of disturbance in the urban environment, and
c. the minor isolation of seed sources.

Finally, some conclusions on the role of alien species during the course of succession will be derived from the results summarized in this paper.

Availability of specific ‘urban’ niches

The changes in environmental factors on urban-rural gradients may provide specific niches which could be realized by aliens, either exclusively or better than by native species. There are some examples of urban-industrial habitats which are virtually exclusively colonized by alien species: *Inula graveolens* and *Chenopodium botrys* on dumps and other industrial sites in North-Rhine-Westphalia (Gödde 1984; Dettmar and Sukopp 1971), the latter species also on burning dumps in Lille (Lampin 1969), and on heavily disturbed calcareous sites in the post-war Berlin (Sukopp 1971). *Buddleja davidii* is often the only woody pioneer colonizing crevices in abandoned housing areas in London (Burton 1983), on gravel in railway areas (Schmitz 1991), or on dumps as dominant species or as a co-dominant of *Betula pendula* (Dettmar 1991). These examples of successfully colonizing alien species refer to sites which may be extreme both in terms of stress (*e.g.*, low pH-values, limited water supply, high temperatures) or disturbance (*e.g.*, mechanical perturbation of soils). They support the view of Trepl (1993) who explained the often stated minor success of alien species in colonizing extreme habitats (*e.g.*, Rejmánek 1989) by the isolation from seed sources rather than by the limited invasibility of these sites.

Rising temperatures, often combined with a limited water supply, are characteristic factors which usually change directionally on rural-urban gradients. Considering the origin of many alien species in warmer areas (Scholz 1960; Sarisaalo-Taubert 1963), their increasing presence in the city centre(s) (usually coinciding with the centre(s) of the urban heat archipel) may be interpreted as a result of the pre-adaptation to urban conditions. In Berlin, for example, the spatial pattern of *Ailanthus altissima* coincides with the city zonation (Table 3) and spatial structure of the heat archipel (Kowarik and Böcker 1984). There is some evidence, that the establishment of urban heat islands which exacerbated the effects of generally rising temperatures since the 1850’s has in some cases enabled or, at least, promoted the invasion of species with high requirements for temperature (Kowarik 1995).

The relevance of temperature on rural-urban gradients is stressed by a corresponding pattern on elevation gradients. Usually, there are less alien species with increasing elevation (Frenkel 1970; Fox and Fox 1986; Brandes 1989), and in both cases, increasing percentages of alien species coincide with rising temperatures. Considering that varying temperatures are only one part of the diversification of urban site conditions stresses the role of specific 'urban' niches in opening invasion windows for alien species.

Man-made disturbance

The increasing proportion of alien species on the rural-urban gradient (up to 50% of all species in the centre of Berlin, Table 3) may be explained by a corresponding change in the intensity of human impact which is presumably highest in city centres. But there are usually less disturbed 'islands' in the centre (*e.g.*, parks, cemeteries) and also heavily disturbed areas in the outskirts (*e.g.*, refuse pits, industrial plants). Thus, the mosaic structure of many cities causes difficulties in interpreting the results of spatial analysis in terms of varying human impact.

However, the theory of disturbance (in the sense of Grime 1979) as promoting the establishment of alien species by reducing the competition of other species and by creating open space (*e.g.*, Crawley 1987) can be tested by using the hemeroby approach (hemeroby as an expression of the man-made components of disturbance, see Kowarik 1990a). Considering the vegetation of more than 5000 sites reveals distinctive differences in the floristic inventory between the nine groups of sites (Fig. 2): there are more aliens compared to native species on sites affected by higher intensities of man-made disturbance.

On the community level, a strong correlation between the level of man-made disturbance and the percentage of alien species has been found (Fig. 3). This trend is crystal-clear when considering vegetation units on a higher aggregated level, see Fig. 3 which refers to phytosociological alliances. However, not all, but many of these alliances include communities which are subjected to a different intensity of man-made disturbance. The more detailed analysis on the level of single phytosociological alliances, however, revealed a less clear, or even lacking, significant correlation between man-made disturbance and the percentage of alien species (Table 4). In general, these results confirm the role of disturbance as a driving force in enhancing alien species. In some vegetation units however, different amounts of alien species may not be sufficiently explained by a varying level of disturbance.

Isolation from seed sources

Considering the role of cities as centres of species' immigration (Sukopp 1976; Jäger 1977; Kowarik 1990a) could explain the high numbers of alien species in central European cities. However, in many cases good immigration conditions (*i.e.*, availability of dispersal vectors and of seed sources, both from unintentionally introduced diaspores and from planted individuals) coincide with the availability of specific urban sites which are often subjected to a high level of disturbance (*e.g.*, urban railway areas). Though the effects of disturbance, availability of specific niches, and the lack of isolation from seed sources can hardly be distinguished from each other.

The role of immigration conditions compared to the relevance of individual features ('ideal' invaders) or community characteristics (invasibility) still seems to be underestimated, although profoundly discussed by some authors (*e.g.*, Schroeder 1972; Trepl 1984, 1993; Orians 1986). Linkola's landmark study from 1916 includes one possibility of testing the relevance of the immigration conditions: a comparison between the flora of isolated settlements surrounded by forests with that of settlements in rural areas which were subjected to the cultural influence of humans for longer. Linkola (1916) found distinctly less alien species in isolated villages. These results indicate the limiting role of the immigration conditions.

Supposing that urban wastelands are less isolated from seed sources of alien species than rural wastelands, comparisons between these and between different urban areas may serve as another approach to elucidate the specifity of the urban environment in terms of immigration conditions. The results shown in Fig. 4 reveal a much higher percentage of alien species in various urban vegetation units compared to non-urban old field successional series and to rural wastelands (Rejmánek 1989; Prach 1994).

Focussing the proportion of alien *versus* native tree species as dominants of several urban wastelands in Berlin showed a broad variance in the area covered by *Robinia pseudoacacia* or by *Betula pendula*. If both the native *Betula* and the alien *Robinia* may become dominant during succession on similar sites (derelict railway areas), the presence or absence of both species can be explained neither by the resistence of preceding successional stages nor by limited resources. Both species can share the same resources, both are able to establish dense stands. This example stresses the role of immigration conditions which seem to be a decisive factor for the dominance of *Robinia* or *Betula* on urban wastelands in Berlin. Considering the virtually unlimited availability of *Betula* seeds, the potential dominance of *Robinia* is limited by the effective dispersal of this species. Conversely, the minor role of *Robinia* in non-urban wasteland succession (Prach 1994) may be due to limited seed sources, less effective dispersal, and, in colder areas, by the higher temperature requirements of *Robinia*.

Impatiens parviflora is a good example of a species whose spread from urban to rural sites is enhanced by improved dispersal conditions: it needed about 50 years to arrive in forests, and today, *Impatiens* is the most common alien species in central European forests (Trepl 1984). Although the invasion of non-urban forests or other rural habitats may not be exclusively a question of resistance or invasibility, but also a question of time. This hypothesis is supported by the higher presence of alien species in forests adjacent to urbanized areas (Asmus 1981; Moran 1984) and by the switch of species, formerly confined to settlements, to rural habitats (*e.g., Amaranthus retroflexus, Sisymbrium loeselii* in the district of Halle, Grosse 1987), and even to natural habitats (*e.g., Reynoutria* species. *Heracleum mantegazzianum*, and *Impatiens glandulifera*, Pyšek and Prach 1993 in the Czech Republic).

The role of alien species in succession

The data on the presence of alien species in the vegetation of Berlin shown in Fig. 4 confirm a decrease of alien species from early successional stages, usually dominated by annuals and monocarpic perennials with up to 60% aliens of all species, to perennial stages with 30–40% aliens. During the course of old field succession. Rejmánek (1989) found an exponential decline of aliens. It started from about 14% at the beginning and fell to less than 4% after about ten years (see also Osbornová *et al.* 1989). As opposed to these successional seres, the decrease in urban vegetation is far from exponential. Also, older successional stages may include a percentage of alien species which is more than twice that of the initial stages of non-urban successional series: a third of all species in the perennial ruderal vegetation of Berlin are aliens despite the dominance of grasses, herbs or even *Robinia pseudoacacia* which is the most frequent alien tree in the woody vegetation of urban wastelands. The oldest stems in urban *Robinia* stands are about 35 years old, and although the demographic results indicate changes in dominance, the switch to other prevailing tree species (presumably the shade tolerant *Acer platanoides* and *A. pseudoplatanus*) will again be a question of decades. Thus, considering both dominance and species richness (in % of all species), alien species may persist much longer than expected in urban successional series.

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Socioeconomics Drive Urban Plant Diversity

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Abstract Spatial variation in plant diversity has been attributed to heterogeneity in resource availability for many ecosystems. However, urbanization has resulted in entire landscapes that are now occupied by plant communities wholly created by humans, in which diversity may reflect social, economic, and cultural influences in addition to those recognized by traditional ecological theory. Here we use data from a probability-based survey to explore the variation in plant diversity across a large metropolitan area using spatial statistical analyses that incorporate biotic, abiotic, and human variables. Our prediction for the city was that land use, along with distance from urban center, would replace the dominantly geomorphic controls on spatial variation in plant diversity in the surrounding undeveloped Sonoran desert. However, in addition to elevation and current and former land use, family income and housing age best explained the observed variation in plant diversity across the city. We conclude that a functional relationship, which we term the "luxury effect," may link human resource abundance (wealth) and plant diversity in urban ecosystems. This connection may be influenced by education, institutional control, and culture, and merits further study.

Keywords: plant diversity · socioeconomics · wealth · phoenix

Cities represent extreme cases of human influence on ecosystem function [1–3] and provide unique opportunities for integrating humans into ecology [4–6]. Spatial variation in plant diversity has been attributed to heterogeneity in resource availability for many ecosystems [7]. However, urbanization has resulted in entire landscapes that are now occupied by plant communities wholly created by humans [8, 9]. Hence in and around cities, plant diversity may reflect social, economic, and cultural influences as well as those recognized by traditional ecological theory. To date, urban ecosystem characteristics have been evaluated largely in terms of the urban-to-rural gradient paradigm [10–13], with the simple linear gradient concept evolving to include gradients of disturbance, land-use intensity, and the polycentric, anisotropic nature of modern cities [14, 15]. However, such studies typically focus on patches of native vegetation within cities [16–19] rather than the full range of land-use types with their human-created plant communities that characterize much of the urban landscape. In addition, conceptual developments have identified the need to quantify gradients of resource availability and disturbance that integrate land use, legacy effects, socioeconomic status, and cultural differences, because these may mediate the human-environment interaction and influence resultant ecological conditions [20–24].

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We focused on spatial variation in plant diversity across the Central Arizona–Phoenix region because plant diversity is an important determinant of overall ecosystem biodiversity that influences the composition and abundance of associated biota [25]. We asked how well a suite of biotic, abiotic, and human-related variables explain site-to-site variations in richness of perennial plant genera (including both exotics and natives). The focus on perennials allowed us to eliminate seasonal variation that might have arisen over the course of the field survey. Spatial variation in plant diversity across unmanaged ecosystems at the regional scale in arid landscapes is typically related to geomorphic controls on water and nutrient supply [26, 27]. Our prediction for the Central Arizona-Phoenix region was that predominantly geomorphic (elevation) controls on diversity in undeveloped native Sonoran desert vegetation outside the city [28, 29] would be replaced by land use as the primary correlate of spatial variation in plant diversity across the city [30, 31] along with a possibility of some variation due to distance from urban center [15, 32].

Materials and Methods

Study Area and Sampling Design

We surveyed perennial plant generic richness across the entire Central Arizona-Phoenix long-term ecological research study area, which covers $6,400\,\mathrm{km}^2$, includes the rapidly expanding metropolitan Phoenix area, surrounding agricultural, and undeveloped native desert land, and contains 3 million people. Using a dual-density, randomized, tessellation-stratified design, we obtained a spatially dispersed, unbiased sample that allowed for maximum postdesign stratification. Probability-based sampling has been used in a number of national and regional studies [33, 34] because it ensures representative and unbiased characterization of ecological resources [35, 36], but application of this technique to a large urban area is unique. The dual-density tessellation-stratified design consisted of a grid of 4×4-km squares, on which was superimposed the "beltway" of existing and proposed freeways that encircles the main developed parts of the Phoenix metropolitan area. To allow for the much greater landscape heterogeneity within the developed metropolitan core [32], we used a sampling density inside/outside the developed urban core of 3:1, with a random sample taken in every square inside and one sample in every third square (chosen randomly) outside that area; this gave an equal number of ≈ 100 samples inside and 100 samples outside the developed urban core, giving a total sample size of 206. Access was negotiated successfully for all but eight sites; six of these plots were relocated to the nearest (within 100 m) similar accessible site. There were only two samples to which access was denied and no suitable surrogate could be found, giving a total of 204 sites surveyed.

Field Survey

A synoptic integrated field inventory was carried out between late February and early May 2000. The sampling unit used was a 30×30-m plot, in which all perennial woody vegetation was mapped, measured, and identified. Because the appearance of many horticultural cultivars can make accurate identification to the level of species difficult, plants were identified to species where possible or to genera where not but was standardized at the level of genus for subsequent analysis. However, in most study plots plant diversity on the genus level corresponds closely to diversity at the species level except in the case of two desert genera (*Ambrosia* and *Cylindropuntia*), of which there were two or three species present in ≈ 30 plots. Each site was also mapped to delineate the cover of the main surface types (e.g., asphalt, concrete, bare soil, and turf) on the 900-m^2 plot; these data were used to calculate the percent impervious surface cover at each site. Soil core samples (2.54 cm in diameter) were collected by using an impactor corer at four randomly determined locations within each plot at the 0- to 10-cm depth interval and combined to give a single sample from each survey site. These samples were sieved (2-mm mesh) seem homogenized, and a 10-g subsample was extracted with 2 M KCl solution; the filtered extract was acidified, stored at 4°C, and analyzed for nitrate-N (NO_3-N) on a Bran-Luebbe TrAAcs 800 autoanalyzer by using the calcium reduction method within 2 weeks as part of the chemical and physical characterization of the soils across the region. Land use at each of the 204 surveyed sites was classified according to a modified version of the Maricopa Association of Governments land-use classification scheme [37]. The five main land-use categories were urban ($n = 91$), desert ($n = 73$), agriculture ($n = 23$), transportation ($n = 6$), and a "mixed" class ($n = 11$). This classification differs from the Maricopa Association of Governments scheme in two main ways. First, managed "open" space within the urban area (i.e., irrigated parks, sports fields) was subsumed as a subcategory under the "urban" top-level category. Second, it differs by the addition of the mixed category for sites where more than one of the main land uses was present in the same survey plot.

Supplemental Data

Data from the field survey were supplemented with several additional key geographic and socioeconomic variables, which consisted of the latitude and longitude used to locate each site, defined by using the Universal Transverse Mercator North American Datum 27 projection. Also, elevation was included, as derived from the United States Geological Survey Digital Elevation Model for the region. Distance of each site from the urban center (defined as Central Avenue and Washington Street) and from the nearest major freeway were calculated by using ARCVIEW GIS. In addition, historic land-use analyses carried out for each survey point [38] were used to determine the number of years each site had been in agricultural use as well as to assign an indicator variable showing whether the site had ever been in agriculture. Three socioeconomic variables (median family income, median age of housing stock, and human population density) were taken from the U.S. Census of Population and Housing for the appropriate block group within which each survey point was located. Census block groups are drawn up to standardize the size of the human population within a certain range and approximate to a neighborhood. Within the developed urban core the average block group size was 5.3 km^2 and were mostly paired with sample sites: Only 17% of the urban sites fell in a block group with another site. Outside the developed core the human population is sparser and block groups are considerably larger (average 168 km^2); 86% of desert sites occurred in a block group with one or more other sites. Hence, block groups varied in size and were necessarily coarser than the size of the field survey plots but constitute the smallest unit for which socioeconomic information were readily available. Moreover, because clear human influences are only directly relevant for the urban model, where the Census block group data well represent the neighborhoods surrounding our survey sites, we do not consider the census block groups too large to invalidate our conclusions. Additional variables collected during the field survey but not used in the analyses presented here included physical (e.g., bulk density and soil textural analysis) and chemical (e.g., ammonium-N, total N, organic C, inorganic C, total C, and pH) characterization; sample collection to determine arthropod, pollen, and prokaryote diversity; mapping and measurement of all permanent, built structures; and documentary photographs in the four cardinal directions.

Statistical Analyses

A total of 13 variables were chosen to represent the main geophysical, geographic, and human characteristics of the study site and to have minimal colinearity (latitude, longitude, elevation, land use, distance from urban center, distance from nearest major freeway, impervious surface cover, soil nitrate-N concentration, number of years in agriculture, whether ever in agriculture, population density, median housing age, and median family income). Spatial variation in perennial plant generic richness was modeled across the whole region as well as separately for the undeveloped desert sites and developed urban sites (of which just over half were in private residential yards) by using a suite of spatial statistical techniques and these independent variables. The selection of the final model in each case primarily depended on the probability distribution used to model the response variable. Because plant diversity were count data (number of woody plant genera per plot), generalized linear modeling techniques were used with a log link and Poisson probability distribution. Pearson residuals revealed no spatial autocorrelation for data from all sites and from the urban-only sites. Generalized linear models were fit, but due to overdispersion, quasilikelihood techniques were applied. Deviance residuals were examined to investigate potential outliers. For the desert sites, the semivariogram of the Pearson residuals indicated spatial autocorrelation in the errors. A spherical semivariogram was fit to the empirical semivariogram of residuals obtained from the ordinary least squares model of the number of woody plant genera (which was square-root-transformed to stabilize the variance). Variance inflation factors between the independent variables used in these analyses never exceeded seven, and most were substantially less, indicating that colinearity is not a significant problem. When making inferences about the parameters, we used an $\alpha = 0.05$ significance level unless noted otherwise. We do not report r^2 values for the resultant models, because in generalized linear modeling there is no statistic that is the counterpart of the r^2 value in regression. Instead, the test statistic and significance level are given for each variable in each model to indicate the marginal effect, i.e., the strength of each the contribution of the variable given that all the other variables are included.

Results and Discussion

Spatial variation in plant diversity across the entire Central Arizona–Phoenix region (Fig. 1) was explained best by a combination of land use, elevation, median family income, and whether the site had ever been farmed (Table 1). The importance of land use (particularly urban and agricultural) as a determinant of overall plant diversity supports the expected relationship between patch type and ecological condition [30, 31]. Urban landscapes across the Phoenix metropolitan area have been established in the presence of a low-cost, abundant water supply, native plants having been replaced with imported exotic genera to create urban "oases" [39], a pattern typical of human settlement across the arid and semiarid regions of North America [40]. This seems to have increased total generic plant richness (γ diversity) across the region as a whole, as has been reported for urbanized regions elsewhere [41, 42]. However, local (α) diversity in perennial plant genera across the urban landscape is actually very similar to the native desert vegetation it replaced, albeit with considerably higher compositional turn-over (β diversity) between urban sites comprised of many exotic genera (Table 2). The median family income for the whole study area was $50,750/year. Plant diversity at sites in neighborhoods with incomes above this amount was on average twice that found in the landscapes of less wealthy areas (11 compared with 5 genera per plot). In addition, across the study region as a whole, sites that were formerly farmed had 43% fewer woody plant genera than locations that had never been cultivated.

Spatial variation in generic plant diversity among urban sites (which were an approximately equal mixture of private residential yards and other urban land uses, e.g., commercial, institutional, recreational, and industrial) was best predicted by human variables exclusively (Table 1). Plant diversity across the urban landscape was positively related to income in the surrounding area (Fig. 2). Wealthier neighborhoods are often located at higher elevations in metropolitan Phoenix, as in many

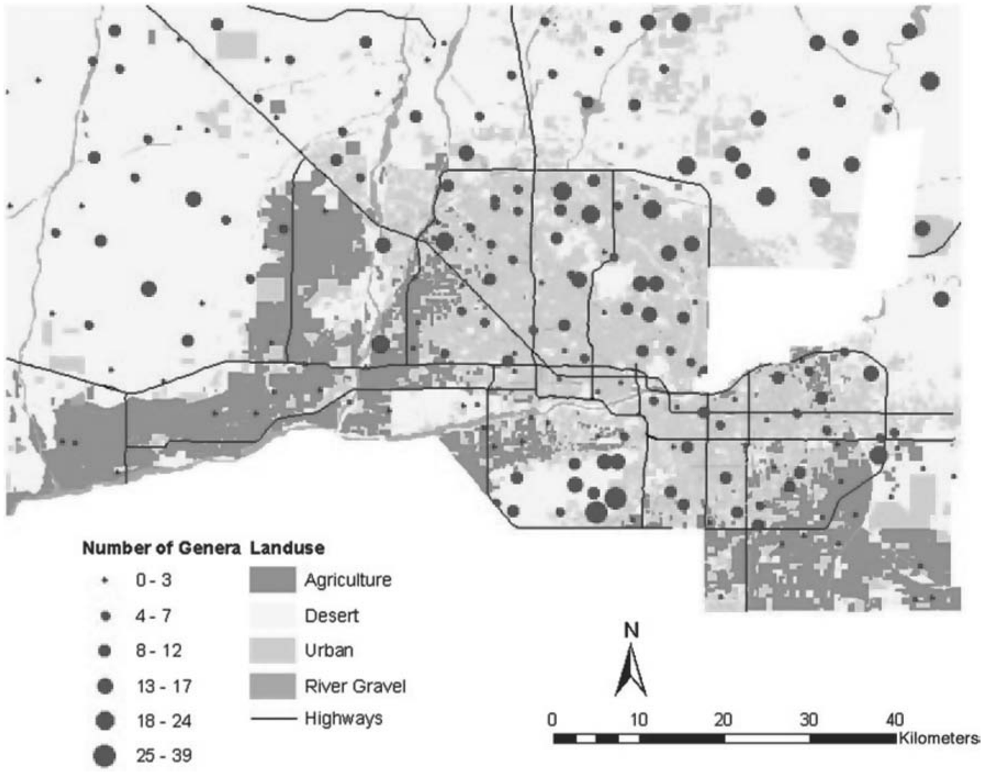

Fig. 1 Number of perennial plant genera per site across the Central Arizona-Phoenix long-term ecological research study area, superimposed on the major land-use categories. The study area encompasses the majority of Maricopa County, Arizona, but excludes Native American Indian reservation lands located in the east and to the south of the area shown

American cities [43], and there was some colinearity between elevation and income (variance inflation factor = 0.506 for the urban sites); hence, it was not possible to quantify the extent to which wealthier people create more diverse landscapes or simply acquire them. We term the relationship between wealth and plant diversity the "luxury effect," whereby as their economic wherewithal

Table 1 Best-fit models of plant diversity

	All sites	Urban	Desert
Model type	Spherical semivariogram	GLM	GLM
Data transformation	Log link	Log link	Square root
Spatial autocorrelation	Absent	Absent	Present
Predictor variables†	Land use	Family income (8.08)*	Elevation (7.59)***
	Urban (27.35)***	Median housing age (−6.65)*	Distance from urban center (−2.89)*
	Agriculture (10.21)*	Ever farmed (4.34)	Median housing age (2.50)
	Elevation (24.88)***		
	Family income (12.72)**		
	Ever farmed (6.04)		

Significant variables are listed in order of importance as judged by the level of significance denoted by asterisks (∗∗∗, $P < 0.0001$; ∗∗, $P < 0.001$; ∗, $P < 0.01$; no asterisk, $P < 0.05$). GLM, generalized linear modeling.

†The appropriate test statistics are shown in parentheses, consisting of χ^2 values for all sites and urban models and a T value for the desert model.

Table 2 Summary statistics for plant diversity (number of perennial genera per site)

	All sites ($n = 204$)	Urban ($n = 91$)	Desert ($n = 73$)	Agricultural ($n = 23$)	Transportation ($n = 6$)	Mixed ($n = 11$)
Total (γ diversity)	188	156	63	9	9	24
Mean (α diversity)	6.8	8.0	8.4	0.5	1.5	2.8
Median	5	6	7	0	1	1
SD	6.5	7.1	5.5	1.3	1.8	3.5
Range	0–39	0–39	1–24	0–5	0–4	0–8
β diversity	27.6	19.5	7.5	17.3	6.0	8.5

increases, humans occupy urban landscapes with higher plant diversity. Is this relationship a robust one given the rapidly changing landscape of the Central Arizona-Phoenix study region? For example, might median family incomes have changed differentially between block groups at time scales shorter than the lifetime of the woody perennials? We believe that this result is a robust one for two reasons. First, prior research has indicated that the composition and longevity of woody perennials in the Phoenix urban landscape is determined largely by human choices and landscape maintenance rather than by natural reproduction and mortality and thus tends to closely reflect current ownership preferences [44]. Second, a comparison of the income data with those from 10 years previously in the same census block groups, showed that income increased evenly across census tracts. Moreover the relationship between wealth and plant diversity would also appear to be similar to the link seen between socioeconomic status, species composition, and physical structure of vegetation in residential yards in several other cities [9, 45–47], leading us to suggest that the relationship between wealth and plant diversity may be characteristic of urban landscapes generally.

Plant diversity of developed urban sites was also related to age of housing, with higher diversity at sites with younger housing. We interpret this finding to reflect changes in landscape design, technology, and cultural values associated with more recent housing developments across the Phoenix metropolitan area. Before 1960 the predominant function of urban landscapes was to provide shading and evapotranspirational cooling, but the advent of widespread air conditioning and an increasing public interest in the conservation of native desert flora and water has resulted in a switch from broad expanses of grass and fast-growing broadleaf deciduous shade trees to xeriscapes with a more diverse suite of desert-adapted trees and shrubs [36, 48]. It may also be that wealthy people prefer newer housing, as indicated by weak colinearity between housing age and median family income (variance inflation factor = -0.379 for the urban sites) and that this also affects plant diversity in the urban landscape. Although the importance of air conditioning is specific to this region, we suggest

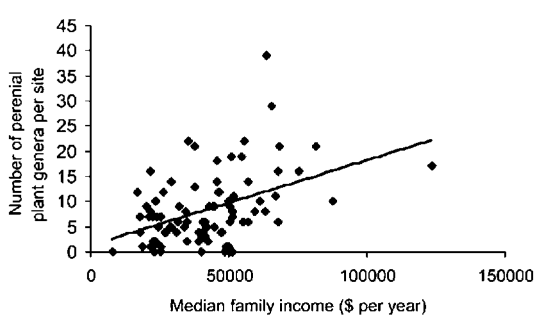

Fig. 2 Variation in the number of perennial plant genera with median family income (in dollars per year) from the U.S. Census of Population and Housing for the block group surrounding each survey size, at the urban sites. The regression line shown is an indicator of the linear relationship in the absence of the other predictor variables

that it represents an example of the underlying influence of technocultural controls, which may change among regions with different climatic regimes and cultural settings [49].

Elevation was a significant predictor of variation in plant diversity (Table 1) for both the undeveloped desert and the entire study region despite significant urbanization across the latter. The effect of elevation, along with the spatial autocorrelation in plant generic richness seen between neighboring samples at undeveloped desert sites up to a distance of 10 km apart, reflects the close correlation of plant communities to landform and geomorphic surfaces that previous studies have shown govern resource availability, in particular water and nutrient supply and hence underlying spatial variation in plant diversity at the regional scale in arid environments [26, 50]. After accounting for spatial relatedness and elevation, perennial plant richness increased with proximity to the urban center, an effect that seems to be mainly due to several large floristically diverse remnants of upland Sonoran desert on mountain preserves located near the metropolitan center. The latter is an example of deliberate human action and is the only instance of a simple linear urban-rural gradient effect seen in the data. In contrast to our findings for the urban sites, plant diversity was positively related to nearby housing age at the desert sites, with less floristically diverse sites tending to occur in block groups containing more recent housing developments, which we interpret as an inadvertent consequence of other factors (e.g., readily available land on flatter, lower elevation sites adjacent to lower diversity desert vegetation) on new housing development [51].

The spatial autocorrelation seen in plant diversity between desert sites was absent in the urban area. It would seem that in the intensively human-managed urban landscape there is an overall trend for plant diversity to be higher as nutrient and water availability increase due to fertilization and irrigation. This was reflected in higher generic richness (gamma diversity) as well as a much greater degree of compositional change from one site to another (β diversity) in the urban landscape compared with the desert (Table 2). However, spatial variation in the α diversity between individual sample sites would not seem to be a simple function of resource availability, as measured by soil NO_3-N, which made only a marginal contribution to predicting plant diversity in the models. Nor did the presence of impervious surface cover within the urban area seem to decrease plant diversity despite the accompanying reduction in growing surface area (due to the higher diversity per unit of cultivated surface area). However, urban sites that were formerly farmed had 57% fewer woody plant genera than locations that had never been cultivated. Complete removal of native vegetation before cultivation apparently depletes the local flora, creating a legacy effect that persists despite the subsequent creation of a vegetated urban landscape.

Our study is necessarily based on observational data, from which a causal relationship can only be implied and conclusions are necessarily limited to the scale at which the study was carried out. Different patterns may emerge with a higher spatial resolution or different grain size of the sample unit [52, 53]. Given these caveats, it would seem that the urban-rural gradient paradigm may provide a suitable model to explain the effects of urbanizing “disturbances” on intact native vegetation communities. However, our results within the urban matrix suggest that human maintenance is modifying the traditional resource availability–diversity relationships, in which a greater number of limiting nutrients allows coexistence of a greater number of competing species [54]. Humans remove resource limitations while simultaneously maintaining high diversity. We suggest that diversity in human-created habitats has less to do with variation in traditional limiting resources and more to do with human preferences for particular landscapes [44, 55, 56] along with the availability of financial resources to realize those landscapes. The resource–diversity relationship then becomes one of financial rather than natural resources, interacting with land use, legacy effects, and other sociocultural factors. Our findings, albeit from a preliminary synoptic survey of a single metropolitan area, suggest that current and former land use human resource abundance (i.e., wealth), and specific technocultural factors related to housing age are paramount. The positive relationship between plant diversity and wealth is particularly interesting, because it seems to mirror the well established link between quality of the social environment and socioeconomic status [57].

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With our synoptic approach it is not possible to capture fully the interactions between the biophysical and socioeconomic variables, because people not only create landscapes with enhanced diversity but also may prefer to live where such landscapes already exist. Repeating our survey at regular intervals may allow us to determine the interactions between these factors and investigate how changes in income levels translate into changes in plant diversity over time. However, more detailed studies of the interplay among factors such as educational level, culture, institutional influences, and controls will also clearly be needed to understand fully the mechanisms determining how human choices drive urban plant diversity.

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Fauna of the Big City – Estimating Species Richness and Abundance in Warsaw, Poland

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Keywords: biodiversity · insects · birds · mammals · reptiles · species diversity · Warsaw

Introduction

While we know much about the diversity of some animal groups in cities - e.g. birds of European cities [Kelcey & Rheinwald 2005], Lepidoptera of London (Plant 1987, 1993), bats of Vienna (Spitzenberger 1990), insects of Rome (Zapparoli 1997), nowhere has there been made an integrated assessment of all wild, multicellular animals living within the administrative boundaries of a city. Therefore, I summarize here the extensive work carried out by the Institute of Zoology of the Polish Academy of Sciences in Warsaw to document the urban fauna of Warsaw, Poland.

Warsaw is a city of 1.7 million people covering an area of 517 km^2 in central Poland. Vegetated habitats (vegetation in housing estates, parks, allotment gardens, cemeteries, periphery forest parks, green open areas) comprise $\sim 28\%$ of this area, and there are still (rapidly decreasing) wide patches of farmland. The river Vistula, with its 28 km-long green belt, crosses the city. The fauna of Warsaw has been studied extensively (Czechowski 1990, Luniak & Pisarski 1994). From 1974–1990 a research project was carried out by the Institute of Zoology into terrestrial invertebrates of Warsaw and the invertebrate fauna of the Warsaw region (specifically Mazowsze, Central Poland). More than 90 families and orders of animals were studied by about 36 specialists. Invertebrate communities were studied in tree canopies, shrubs, herbaceous ground cover, and in the soil of urban and seminatural green habitats using standard methods (Czechowski & Pisarski 1981). Major results were summarized by Pisarski (1982, 1990) and in five volumes: "Species composition and origin of the fauna of Warsaw" – Part 1 (Czechowski & Pisarski 1981), Parts 2 and 3 (Czechowski et al. 1982a, 1982b), and "Structure of the fauna of Warsaw; effects of the urban pressure on animal communities" (Czechowski & Pisarski 1986, 1987). In addition to these invertebrate surveys, Luniak et al. (2001) and Nowicki (2001) developed an atlas of Warsaw's birds based on field studies carried out by their team of 50 observers in 1986–1990 and 1999–2000. Mammals of Warsaw were described in the review by Luniak & Nowicki (1990), as were bats (Lesinski & Fuszera, 2001), and amphibians (Mazgajska, 1996, 1998).

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How many species live in a big city?

These studies identified $\sim$ 3800 species of terrestrial invertebrates (Chudzicka & Skibinska 1994) and $\sim$ 320 species of vertebrates as living in the administration area of Warsaw during the past few decades. This estimate for vertebrates is likely accurate. But the list of invertebrate fauna is fairly incomplete and it is constantly being increased by new investigations, e.g. recent studies on butterflies and moths (Winiarska 2002, 2003, 2004) added several tens of species to the list. Aquatic invertebrates, invertebrates living in house interiors and other structures (e.g., sewer systems), and parasites of wild animals have not yet been assessed and cannot be included. Considering these omissions, the diversity of multicellular animals living in Warsaw likely approaches 6–7 thousand species.

In the European literature there are very few data which could be compared with this estimate. In Lodz (city of 800,000 people in Central Poland), 3059 animal species (including the group described by these authors as "Protozoa") have been recorded since the 1920s, and the total number of species is estimated at 10-14,000 (Markowski et al. 2004). In Rome, 5151 insect species have been recorded from about 650 urban sites "... from the first half of 19th century up to 1996" (Zapparoli 1997). Klausnitzer (1988, 1993) made an extensive ecological and taxonomical review of urban fauna and estimated (1988) that cities of Central Europe (together) could be inhabited by about 18,000 animal species.

Even if these estimates are exaggerated, or my calculation for Warsaw is low, these results suggest that richness of the invertebrate fauna in big European cities exceeds several thousand species and that the knowledge concerning this important component of urban wildlife is poor.

Our knowledge concerning vertebrates of cities is much better. Among about 320 species of this group which have been recently recorded from Warsaw there are about 40 species of mammals, $\sim$247 species of birds ($\sim$187 regularly occurring, $\sim$131 – breeding), 5 reptiles, 11 amphibians, and $\sim$ 30 fishes. Data from cities like Berlin (Sukopp 1990, Otto & Witt 2002), Moscow (Risin 1998), and Lodz (Markowski et al. 2004) indicate that each city contains 35–46 species of mammals, 75–160 species of birds (Kelcey & Rheinwald 2005), 3–5 species of reptiles, 11 species of amphibians, and 30–42 species of fishes. (in Łodz.14 species as the city has no large water bodies). Thus Warsaw seems to be similar to these cities.

Relative species richness

The total number of about 4100 species in Warsaw is about 12% of all "wild" animal species ($\sim$ 35,400 species) which have been recently identified for all of Poland (Chudzicka & Skibinska 2003). For Łodz (Markowski et al. 2004), 3059 animal species were recorded since the 1920s, or about 9% of the total fauna of Poland. If we use the above sources and compare only vertebrates (a relatively well recognized group for both cities), we find that Warsaw has $\sim$ 320 spp, and Lodz $\sim$206 spp: all of Poland has 659–690 spp, so the proportions are 48–50% for Warsaw and 30–31% for Lodz.

Urban animal communities usually have only a small proportion of the species found in natural habitats in nearby non-urban areas (Luniak 2004, Marzluff 2005). This is the case for Warsaw's birds. The entire city includes 65% of the region's avifaunal species, but only 37% of them are found within the most developed parts of the city (Luniak et al. 2001, Nowicki 2001). The invertebrate fauna of Warsaw relative to the regional fauna is increasingly impoverished as one moves along a gradient of increasing urbanization. Pisarski (1982, 1990) documented 3534 species in the region, but the sources above only found 2005 species (57% of the regional pool) in green suburbs of Warsaw, 31% (1109 spp) in urban parks, and 14% (489 spp) in small green patches in the city center. This general decrease in species richness with increasing urbanization is pronounced in almost all taxonomic groups in Warsaw (Table 1).

Table 1 Species richness along Warsaw's gradient of urbanization. Regional data are from the semi-natural habitats in the Mazowsze region where Warsaw is situated. "Warsaw" includes various habitats in the city and suburbs of Warsaw. "Parks" are urban parks. "City" includes the green areas in the city center such as courtyards, lawns, and street trees

Animal taxa	Region	Warsaw	Parks	City
Terrestrial snails Gastropoda (terrestra)	55	32	25	5
Earthworms Lumbricidae	12	15	8	8
Spiders Aranei	424	254	134	43
Carabids Carabidae	323	276	96	44
Ladybirds Coccinellidae	58	51	28	14
Neuropterids Neuriopterida	56	43	35	14
Flies Diptera, Tabanomorpha	131	95	29	10
Ants Formicidae	43	37	21	11
Wasps Vespidae and Eumenidae	42	14	10	10
Leafhoppers Auchenorryncha	270	171	97	43
Noctuid moths Noctuidae	309	270	49	90
Mosquitos Culicidae	35	26	13	7
Springtails Collembola*	61	75	57	55
Percentage spp of 13 taxa above	100%	75%	33%	19%
Total spp of all taxa studied in Warsaw	3534	2005	1109	489
Percentage of all taxa studied in Warsaw	100%	57%	31%	14%

* Data from Sterzynska (1990).

Invisible urban fauna

A significant, but poorly known aspect of the urban fauna is the invertebrates of the soil and litter. Investigations described above demonstrated that $1\,\mathrm{m}^2$ of soil from Warsaw park lawns hold an average of several thousand (in some cases even 40,000) individual invertebrates. Among the most numerous animal groups recorded were soil mites Acari ($\sim 20,000$), enchytreids Enchytreidae (5–25,000), springtails Collembola (2–7,500), ants Formicoidea (~ 500), earthworms Lumbricidae (60–160), click beetles Elateridae (40–100), and many (above one hundred) insect larvae, particularly aphids Aphidodea, beetles Staphylinidae and flies Diptera, also spiders Aranei and snails Gastropoda. This incredible biomass does not include many thousand/m^2 of nematodes (Nematoda) which live "free" in the soil or inside plant roots. Density was lower in $1\,\mathrm{m}^2$ of ground and leaf litter, but included several hundred invertebrates. Among them were mainly ants Formicoidea (~ 200), various groups of beetles Coleoptera (~ 70), snails Gastropoda, and spiders *Aranei* (some tens of each group).

Shrubs and tree canopies also harbored significant invertebrate communities. Shrub invertebrates were dominated by insects (flies Diptera, leafhoppers Homoptera Auchenorryncha, and aphids Aphidodea). In $1\,\mathrm{m}^3$ of urban park tree canopy live 2–3 thousand of invertebrate animals. Most numerous among them were aphids Aphidodea, flies Diptera and leafhoppers Homoptera, Auchenorryncha, and Hymenoptera.

The ecological role of this "unnoticeable" and uncountable part of the urban fauna should be very significant by their "volume" and also by their variety (Table 1). These animals live inside the soil, on the ground in grass and in leaf litter, in all strata of vegetation, inside tree trunks, in water bodies, even inside technical structures like interiors of buildings and sanitary installations. They are indispensable for all functions and for all links of the energy/matter flow in the ecosystem. They stimulate soil processes, they are consumers of plants and they are predators limiting these consumers. They are the basic food of many vertebrates, but in general they are poorly known (e.g., Whiteley 1994).

The studies above revealed that the abundance of invertebrate communities in urban areas could be relatively high, and – surprisingly perhaps – in many cases it exceeds that in rural habitats. This often results because of very abundant species that sometimes occur en masse – super-dominants. For example, among groups which usually show much higher urban (vs rural) dominance are ants Formicoidea, wasps Vespidae, and leafhoppers Auchenorryncha and Sphecidae (Chudzicka & Skibinska 1994). In the soil, however dominance by single species in urban environments is usually lower than in semi-natural ones. For example, the density of springtails Collembola on $1\,\mathrm{m}^2$ of urban lawn averaged $\sim$ 4000 individuals, but abundance in agricultural meadows was $\sim$ 9000 (Sterzynska 1990).

Avian abundance in the city

Among vertebrates, the most abundant and exact data is for the birds. The total population of the breeding avifauna of Warsaw was estimated to be 150–350,000 breeding territories. This translates into about 300–700 pairs/km^2 in the overall municipal area of the city. This density is similar to comparable data from Berlin (360-670,000 territories, $\sim$400–700 pairs/km^2, Otto & Witt 2002) and Hamburg (410, 000 territories, $\sim$540 pairs/km^2; Mitschke & Baumung 2001). The mean population density of birds in the highly urbanized areas of inner Warsaw (from a sample of $52\,\mathrm{km}^2$) was estimated at 830–1590 pairs ($\sim$435 kg of biomass) per km^2 in the breeding season and 2.5–4.5 thousand individuals ($\sim$964 kg) per km^2 in winter (Nowicki 2001). Such high concentrations of birds in urban areas occurs throughout Europe (e.g. Nourteva 1971, Tomialojc & Profus 1977, Bezzel 1982, Sasvari 1990, Tomialojc 1998), but not in rural habitats of the European temperate zone.

General statements

1. Total number of species of the fauna of Warsaw, in all habitats within its municipal area, is estimated to be 6–7 thousand, including $\sim$320 species of vertebrates. This is typical for big European cities, whose fauna ranges between 5–10 thousand species including 300–400 vertebrates.
2. Species richness of a big city's fauna decreases considerably along the gradient of urbanization from rural areas in the region towards the city center.
3. The population density of the urban fauna is considerable and often reflects superabundant single species. In some groups (e.g. birds, wasps, ants) that density is much higher than in rural areas.
4. Only a very small part of this rich animal world, mainly the few terrestrial vertebrates, is noticeable to city dwellers.

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Island Biogeography for an Urbanizing World: How Extinction and Colonization May Determine Biological Diversity in Human-Dominated Landscapes

John M. Marzluff

Abstract Urbanization is increasing worldwide with potentially important implications to biological diversity. I show that bird diversity is responsive to the reduction of forest cover associated with urbanization in the Seattle, WA, USA metropolitan area. Bird diversity peaks at intermediate levels of human settlement primarily because of the colonization of intermediately disturbed forests by early successional, native species. Extinction of native forest birds and colonization of settlements by synanthropic birds have lesser effects on the overall pattern of avian diversity with respect to the level of urbanization. However, extinction increases linearly with loss of forest and colonization by synanthropic species decreases curvilinearly with reduction of urbanization. These findings have biological, theoretical, and practical implications. Biologically, intermediate disturbance appears to drive diversity by increasing the heterogeneity of the local land cover. Theoretically, I present a graphical model and use it to derive testable hypotheses about how extinction and colonization are affected by urbanization to determine local diversity. Practically, maintaining high local diversity without reducing regional or global diversity will require planning so that the same landscapes are not promulgated everywhere. This will require cooperation among a diverse group of planners, ecologists, policy makers, home owners, educators, and activists.

Keywords: biological diversity · birds · colonization · disturbance · extinction · urban forests

Introduction

Increasing human population and associated industrialization has swelled our cities. In 1900 only 10% of humans lived in cities, but by 2000 nearly 50% did so, and 60% are expected to do so by 2030 (Sadik, 1999). Depending on economics, social preferences, and land use policies, the growth of urban populations causes cities, and even more profoundly their suburbs, to spread across large expanses of former agricultural and natural lands (Robinson *et al.*, 2005). The world-wide extent of sprawling settlement is obvious in nighttime images of Earth from space (Elvidge *et al.*, 1997). These images reveal that substantial portions of the north temperate zone are heavily settled, most ice-free coastlines are settled, our most fertile lands are quickly being developed, and overall about 3% of Earth's land area is urban (Lawrence *et al.*, 2002; Imhoff *et al.*, 2004). As human populations grow, the extent of urbanization will increase. But at what cost to biological diversity?

Human settlement has profound effects on the flora and fauna of a region. Settlements reduce native vegetation, sever connections among remaining native vegetation patches, and perforate large

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patches (Matlack, 1993; Robinson *et al.*, 2005; Hansen *et al.* in press). Associated horticultural activities introduce exotics, degrade and simplify ground cover, and homogenize regional plant diversity (Reichard and White, 2001). Many animals, especially those sensitive to predation, competition, and disturbance decline in response to these changes (Marzluff, 2001). The effects of urbanization are longer lasting and more extreme than those accompanying other anthropogenic land uses (Marzluff and Ewing, 2001), which may be why urbanization is a leading cause of species endangerment in the US (Czech and Krausman, 1997). However, for all its apparent evil, settlement benefits some wildlife by reducing predation, ameliorating climate, increasing available water, supplementing food resources, providing new nest sites, and increasing edge and vegetative diversity (Marzluff, 2001).

The varied influence of settlement on plant and animal populations affects emergent properties of communities, such as their biomass or diversity. Because some species that benefit from settlement often attain large population sizes, animal densities often increase with human settlement (DeGraaf and Wentworth, 1986; Blair, 1996; Sewell and Catterall, 1998; Donnelly and Marzluff, 2004a). Depending on the scale of inquiry, community diversity may increase, decrease, or remain unchanged in response to human activities such as urbanization (Olden and Poff, 2003; Olden *et al.*, 2004). Globally, diversity is decreasing across taxonomic groups (Pimm, 2001; Wilson, 2002; Sax and Gaines, 2003). Locally, however, diversity often increases as native species are joined by tolerant, cosmopolitan, and often exotic species (as summarized for fish, reptiles, mammals, and invertebrates by Sax and Gaines 2003). As native and exotic species interact through time, local diversity may decrease if exotic species drive native species to extinction (Scott and Helfman 2001). But if tolerant species simply replace sensitive ones, local diversity will remain unchanged for substantial lengths of time (Parody *et al.*, 2001). If land transformation increases habitat heterogeneity, and invading species do not dramatically consume or compete with native species, high local diversity may be maintained (Leopold, 1933; Blair, 1996, 2004; Porter *et al.*, 2001). This appears to be the case in Australian shrublands and Arizona grasslands, where bird communities were richest in suburban and exurban settlements, respectively (Sewell and Catterall, 1998; Zach Jones and Carl Bock, personal communication, 2004).

Understanding the processes controlling biological diversity in urbanizing landscapes may allow us to explain enigmatic results and anticipate the changes in diversity that accompany human settlement. MacArthur and Wilson (1963, 1967) identified the key processes governing biological diversity nearly four decades ago. In their models and empirical data, and in the rich literature that they spawned (notably Brown and Kodric-Brown, 1977; Lomolino 1999), diversity of an island or area was simply the balance between colonization and extinction. Colonization and extinction have historically responded to the demographic and life history characteristics of organisms, most notably their survival, reproduction, and dispersal (Marzluff and Dial, 1991; Marzluff *et al.*, 2000; Bolger, 2001). In today's human-dominated world, colonization and extinction are affected by direct and indirect human action. Extinction now occurs in response to land cover change or new selective forces applied by novel climatic regimes, predators, diseases, and competitors (Scott and Helfman, 2001; Sax and Gaines, 2003). Colonization is greatly accentuated as people remove barriers to dispersal, juxtapose a variety of land covers, and directly or indirectly introduce species outside of their native ranges (Kühn *et al.*, 2004). In a human-dominated world, diversity still emerges as the balance between extinction and colonization, but the amount, identity, and actions of invading species take on greater prominence (Olden and Poff, 2003, 2004).

In this paper I begin the process of understanding colonization and extinction in an urbanizing landscape. I extend the work of Donnelly and Blewett (Donnelly 2002; Donnelly and Marzluff, 2004a; Blewett and Marzluff, 2005) on bird communities in the Seattle, WA, USA metropolitan area. I determine the relative importance of colonization versus extinction to bird communities in Seattle and extend this observation to a general theory of avian diversity in urban habitat islands. By 'colonization,' I mean local additions to the avifauna by immigration and invasion of species not typical of local coniferous forests. By 'extinction,' I mean local extirpation (Olden and Poff, 2003; Sax and Gaines, 2003). I use my theory to suggest general planning and management considerations for those interested in maintaining biological diversity in urbanizing landscapes.

Methods

Study area

The Seattle metropolitan area (47°, 40' N; 122°, 20' W) is located within the Western Hemlock (*Tsuga heterophylla*) Zone of the Pacific Northwest (Franklin and Dyrness, 1988), where forest cover was dominant before European settlement (Booth, 1991). The metropolitan area inhabited by nearly 3 million people is composed of a large business district on the east side of the Puget Sound flanked by spawling residential developments and satellite business districts east into the Cascade Mountain foothills (Fig. 1).

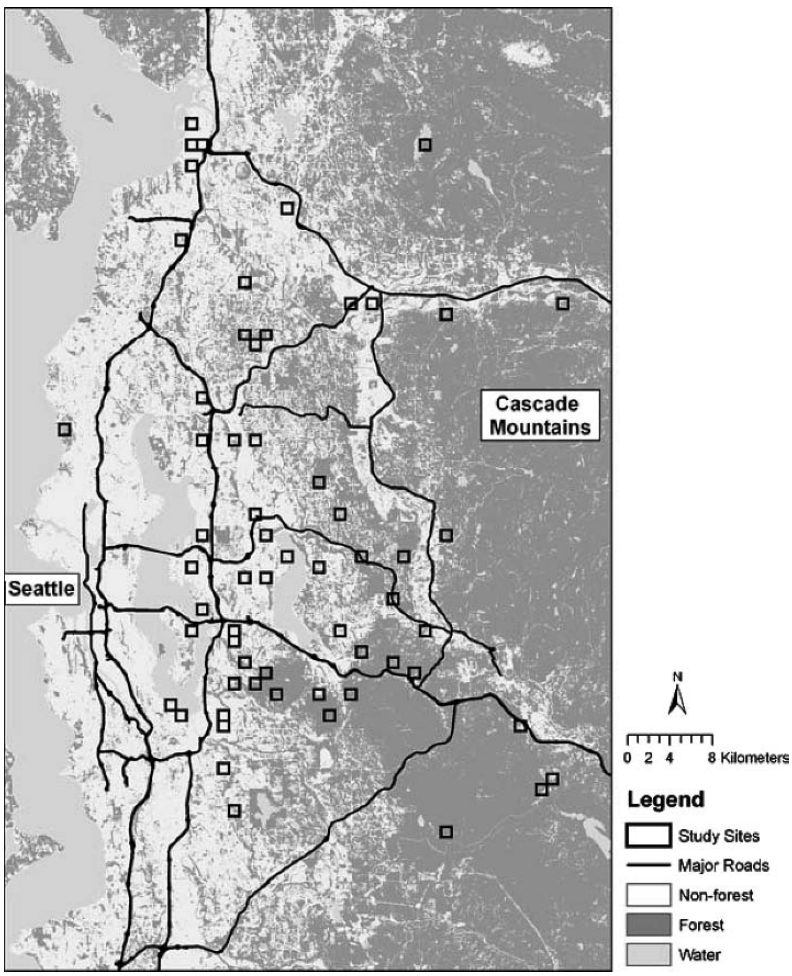

Fig. 1 Map of the study area indicating forest and urban land cover, location of study sites (open squares), and size of study landscapes ($1\,\mathrm{Km}^2$)

Site selection

I selected 61 sites/landscapes representing the range and combination of habitat quantity (percent urban landcover) and habitat pattern (mean urban patch size, forest aggregation) by stratified random sampling. Details are in Donnelly (2002) and summarized here. I chose $1\,\mathrm{km}^2$ as the standard landscape size because it was comparable to the size of typical residential developments and territories of common nest predators like the American crow (*Corvus brachyrhynchos*; Marzluff *et al.*, 2001a). I quantified habitat quantity and pattern from a classified 1998 LANDSAT satellite image (Botsford, 2000). Forest was $\geq 70\%$ trees and $< 20\%$ impervious surface (e.g., pavement). Urban forest was $\geq 25\%$ trees and 20–60% impervious surface. Urban was $\geq 60\%$ impervious surface. Other was $\geq 75\%$ open water or bare soil. Throughout this paper when I refer to "forest" I am only referring to the "forest" category, not the "urban forest" category. Within each landscape, I estimated the representation of each landcover class and the size of urban patches (i.e., continuous urban areas) using the Geographic Resource Analysis Support System and the r.le add-on programs (Baker, 1997; Alberti *et al.*, 2001) and forest connectivity using the Aggregation Index produced by Fragstats 3.1 (McGarigal *et al.*, 2002). Once I identified a set of landscapes below 1000 m in elevation that represented a range of landcover composition and connectivity using the remotely sensed data, fieldworkers visited sites to select those that were predominately single family residential, similar in forest structure and composition, and without extensive agricultural activity.

I selected landscapes to span the available range of variables in the study area. Urban landcover ranged from 4–77% with a mean ($\pm$ S.E.) of 36 ± 3. Urban patch size ranged from 0–89 ha with a mean of 12 ± 3. Forest aggregation ranged from 0–0.96 with a mean of 0.70 ± 0.03. I could not include some combinations of variables, such as low percent urban landcover/high mean urban patch size, because they did not exist in the metropolitan region.

Bird surveys

Each study site was surveyed for birds in a single year (1999, 2000, 2001, or 2004; see Donnelly and Marzluff, 2004a, b for details). Measured diversity of a site increases with additional years of study, but in the relatively species-poor bird communities of western Washington this increase is insufficient to obstruct the strong response of bird diversity to disturbances as large as urbanization. A survey consists of four visits to each site to count birds during the breeding season (roughly April to August). During each visit we recorded all birds detected in or just above the canopy by sight or sound during 10 min within a fixed area (50 m from the count location; Ralph *et al.*, 1993). We surveyed eight points within each landscape (2 in forest fragments and 6 in settlements) during each of our four visits. All points were in forest at our seven forested reserve sites. We allocated more effort to settlement than forest because a previous study in the same region indicated that birds and vegetation were more variable in settlements (Donnelly, 2002). All points were $> 150\,\mathrm{m}$ apart, with the exception of a few forest points where we maximized separation within the only forest fragment that existed on the landscape. We did not conduct more than four surveys per landscape because < 2 new species are detected in forests with increased effort (Donnelly, 2002). We did not consider migrant birds that did not breed in our study area, birds that bred primarily in riparian corridors, birds that bred in low density below 1000 m, or birds that ranged over large areas because our survey technique was unable to assess how these birds were using the field sites.

Surveyed birds were classified into three guilds (Table 1). Native forest birds ($n = 19$) were those routinely found in large, second growth, coniferous forests in the region. Synanthropic birds ($n = 9$) were those dependent on human settlement (Johnston, 2001). Early successional species ($n = 30$)

Table 1 Songbirds surveyed in the urbanizing landscapes in and around Seattle, WA, USA categorized into three guilds relevant to urban ecosystems. Native forest species are routinely found in the mature, second growth, coniferous forests that form the natural vegetative matrix of the region. Syanthropic species obtain critical resources from humans and are common inhabitants of human settlements. Early successional species are native species that are rare in mature coniferous forests, but common in fields, meadows, regenerating forests, edges, grasslands, ponds, and deciduous, riparian areas

Native forest	Synanthropic	Early successional
American robin *Turdus migratorius*	American crow *Corvus brachyrhynchos*	American goldfinch *Carduelis tristis*
Black-throated gray warbler *Dendroica nigrescens*	Anna's hummingbird *Calypte anna*	Band-tailed pigeon *Columba fasciata*
Brown creeper *Certhia americana*	Barn swallow *Hirundo rustica*	Bank swallow *Riparia riparia*
Chestnut-backed chickadee *Poecile rufescens*	Brewer's blackbird *Euphagus cyanocephalus*	Bewick's wren *Thryomanes bewickii*
Dark-eyed junco *Junco hyemalis*	Brown-headed cowbird *Molothrus ater*	Black-capped chickadee *Poecile atricapillus*
Downy woodpecker *Picoides pubescens*	European starling *Sturnus vulgaris*	Black-headed grosbeak *Pheucticus melanocephalus*
Golden-crowned kinglet *Regulus satrapa*	House finch *Carpodacus mexicanus*	Bushtit *Psaltriparus minimus*
Hairy woodpecker *Picoides villosus*	House sparrow *Passer domesticus*	Cassin's vireo *Vireo cassinii*
Hammond's flycatcher *Empidonax hammondii*	Rock pigeon *Columba livia*	Cedar waxwing *Bombycilla cedrorum*
Hermit thrush *Catharus guttatus*		Common yellowthroat *Geothlypis trichas*
Hutton's vireo *Vireo huttoni*		Killdeer *Charadrius vociferus*
Pacific-slope flycatcher *Empidonax difficilis*		MacGillivray's warbler *Oporornis tolmiei*
Red-breasted nuthatch *Sitta canadensis*		Northern flicker *Colaptes auratus*
Spotted towhee *Pipilo maculatus*		Olive-sided flycatcher *Contopus cooperi*
Steller's jay *Cyanocitta stelleri*		Orange-crowned warbler *Vermivora celata*
Swainson's thrush *Catharus ustulatus*		Pine siskin *Carduelis pinus*
Western tanager *Piranga ludoviciana*		Purple finch *Carpodacus purpureus*
Wilson's warbler *Wilsonia pusilla*		Red crossbill *Loxia curvirostra*
Winter wren *Troglodytes troglodytes*		Red-winged blackbird *Agelaius phoeniceus*
		Northern Rough-winged swallow *Stelgidopteryx serripennis*
		Rufous hummingbird *Selasphorus rufus*
		Savannah sparrow *Passerculus sandwichensis*
		Song sparrow *Melospize melodia*
		Tree swallow *Tachycineta bicolor*
		Violet-green swallow *Tachycineta thalassina*

Table 1 (continued)

Native forest	Synanthropic	Early successional
		Warbling vireo *Vireo gilvus* White-crowned sparrow *Zonotrichia leucophrys* Western wood pewee *Contopus sordidulus* Willow flycatcher *Empidonax traillii* Yellow-rumped warbler *Dendroica coronata*

were a diverse suite of birds that are found in greatest abundance in meadows, fields, edges, young forests, and deciduous, riparian woodlands.

Statistical analyses

I completed all statistical analyses using the Statistical Package for Social Sciences 10.1.3 (2001), except non-linear regression where I used Sigma Plot 8.0. To meet the assumptions of parametric tests, I transformed percentages (arcsine square root) prior to analysis.

Results

Empirical bird diversity in Seattle

The number of bird species in a 1 km^2 landscape comprised of single family housing and fragments of native coniferous forest was strongly correlated with the percentage of native forest (Fig. 2). This was not a linear relationship, but a significantly quadratic one (Richness $= 18.1 + 43.6$ (%forest) -41.2 $(\%\text{forest})^2$; $F_{2,60} = 19.8$, $P < 0.0001$). Over a third of the variation in bird species richness was accounted for by this relationship ($R^2_{\text{adjusted}} = 38.5\%$). Richness peaked at 50–60% forest in the landscape.

Bird species richness is determined by the balance between retention of native forest birds and the gain of synanthropic and early successional species (Fig. 3). Loss of native forest birds was linearly

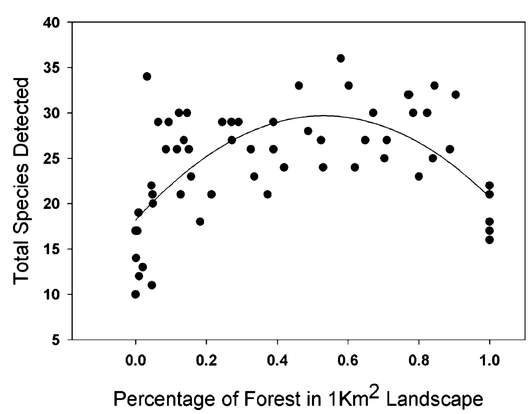

Fig. 2 Change in avian diversity with progressively less settlement (more forest). Each point is a study site; control sites ($n = 7$) have 100% forest in their landscapes

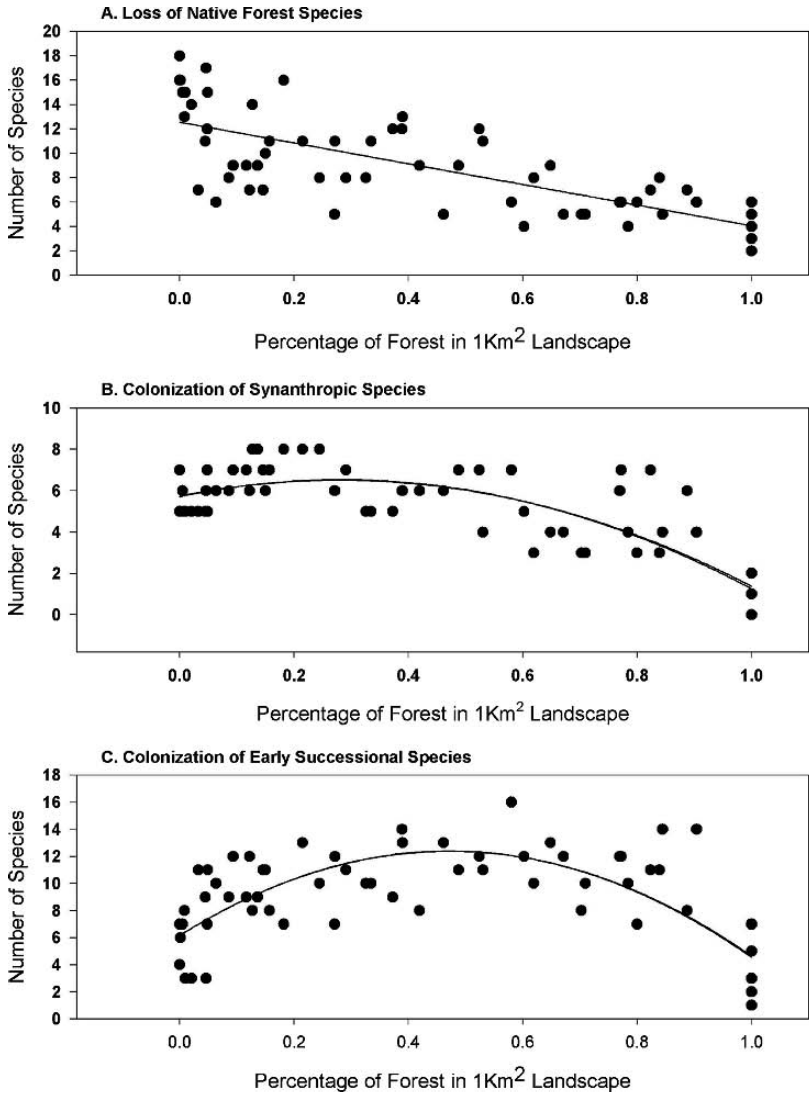

Fig. 3 Reponses of three guilds of birds to reductions in settlement (increased forest). Individual species in each guild are listed in Table 1

related to loss of forest ($R^2_{\text{adjusted}} = 53.8$; $F_{1,60} = 70.9$; $P < 0.0001$). Colonization of synanthropic species declined quadradically with gain in forest ($R^2_{\text{adjusted}} = 63.1$; $F_{2,60} = 52.3$; $P < 0.0001$). Colonization of early successional species peaked in landscapes with 50–60% forest ($R^2_{\text{adjusted}} = 48.4$; $F_{2,60} = 29.1$; $P < 0.0001$). Thus, bird communities in landscapes 50 – 60% forest have high species diversity because they support rich mixes of native forest birds, early successional species that use grasslands and forest openings, and synanthropic species that benefit from people. Bird communities in more urban areas are impoverished because only about eight synanthropic species and fewer than five native species exist in mostly paved landscapes. Likewise, communities in mostly forested areas are impoverished because they are composed nearly entirely of 15 native forest birds.

Colonization of suburbs by early successional species, most of which are native to the region (Table 1), appears most influential to the relationship between bird diversity and human settlement (Fig. 4). The gain in early successional species, and to a lesser extent synanthropic species, exceeded

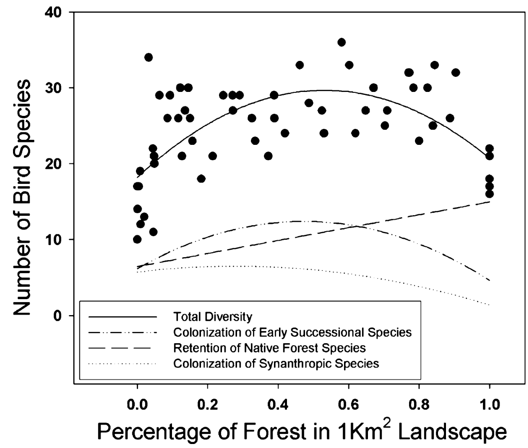

Fig. 4 Influence of three guilds of birds on total avian diversity. Solid curve and points are the change in diversity from figure 2. Dashed and dotted lines represent three guilds and can be summed to equal the total diversity at any point on the gradient of urbanization (amount of forest)

the loss of native forest species at all sites where forest cover was between 20 and 90% (Fig. 5). Extinction outpaced colonization at the most urban and most wildland (forested) sites.

Development of a General Theory

Extinction and colonization of human settlements by mobile species like birds appears to vary with the proportion of natural vegetation remaining in the landscape. Moreover, colonization can be different for synanthropic species and native species characteristic of seral stages created or simulated by settlement. Diversity at any point along a gradient of urbanization, represented in my model by the amount of natural vegetation in a landscape, therefore equals:

$$(C_{\mathrm{syn}} + C_{\mathrm{ser}}) - E,$$

where, C_{syn} is the number of synanthropic species colonizing the landscape, C_{ser} is the number of species colonizing the landscape from newly created seral stages, and E is extinction of native seral stage species from the landscape.

Here I consider some possible relationships between extinction, colonization, and the amount of remaining natural vegetation in an attempt to pose testable hypotheses useful in future investigations of biological diversity in urbanizing landscapes.

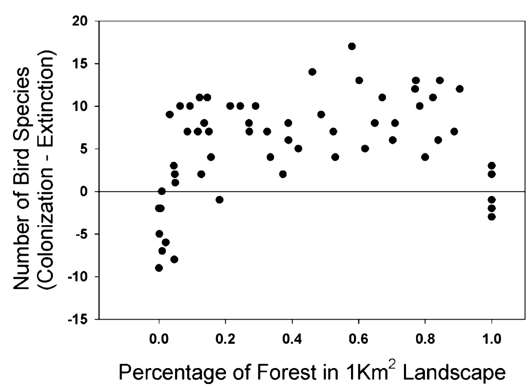

Fig. 5 Difference in the amount of colonization by synanthropic and early successional species and extinction of native forest birds in communities as a function of settlement (reduction in forest). Values above the horizontal line indicate that colonization was greater than extinction. Values below the horizontal line indicate that extinction was greater than colonization

Extinction will certainly increase as more native vegetation is replaced by settlement, but the form of this relationship and magnitude of the loss may vary with aspects of the remaining vegetation. Consideration of the variation in human use and the arrangement of the natural vegetation that remains (Fig. 6(A)) suggest two testable hypotheses:

(1) Where remaining natural vegetation is widely scattered and large patches are rare, extinction rates should be higher and rise more sharply with urbanization than where remaining vegetation includes some large or connected patches (Shafer, 1999; Donnelly and Marzluff, 2004a).
(2) Where human use of remaining natural areas for recreation, resource extraction, or other purposes that introduce exotic plants and animals, simplify structural complexity, reduce shrub and ground cover, or reduce the area's productivity, extinction should rise sharply with urbanization regardless of how the natural areas are configured.

Colonization curves may take a variety of shapes with respect to the occurrence of natural vegetation in a landscape, but certainly they will generally decline with reduced land cover change (i.e., a high percentage of native vegetation in the landscape; Fig. 6(B) and (C)). In Seattle, colonization was aided by the use of interspersed built and unbuilt areas by black-capped chickadees, downy woodpeckers, and red-breasted nuthatches, and by the abilities of violet-green swallows to find, exploit, and pack into subdivisions where the forest canopy was perforated by houses.

This leads me to four hypotheses:

(3) Where colonizing species benefit from edge and habitat, colonization will peak at intermediate levels of land cover change.

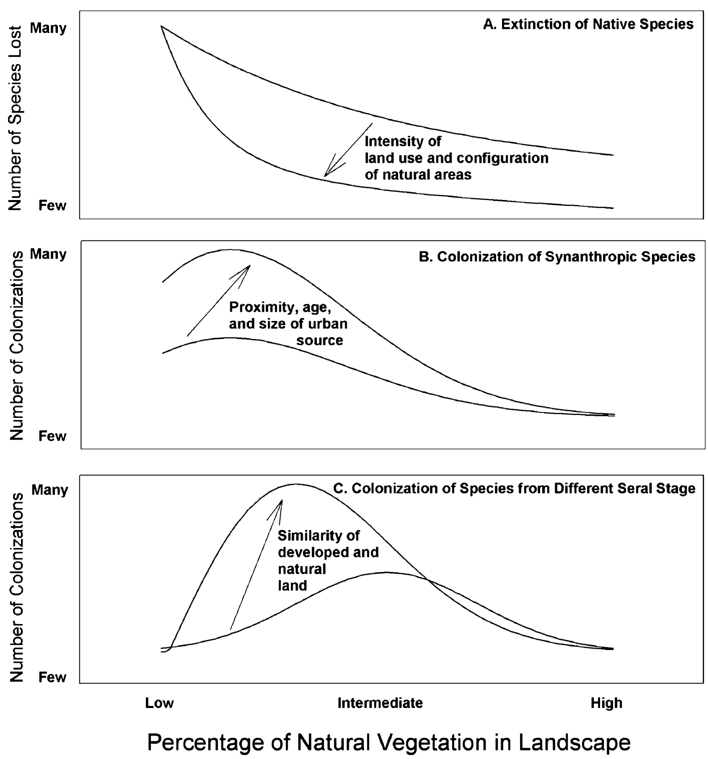

Fig. 6 Hypothesized variation in extinction and colonization as a function of urbanization (loss of forest cover)

(4) Where colonizing species benefit from release from predators or competitors sensitive to human activity, colonization will peak in mostly urban landscapes.
(5) Where colonists have high dispersal abilities and are able to use small habitat patches, colonization will also peak at intermediate levels of land conversion (Fig. 6(C)).
(6) Colonization by synanthropic species will be proportional to the amount, proximity, and age of settlement (Fig. 6(B)).

The pattern of diversity with respect to the amount of urbanization is determined by the difference in colonization and extinction. Colonization should strongly influence the pattern if it exceeds extinction. This was the case in my empirical example, probably because many synanthropic and early successional colonists were nearby. However, in landscapes with frequent extinctions and rich native faunas, the shape of the extinction curve is expected to exert a strong influence on the pattern of diversity with respect to urbanization. Four hypotheses are testable:

(7) Colonization will determine the pattern of diversity with respect to urbanization where urban areas provide rich, large, and proximal pools of synanthropic colonists (Fig. 6B).
(8) Colonization will determine the pattern of diversity with respect to urbanization where land transformation creates seral stages that are very dissimilar from existing natural vegetation. Dissimilarity between the natural and built environments will allow native species not normally found in the existing natural landscape to colonize urbanizing landscapes (Fig. 6(C)).
(9) Extinction will determine the pattern of diversity with respect to urbanization where extinction exceeds colonization. The resulting pattern of diversity should follow the shape of the extinction curve in direct proportion to the magnitude of extinction relative to colonization.
(10) Where extinction is substantial, it may also balance colonization at each point along the gradient of urbanization leading to a constant value of diversity regardless of the amount of urbanization. In this case, turnover in species composition is expected despite constancy in species diversity.

The preceding theory and predictions are concerned with local, or alpha, diversity. Regional and global diversity will also respond to urbanization, primarily through the process of biotic homogenization (Olden *et al.*, 2003). Homogenization, or the reduction in regional and global diversity as cosmopolitan exotic species invade and eventually replace native endemic species, is thought to depend primarily on the type and number of invasive and native species, the historical similarity among donor and recipient communities, and the richness of recipient communities (Olden and Poff, 2003). Considering biotic homogenization in urbanizing landscapes from the colonization and extinction perspective I have detailed above provides three hypotheses:

(11) The cosmopolitan nature of synanthropic species will determine the degree of homogenization observed in urban plant and animal communities. Where most synanthropic colonists are from the region of study, homogenization will be less than where synanthropic colonists occur globally in association with humans.
(12) The relative importance of synanthropic versus seral stage colonists will determine the degree of homogenization in urban plant and animal communities. Because most seral stage colonists are native to the locale or region, where they colonize more frequently than cosmopolitan, synanthropic species, homogenization of regional (and especially global) urban communities will be slow.
(13) The degree of homogenization in urban plant and animal communities will be greatest when colonization precipitates extinction, for example when colonists prey on or compete with remaining native species.

Discussion

The diversity of birds in small landscapes (1 Km^2) varied with the amount of natural vegetation, and its converse, the amount of urban land cover. Landscapes with intermediate amounts of settlement and forest cover had the greatest diversity; 20–35 species of songbirds. In contrast, most landscapes with either extensive settlement or no settlement had fewer than 20 species of songbirds. Low diversity in the most natural landscapes was consistent, but at first perplexing. All seven reserves had low bird diversity that appeared to reflect the relatively homogeneous, coniferous forest. Among the seven reserves, those that had the most diverse land covers (riparian, regenerating forests) also had the greatest bird diversity. But even these sites only had a maximum diversity of 22 species; well below that found in moderately settled areas. The correlation between urbanization and bird diversity has biological, theoretical, and practical implications.

Biological Implications

The response of bird diversity to the disturbance of settlement is generally consistent with the intermediate disturbance hypothesis (Roxburgh *et al.*, 2004), and specifically consistent with the disturbance heterogeneity model (Porter *et al.*, 2001). Where disturbance is extreme, syanthropic species dominate bird communities. Where disturbance is rare, native forest species dominate. But where disturbance is intermediate, a rich diversity of species coexist. Determining the factors allowing for such coexistance has motivated ecologists for five decades (Roxburgh *et al.*, 2004). In the urbanizing region around Seattle, intermediate disturbance increases local bird diversity most clearly by increasing the local diversity of resources, confirming Porter *et al.*'s (2001) extension of the disturbance heterogeneity model to urban ecosystems. As expected, diversity peaked where the novel urban land cover occupied approximately 50% of the landscape. Settlement of this principally forested region produces edges, grasslands, small ponds, canopy breaks, gardens, anthropogenic nest sites, and deciduous woodlands that are rapidly colonized by species rarely found in pure coniferous forests of the Pacific Northwest. Intermediate disturbance combines rich mixes of land covers into small areas, each of which is inhabited by a unique set of birds. Birds characteristic of built and early successional landscapes invaded the formerly continuous coniferous forest to increase diversity. In a similar way, urbanization increased tree species diversity, which together with increased landscape patchiness accounted for increased bird diversity in moderately-settled suburban areas of California and Ohio (Blair, 2004).

Settlement does not appear to stymie the superior competitive abilities of some species, and therefore does not appear to alleviate competitive exclusion as may occur in some intermediately disturbed systems (Connell, 1978; Huston, 1979). In fact, settlement may increase competitive interactions within a locally diverse community. Urbanization in Seattle adds a second wren (Bewick's wren), several titmice (bushtit, black-capped chickadee), a host of aerial insectivores (swallows and flycatchers), a versatile ground forager (song sparrow), and many seed eaters which could compete with native forest winter wrens, Pacific-slope and Hammond's flycatchers, spotted towhees, and chestnut-backed chickadees. I have no evidence of this, but my study is limited by the relative newness of European settlement in the western United States.

The characteristics of colonizing species suggests that biotic homogenization will be slow and may be confined to regional, rather than global scales. Cities in the coniferous forests that flank the northern Pacific Ocean are likely to develop similar avifaunas as a few cosmopolitan, synanthropic species (European starling, rock pigeon, American crow, house finch, and house sparrow) and a diverse collection of early seral stage, regional natives invade human settlements. While this new collection of species will be similar among cities in the region, this is not unnatural. Pacific coniferous forests have similar avifaunas because they have few locally endemic birds. Regional homogenization is unlikely to cross natural boundaries because nearby mountain, dry forest, shrubland, or grassland cities would each attract a distinct set of early seral stage colonists. The sort of invasion of urban bird communities I documented is unlikely to produce rapid and global homogenization because few globally-distributed species occurred and these did not directly cause the extinction of native species. House sparrows and rock pigeons do not appear to compete with native sparrows and pigeons, although they may increase the susceptibility of native species to exotic diseases. European starlings have minimal effects on native cavity nesters (Blewett and Marzluff, 2005).

Seattle is a young city, barely 100 years old. Most of my study areas are in the region where development is rapidly occurring and has only been occurring for a few decades. Therefore, persistence of high diversity where disturbance is intermediate and maintenance of heterogeneous avifaunas may not be stable in ecological time (e.g., over tens of bird generations). In fact, we can already detect some relaxation of diversity in intermediately-disturbed subdivisions that vary in age from 20 to 100 years (Donnelly 2002; Ianni and Marzluff, in preparation). However, relaxation of diversity is not sufficient to erase the peak in diversity seen in moderately settled landscapes. It appears that, in the variable geography of the Pacific Northwest, heightened bird diversity in moderately urbanized locales does not warn of impending collapse of diversity as it does in many fish communities (Scott and Helfman, 2001). I suspect urban bird diversity will remain high in the Pacific Northwest because invading synanthropic species offer little competition to native species. If current processes remain in operation, then bird communities should remain diverse, but differ in composition from historical communities as settled forests change in quality, age, and perhaps predator loads. Changing bird communities remain effective early warning systems, alerting people to the consequences of their actions. But, unlike plants and animals facing severe competition or predation from invading species (e.g., Scott and Helfman, 2001), diverse urban bird communities may alert people to the possibility of balancing their needs with the needs of other species. Achieving this requires understanding and planning (see **Practical implications**).

Other nuances of Seattle may also make my results relatively site-specific. Seattle is surrounded by expanses of natural lands that may function as important reservoirs of native forest birds. Perhaps these are sources that continually restock urban areas. As a coastal city, Seattle's climate is benign, but its northern latitude and relatively simple, coniferous forest produces a simple bird community. Perhaps richer avian assemblages, or more seasonally variable ones, will respond differently to urbanization. This has not been observed, as researchers have found moderate settlement (suburban to exurban; definitions in Marzluff *et al.*, 2001b) to increase bird diversity in a variety of areas (forests, shrublands, and grasslands from coastal, desert, and inland biomes; Sewell and Catterall, 1998; Blair, 2004; Zack Jones and Carl Bock, personal communication). But little is known about urban bird dynamics in rich tropical locales (Marzluff *et al.*, 2001b).

Ongoing research is shedding light on the mechanisms underlying extinction of native forest birds in my study area. Reproduction and dispersal do not appear to be compromised by settlement (Donnelly 2002; Donnelly and Marzluff, 2004a; Blewett and Marzluff, 2005; Kara Whittaker, unpublished data). Rather, low population size resulting from reduced amount and quality of natural forest appears to reduce long-term viability of populations (Donnelly 2002; Donnelly and Marzluff, 2004a). Reduced survival of fledged young and perhaps breeding adults may also be important.

The generally greater response of bird diversity to the type and amount of vegetation, rather than to its configuration is likely taxon-specific. Birds are exceedingly mobile which allows them to quickly recolonize small habitat patches and travel between disjunct patches. Species less able to traverse landscapes, for example aquatic insects or small mammals, may be more responsive to the pattern of urbanization (Alberti and Marzluff, 2004; Hansen *et al.* in press).

Theoretical Implications

Investigating the processes of extinction and colonization separately helps to untangle the opposing forces that define an area's standing diversity. This approach has been successfully pioneered in urban settings by Blair (1996, 2001a, 2004). My expansion of these ideas into a graphical format with explicit hypotheses about factors that may affect colonization and extinction in urbanizing landscapes (Fig. 6) is meant to stimulate others to test, refute, and refine the ideas. One important area in need of refinement is the actual measurement of extinction and colonization. I simply determined these by an annual assessment of presence or absence. It may be better to consider a longer time span of absence before concluding extinction. Detailed study of the process of colonization is also needed. Do some colonists visit a site early in the season, but not stay? Do others use a site inconspicuously without breeding there? These refinements would aid our understanding of the dynamics of bird diversity. Studies of extinction and colonization may be especially insightful if they can be done before, during, and after development.

The three guilds of birds I have used make sense in an urbanizing environment where settlement adds novel anthropogenic resources, produces a variety of seral stages, and removes currently natural vegetation. Further insights could come by subdividing these guilds into more traditional ones like cavity versus ground nesters or aerial versus foliage foragers. However, an even more insightful approach is to look at the colonization and extinction of individual species and relate this to land cover change resulting from urbanization. Testing for community nestedness and relating this to urban gradients is powerful (Lomolino, 1996). In my study area, this approach showed that many species have thresholds of occurrence with respect to the amount of urban or forest land cover (Donnelly, 2002; Donnelly and Marzluff, 2004a). In response to settlement, back-capped chickadees, song sparrows, American crows, black-headed grosbeaks, and bushtits are the first to colonize. Black-throated gray warblers, hairy woodpeckers, western tanagers, Pacific-slope flycatchers, brown creepers, and winter wrens are the first to disappear.

Practical Implications

Planners, developers, policy makers, managers, and homeowners can use the results of ecological studies in urban environments to increase the sustainability of human settlement. Sustainable development must have at its foundation ecological sustainability. Ecological sustainability is related to diversity (Loreau, 2000), so providing for diverse bird communities in urbanizing landscapes is one step down the path to sustainability. My results suggest that, at the local scale, bird diversity is enhanced by moderate settlement. But this does not mean that moderately settling all land will enhance bird diversity regionally. Widespread settlement and globalization clearly do not enhance diversity. They homogenize and reduce it (Lockwood *et al.*, 2000; Blair, 2001b), especially as larger geographic scales are considered (Sax and Gaines, 2003). While I suspect the benign nature of invading species and geographic complexity of my study region will slow and limit homogenization, if urban planners and land managers consider the needs of native forest species, early successional species, and synanthropic species separately they can actively work against homogenization. Moderate settlement enhances diversity in my study area by providing habitats used primarily by early successional and deciduous forest birds. Synanthropic and native forest birds occur in moderately settled areas, but to provide more explicitly for them, requires maintaining some extremely developed as well as some undeveloped land. Providing for the full diversity of birds requires the full diversity of habitats—from developed to undeveloped. Simply stated, fighting homogenization and maintaining bird diversity is best accomplished by not doing the same thing everywhere (Bunnell, 1999).

Table 2 Conservation strategies for birds in urban areas and the participants who must cooperate for effective implementation. Strategies are from Donnelly (2002), Donnelly and Marzluff (2004), and Blewett and Marzluff, 2005

Strategy	Regional planner/ policy maker	Local planner/ policy maker	Large-scale developer	Conservation activist	Conservation/ restoration scientist	Educator/ interpreter	Local open space manager	Builder	Home owner
Don't do the same thing everywhere	X	X	X	X	X	X	X	X	X
Create large public reserves	X	X	X				X	X	
Implement growth management	X	X	X				X	X	
Monitor land cover change	X			X	X				
Maintain/restore native vegetation	X				X	X	X		X
Demonstrate value of open space				X	X	X			
Keep interior conditions in some open space				X		X	X		X
Reduce anthropogenic food supplementation						X	X		X
Maintain snags and short-lived trees		X				X	X	X	X
Increase native shrub and ground cover							X	X	X
Encourage premium pricing for functional open space		X	X	X		X		X	X
Manage exotic predators							X		X
Educate home owners about edge effects and disturbance				X	X	X			
Create stewardship incentives	X	X		X					

Planners who encourage the same style of development across a landscape may increase local diversity or favor one group of species over another, but at the regional scale they will reduce biological diversity and therefore lower the sustainability of development.

Not doing the same thing everywhere requires planning. Often in my study area developers lobby for increased settlement, while conservation activists argue for large, undisturbed reserves. If both "win", birds will suffer because few areas of moderate settlement may remain. In terms of Fig. 2, we will end up with both low diversity ends of the curve on the landscape and miss the high diversity peak. Planners, policy makers, and open space managers interested in maximizing biological diversity should devise strategies and incentives to maintain moderately-settled areas in the region and balance their occurrence with undeveloped and highly developed landscapes.

Planning at the local to regional scale requires the cooperation of developers, policy makers, urban planners, homeowners, and urban ecologists. Each and every strategy that our work suggests will increase bird diversity requires actions to be carried out by a diversity of participants (Table 2). For example, activists, scientists, and educators will need to cooperate to inform residents, planners, and policy makers about relevant research like the effects of disturbance on forest birds. Likewise, builders, developers, regional planners, county commissioners, and other policy makers will need to cooperate to implement effective growth management.

Urban ecologists will increasingly be called on to share their science with planners, policy makers, developers, and homeowners. To effectively serve this varied clientele requires nontraditional training. Specifically, urban ecologists will increasingly require interdisciplinary training to understand how policies are formulated and implemented, how planners design landscapes, and what people want from their immediate surroundings. New interdisciplinary programs are emerging (Alberti *et al.* 2003, Tress *et al.* 2003), but students must open their eyes, ears, and minds widely. International travel is increasingly important. Take the window seat and look at patterns of development. Ask whether the patterns below you provide local, regional, and global diversity. Learn about foreign policies and value systems that seem to result in diverse landscapes. Help globalize knowledge so that we do not continue to do the same thing everywhere.

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A Long-Term Survey of the Avifauna in an Urban Park

Michael Abs and Frank Bergen

Abstract Eight censuses of the breeding bird community in a 10 ha urban park in Dortmund, Germany, conducted over a time span of 43 years, revealed an increase in species number as well as in breeding density (territories/10 ha). We found a high species turnover rate of 42 % favouring generalist species and perhaps woodland species. Indicator species according to Flade (1994) are discussed. The ratio of the number of Blackbird *Turdus merula* territories to the number of Chaffinch *Fringilla coelebs* territories is used to describe the progress of urbanisation in breeding bird communities.

Keywords: urban habitat · breeding bird community · long-term trends · species turnover · urbanization.

1 Introduction

Long-term studies based on field observations of birds are urgently needed because they can provide the basis for statements with prognostic value. In this paper we discuss the development of the breeding bird community in an urban park over a period of 43 years. The study was carried out in Westpark, which covers an area of 10 ha and is situated 1.4 km from the city centre of Dortmund, Germany (51°30′ N; 7°28′ E). Founded in 1811 as a cemetery Westpark was transformed into a park after 1945. It is isolated from other green areas by extensive housing areas surrounding it. The tree vegetation is dominated by ash (26%) besides maple and birch, but oak, poplar, plane tree and horse chestnut also occur, whereas conifers are rare. The oldest trees are about 100 years of age. Since 1954 ground cover of the shrub layer changed very little from 18 % to 20 %. Regularly mown lawns cover the open spaces and much of the area under the trees. In the 1950s, 30 nestboxes were installed of which only one remained in 1997.

2 Material and Methods

The first five censuses were carried out by Erz (1956, 1964). Bergen (1996) monitored the breeding bird density in the 1990s using Oelke's (1974) methods. Between March and June ten visits took place in every study year. Data were analysed according to Hustings *et al.* (1989). In order

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to characterise the dynamic of the species composition we calculated the species turnover rate (T) using the slightly modified formula given by Mühlenberg (1993):

T = $(J+E) \times 100/(S_I + S_{II})$ with

J: number of additional species in season II

E: number of species that disappeared between season I and II

$S_I(S_{II})$: number of species found in season I (season II)

Statistical analysis was carried out using Mann-Whitney-U-Test.

3 Results and Discussion

The species number increased from 17 species (median) in the 1950s to 21 (median) in the 1990s. Breeding bird density also increased from 100 to 138 territories/10 ha. Both of these increases already occurred in the 1960s. The overall turnover rate of 42.1 % indicates that nearly half of the species in Westpark changed. Only ten species were found in all eight years of study (table 1). This is comparable with the turnover rate of the breeding bird community at a cemetery in Dortmund (Ostfriedhof) over a period of 35 years (T = 42.6 % according to Erz [1964] and Weiβ [1997]). For the avifauna of a cemetery at Lausanne, Switzerland, a turnover rate of 35.0 % was found over a period of 24 years (Ravussin & Mellina 1979). We calculated turnover rates of 25.0 % for birds in pine forests (according to data given by Dierschke 1973) and of 27.5 % for birds in deciduous forests (data from Tischler 1976). These turnover rates in nearly natural habitats may be related to succession as well as to ageing of the trees. In comparison with this, the higher turnover rates in urban parks like Westpark or cemeteries may be related to the long-lasting process of urbanisation of certain species.

Following the definition of dominance values given by Bick (1989; eudominant > 10%), in the 1950s three eudominant species occupied about 50 % of all territories (Blackbird 20.2 %, House Sparrow *Passer domesticus* 16.6 % and Chaffinch 12.5 %). In the 1990s, however, only two eudominant species filled nearly 40.0 % of all territories (Blackbird 26.1 % and Wood Pigeon *Columba palumbus* 12.3 %) indicating a decline in species diversity. During the entire study period from 1954 to 1997 seven of the nine indicator species typical for the landscape type 'parks' (according to Flade 1994) occurred in Westpark. Five indicator species bred in Westpark in 1961, four in 1954 and 1962 and three in all other years of study. Only one indicator species, the Spotted Flycatcher *Muscicapa striata* was observed in all censuses (Table 1). Using Flade's equation for the indicator species-area relationship, four indicator species can be expected for an urban park of 10 ha. Furthermore, if we consider the total number of breeding species, Flade's equation leads to an expectation of 26 species instead of 21 which breed there now. In summary, the habitat and resources in Westpark seem to favour generalists. Species like Blackbird, Great Tit *Parus major* or Blue Tit *Parus caeruleus* profit from additional food provided by people. But for more specialised birds like Whitethroat *Sylvia communis* or Icterine Warbler *Hippolais icterina*, both disappeared after 1964, Westpark no longer seems to offer suitable breeding habitat. This may be due to the high rate of human disturbance. The immigration of the Nuthatch into Westpark after the 1960s and its first occurrence in a cemetery of Lausanne in 1965 (Ravussin & Mellina 1979) support the idea of an ongoing occupation of suboptimal habitats by this species (Gatter 1998).

In search for a measure for the ongoing process of urbanisation we took the ratio of Blackbird versus Chaffinch territories. These two species show no interspecific competition. We compare our data with those from urban parks of east-central European cities (Biadun 1994; Luniak 1981; Müllerova-Franekova & Kocian 1995) and from Bialowieza National Park (Tomialojc & Wesolowski 1990) as a natural woodland reference. Figure 1 shows that the ratio obtained in Westpark in early years is low, while values from the 1990s are significantly higher (U-Test; $p<0.005$). In two cemeteries in Dortmund and in Lausanne this ratio also grew from 2.2 in 1962 to 4.6 in 1996 (Weiβ 1997) and

Table 1 Numbers of species breeding in Westpark, Dortmund, breeding density (territories/10ha) and turnover rate (species which were observed at all years of study are printed in bold; indicator species are printed in italics)

	Territories / 10 ha		1954	1955	1956	1961	1962	1994	1995	1997
1	Woodpigeon	*Columba palumbus*	7.1	7.1	3.6	8.9	10.7	17.0	24.0	14.0
2	*Collared Dove*	*Streptopelia decaocto*	–	–	–	*3.6*	*5.4*	*1.0*	*2.0*	–
3	*Green Woodpecker*	*Picus viridis*	–	–	–	–	–	–	–	*1.0*
4	Great Spotted Woodpecker	*Picoides major*	–	–	–	–	–	–	–	1.0
5	Wren	*Troglodytes troglodytes*	–	–	–	–	1.8	3.5	7.0	4.0
6	Dunnock	*Prunella modularis*	1.8	1.8	3.6	3.6	3.6	4.0	7.5	4.5
7	Robin	*Erithacus rubecula*	–	–	–	1.8	1.8	3.0	3.0	2.0
8	*Redstart*	*Phoenicurus phoenicurus*	*5.4*	*5.4*	*5.4*	*5.4*	*5.4*	–	–	–
9	Blackbird	*Turdus merula*	17.8	21.4	21.4	25.0	23.2	43.0	41.5	30.0
10	Song Trush	*Turdus philomelos*	–	–	–	1.8	1.8	3.5	6.0	2.0
11	Mistle Trush	*Turdus viscivorus*	–	–	–	–	–	1.0	1.0	–
12	*Icterine Warbler*	*Hippolais icterina*	*1.8*	–	*1.8*	*3.6*	*3.6*	–	–	–
13	Whitethroat	*Sylvia communis*	1.8	1.8	1.8	1.8	1.8	–	–	–
14	Blackcap	*Sylvia atricapilla*	–	–	–	1.8	1.8	3.0	6.0	5.5
15	Chiffchaff	*Phylloscopus collybita*	3.6	3.6	5.4	5.4	5.4	3.0	6.5	7.0
16	Spotted Flycatcher	*Muscicapa striata*	1.8	1.8	1.8	1.8	1.8	2.0	3.0	1.0
17	Long-tailed Tit	*Aegithalos caudatus*	–	–	–	–	–	–	1.0	–
18	Marsh Tit	*Parus palustris*	–	1.8	–	–	–	–	–	–
19	Blue Tit	*Parus caeruleus*	1.8	1.8	1.8	3.6	1.8	11.5	17.5	13.0
20	Great Tit	*Parus major*	3.6	3.6	3.6	3.6	1.8	12.5	14.5	11.0
21	*Nuthatch*	*Sitta europaea*	–	–	–	–	–	*1.0*	*2.0*	*3.0*
22	Short-toed Treecreeper	*Certhia brachydactyla*	–	–	–	–	–	3.0	5.0	4.0
23	Magpie	*Pica pica*	–	–	–	1.8	1.8	1.0	1.5	1.0
24	Carrion Crow	*Corvus corone*	1.8	1.8	1.8	–	–	–	–	–
25	Starling	*Sturnus vulgaris*	7.1	5.4	5.4	8.9	7.1	6.0	7.5	8.0
26	House Sparrow	*Passer domesticus*	17.8	14.3	17.8	16.1	17.9	–	–	–
27	Tree Sparrow	*Passer montanus*	3.6	5.4	3.6	7.1	7.1	–	–	–
28	Chaffinch	*Fringilla coelebs*	12.5	12.5	12.5	14.3	16.1	12.0	14.0	10.0
29	*Serin*	*Serinus serinus*	*3.6*	*1.8*	–	*1.8*	–	–	–	–
30	Greenfinch	*Carduelis chloris*	5.4	7.1	8.9	10.7	10.7	4.0	7.0	2.0
31	Goldfinch	*Carduelis carduelis*	1.8	–	–	–	–	2.0	1.0	1.0
32	Bullfinch	*Pyrrhula pyrrhula*	–	–	–	–	–	–	0.5	–
33	Hawfinch	*Coccothraustes c.*	–	–	–	–	–	1.0	1.0	–
	Number of species		18	17	16	21	21	21	23	20
	Number of territories		100.1	98.4	100.2	132.4	132.4	138.0	180.0	125.0
	turnover rate between successive years of study			8.6	9.1	18.9	4.8	23.8	4.5	16.3
	overall turnover rate		42.1							

from 1.9 in 1953 to 2.8 in 1978 (Ravussin & Mellina 1979), respectively. The ratios from plots in Bialowieza National Park are by far the lowest we calculated.

These results show that high Blackbird/Chaffinch ratios are typical for urban habitat types such as parks and cemeteries, not in the past but at present. We therefore regard this ratio as a suitable measure to describe the progress of urbanisation in breeding bird communities.

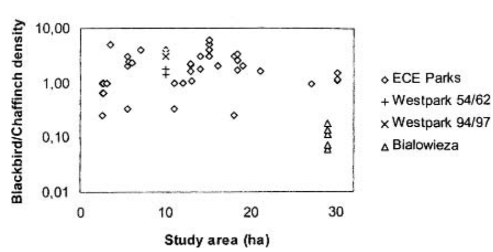

Fig. 1 Density ratio of Blackbirds to Chaffinches in different habitats

There could be an excess of Blackbird males in urban parks (Erz 1964) so that the number of breeding pairs in the Westpark might be lower than the number of singing males counted and taken as an indication for Blackbird territories. However, this has no effect on the density ratio of Blackbirds to Chaffinches as a measure of urbanisation.

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Biodiversity in the Argentinean Rolling Pampa Ecoregion: Changes Caused by Agriculture and Urbanisation

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Summary The metropolitan area of Buenos Aires is located in the Rolling Pampa, one of the most productive ecoregions of the world (44, 000 km^2, 33° S − 39° S). This region has undergone deep transformations caused by agricultural, residential, industrial and commercial land-uses. The purpose of this paper is to compare to what exient plant and avian richness is influenced by urban and agricultural uses. To capture the land-use effects a comparison between different sectors was made. Green spaces and farmland, located in areas of contrasting land-use, extending from the La Plata's river shore to semirural and rural areas, were sampled. Vascular plant richness, floristic composition and bird presence were considered. To compare the different sites, biodiversity indexes and Sørensen's coefficient of similarity were calculated and the percentage of forest, grassland, shrubland, wetland and rivers/streams was estimated. All collected data were correlated using Principal Components Analysis (PCA). In general, the decrease of native plants and bird richness towards the city centre is consistent with the studies of other cities. The results of the present study confirm that the actual cultivation practices are extremely more dangerous for conservation of native land than urban sprawl.

Keywords: Biodiversity · Argentina · Buenos Aires · plant species · bird species · species diversity

Introduction

In 1774 Thomas Falkner described the Argentinean Pampa as "One immense sea of grasslands, with scattered forest islands . . ." (Ghersa a. León 2001, 571). With this statement he clearly transmitted the perception of the native inhabitants and the reasons why they named this landscape Pampa – in the natives' language Quechua "Pampa" means "uniform extensive plain". Since then, this landscape has gradually disappeared by high pressure of agriculture and urbanisation.

In general, urban areas are very dynamic and consist of very distinct environmental gradients promoted by anthropogenic effects. This leads, for example, to a fragmentation of native vegetation and to an introduction of alien species.

Agriculture is a threat to biodiversity and may even be greater than the threat of urbanisation. Ricketts and Imhoff (2003) recommended identifying where high levels of human pressure on ecosystems and biodiversity coincide. Nowhere in Argentina does this issue acquire greater prominence than in the urban/rural interface of the Rolling Pampa. Matteucci et al. (1999) and Morello et al. (2003) emphasised that urban sprawl is responsible for the dramatic conversion of land that is

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still going on. The location of settlements on loessial soils of high quality contributes substantially to the destruction of the regional biodiversity.

Living biota depends on the spatial heterogeneity within cities and the surrounding landscapes (Pickett et al. 2001). Rebele (1994) and Kowarik (1990, 2003) proved that the proportion of coloniser and exotic species is characteristically higher in urban vegetation than in near-natural habitats. According to Melles et al. (2003) the variety of bird species decreases as a consequence of an increasing urbanisation. Many authors have suggested that, on the local scale, plants and birds can be used as indicators of environmental changes (Blair 1999; Faggi et al. 2003). Porter et al. (2001, 132) recommended quantifying those changes caused by urbanisation "in order to move beyond simplistic descriptions of urban habitats". Moreover, they pointed out the significance of partially developed landscapes which may serve as an important refuge for many species.

The purpose of the present paper is a) to compare to what extent plant and bird richness in the Rolling Pampa are influenced by urban and agricultural uses; b) to analyse the changes in plant diversity caused by more than 50 years of agriculture, comparing present and historical data.

Rolling Pampa

The Rolling Pampa, as one part of the whole "Pampa" area, is characterized by the presence of low round-topped hills (1–3°) (SAGYP-INTA 1995), which gives the landscape an undulating aspect. Figure 1 locates the Humid Pampa region. Within this area, the Rolling Pampa ($44,000\,\mathrm{km}^2$) is marked and the study locations are pointed out.

The climate of the region is subtropically humid, characterised by long warm summers and mild winters, with an average air temperature of 11 °C in July and 25.5 °C in January and a mean annual precipitation of 1,147 mm (climate station 34°35'S, 58°29'W) (Sträber 1999, 166).

Before the European settlement, this area was presumably a mosaic of dry and riparian forests and other wetlands, which were dominated by grasslands on the eolian terrace. When the Spanish conquerors began to settle down at the end of the 16th century they used the forests as combustible.

In the middle of the 19th century two main transformation processes started: a) the eolian terrace was converted into agriculturally productive land, which led to a loss in biodiversity and

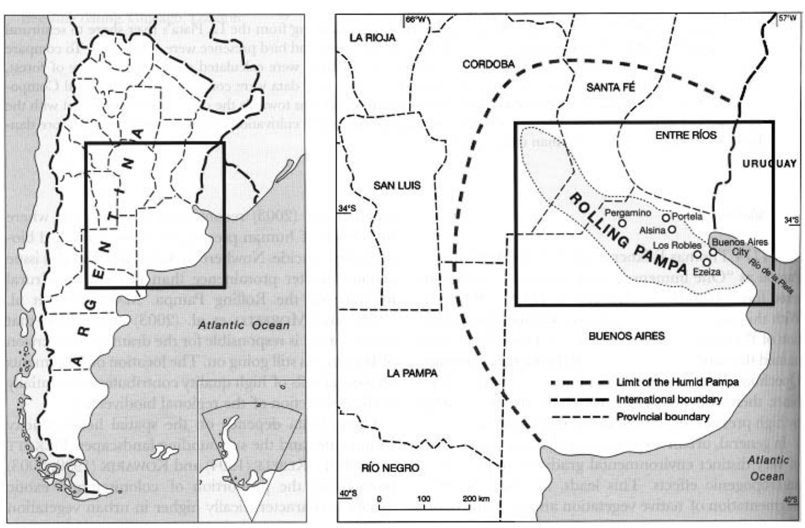

Fig. 1 Location of the study area

b) an important immigration took place and the small settlements were converted into cities. The immigrants brought many of the ornamental exotic plants which today are important elements of the urban vegetation. On the eolian terrace an intensive change from agriculture to residential, industrial and commercial land-use started in the year 1940 (Morello et al. 2003). Under the pressure of a greater need for building space, important parts of the river shore were filled up. In some areas of this land enhancement a recovery of native biodiversity took place.

The Rolling Pampa is one of the most productive ecoregions of the world. Since 1990 intensive agriculture aided by modern technology is being performed (Viglizzo et al. 2002). Today, the dominant land use of the Rolling Pampa is agriculture with soybean, sunflower, maize and wheat as principal crops.

Material and methods

Study area

The study has been conducted in urban, suburban, periurban, semirural and rural sectors in the Rolling Pampa. Every sector includes specific sampled sites (Table 1). The population density ranges from 250–587 hab/ha in the urban area and is $\leq 250\,\mathrm{inh./ha}$ in the suburban area (SECRETARIA DE PLANEAMIENTO URBANO Y MEDIO AMBIENTE, GCBA 1998). Periurban, semirural and rural areas show lower densities between 4 and 5 hab/ha (INDEC 2001).

Characterisation of the sampled areas

1. Urban sector:

   a) The Natural Reserve Costanera Sur with a surface area of 350 ha is located one km away from the Buenos Aires city centre. It is an artificial land enlargement at La Plata's riverside, which was created in the early seventies of the 20th century using remains of the buildings which had to be pulled down for a new high-way construction. The area within the embankments was filled with sand and silt from the river and was partially drained afterwards. Spontaneous natural plant succession took place and many ecosystems developed. Therefore it has been considered as a blind probe for the present study. The diversity of biotopes such as lagoons, wetlands, grasslands and forests guarantees the presence of a rich fauna. In 1986 the City Council, following the demand of many non-governmental environmental organisations, declared this reclaimed land a protected area. Figure 2 shows a photograph of the Costanera Sur with the skyline of Buenos Aires in the background.

Table 1 Sectors and sites of the study area

Sector		Sites
Urban	Natural Reserve Costanera Sur	
	Urban Parks	Lezama, España, Patricios, Chacabuco
Suburban	Park Indoamericano	
	Park Ribera Sur	
Periurban	Ezeiza	
Semirural	Los Robles	
Rural	sector at the middle of 20^{th} century (rural 1930)	Pergamino
	sectors at the end of the 20^{th} century (rural present)	Ireneo Portela
		Alsina

Fig. 2 Buenos Aires downtown seen from Costanera Sur Reserve

b) Urban Parks: the Lezama (7.36 ha), España (5.78 ha), Patricios (15.41 ha) and Chacabuco (24.41 ha) parks are completely surrounded by densely built-up areas with few green or open spaces. The parks consist of planted tree areas with lawns or flower beds. Figure 3 shows Parque España represented by a photograph and an IKONOS satellite image.

2. Suburban sector:
   The Parque Indoamericano with an extension of 55.2 ha and the recreational area Parque Ribera Sur (38.43 ha) were considered. These green areas show planted trees and lawns and are surrounded by open spaces visible in the IKONOS-subsets of the Parks (Fig. 4).
3. Periurban sector:
   Ezeiza has a surface area of ca. 708 ha. Isolated buildings are surrounded by parks. Most of the area is covered by afforestations alternating with savannahs, grasslands and wetlands. This area borders on residential settlements, recreational clubs, grazing fields and the infrastructure of the Buenos Aires international airport. In the last decade cattle-breeding stopped and today, the vegetation is left to its own devices. Past and present cartography shows that this area, once used for farming, was converted from mesic and hydrophytic grassland into a mosaic of grassland and forests.
4. Semirural sector:
   The reserve Los Robles (568 ha) is located at about 43.5 km North/East of the city centre (34°40'S and 58°52'W). It is used as a recreational area and for environmental education. The reserve is surrounded by agricultural and horticultural fields. Los Robles borders on an artificial lake with a surface area of about 400 ha which was formed by the construction of a dike between 1968 and 1972. The vegetation of the reserve consists of native dry forests, savannahs, forests with exotic species, grasslands, wetlands and crop fields (Burgueño 2004).

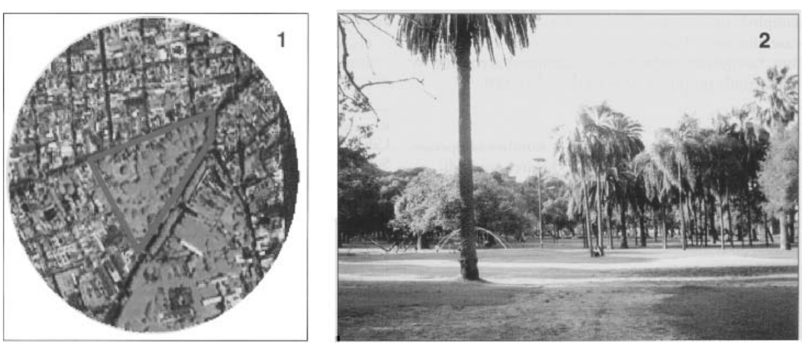

Fig. 3 Urban sector Parque España (+ surroundings)
(1) Satellite image IKONOS (multispectral 4 m resolution, Green-NIR-Red)
(2) Parque España

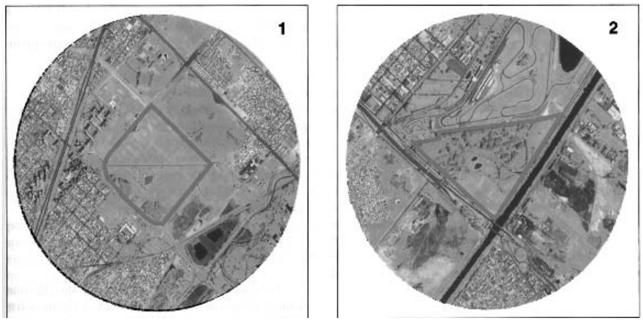

Fig. 4 Suburban sector (1) Parque Indoamericano and (2) Parque Ribera Sur
Satellite image IKONOS (multispectral 4 m resolution, Green-NIR-Red)

5. Rural Sector:
   Two temporal situations were chosen: 1. a rural sector at the end of 20th century and 2. a rural sector at the middle of the 20th century.

   For the rural sector at the end of the 20th century two areas in the northern part of the province of Buenos Aires were considered. Due to their mild climate, gentle topography, fertile soils and water supply, these rural localities are representative for agriculturally productive land. The vegetation of Alsina (33°55'S and 59°20'W, 21,909 ha) was described by Faggi (1986) and Ireneo Portela (34°00'S and 59°30'W, 42.550 ha) by Faggi (1996).

   For the rural sector at the middle of 20th century Pergamino district was chosen, because topography, soil and climate conditions of this area are very similar to Ireneo Portela and Alsina (SAGYP-INTA 1995) and scarcely any other studies concerning flora and fauna were made in the past. Pergamino District (312,600 ha, 33°53'S, 60°36'W), devoted to agriculture, was analysed over 20 years by Parodi (1930). At that time the principal crops were maize, flax, wheat, alfalfa, oats and barley. He recorded 488 species, 318 natives, 120 exotics and 50 introduced from other regions of South America. At that time, the abundance of many grasses and herbs in the grasslands was striking.

Data acquisition

To capture the land-use effects on biodiversity (native species losses and exotic species increase) and the existence of thresholds, a comparison of the different sectors was made. Green spaces and farmland, located in areas of contrasting land-use, extending from the La Plata's river shore to semirural and rural areas, were sampled. The Natural Reserve Costanera Sur was analysed as a "blind probe" to contrast the results. To analyse the changes in plant diversity caused by more than 50 years of agriculture and cattle grazing, present and historical data are compared.

It was analysed how urbanisation and agriculture affect species richness and community similarity. In order to compare the different sample areas, the percentage of surface cover of the plant physiognomical units, here defined as wetland, grassland and woodland, was estimated.

Species richness

For each area vascular plant richness, floristic composition and bird presence were considered. In the urban, suburban and periurban areas vascular plants and birds were recorded by an exhaustive walkover of each study area between October and December 2002. In order to compare the selected areas, presence of plants and birds were listed. Richness of plants and birds for the semirural sector came from Burgueño (2004). Values for the richness of birdlife in the rural areas were extracted from Cueto and Lopez de Casenave (1999). Vegetation was studied by Faggi (1986, 1996). As the same methodology was used for all of the studies considered in the present paper, a comparison of the data is possible.

In general, the richness of flora and fauna is expressed as a number of individuals or as a biodiversity index which takes in account the number of individuals per area in a logarithmic way. For the comparison of sampled areas with different sizes, as in the present case, the use of biodiversity indexes is essential. Therefore, biodiversity indexes were calculated according to the formula proposed by Squeo et al. (1998):

$$\mathrm{Bi} = \mathrm{n}/\ln \mathrm{A}$$

where Bi = Biodiversity index; n = number of species and $\ln \mathrm{A}$ = natural logarithms of the area studied.

The biodiversity indexes of planted species, spontaneous exotic plants, spontaneous native plants, spontaneous annual plants and birds were considered.

Similarity

To quantify similarities between the green areas on the basis of self-growing plants (spontaneous species) the Sørensen Similarity Index was calculated (Sørensen 1948). It reflects the number of species two areas have in common. By using the following equation, different areas were compared by pairs:

$$\mathrm{S} = 2 * \mathrm{C}/\mathrm{s1} + \mathrm{s2}$$

where S = Sørensen's coefficient of similarity; C = number of plants common to the two sampled areas; s1 = number of plants in study area1 and s2 = number of plants in study area 2.

Values of this index vary from 0 to 1. The value 0 indicates that species assemblages differ totally (dissimilarity) and 1 that they are identical. If the Sørensen's coefficient is higher than 0.75, it was considered to reflect very high similarity, values in the range between 0.51–0.75 reflect high similarity, and values between 0.26–0.50 describe moderate similarity. Low similarity is described by values below 0.25 (Ratliff 1993). According to Ellenberg (1956) the vegetation of two sampled areas belongs to the same community if the index is higher than 0.25.

Plant physiognomical units

For each sector the percentage of the surface covered with forest, grassland, shrubland, wetland and rivers/streams was estimated by means of visual aerial photograph and satellite image interpretation (Table 2). The land uses were combined into three plant physionomical units: wetland (including river/ streams), grassland and woodland (which includes forest and shrubland).

Table 2 Percentage of plant physiognomical units

	Wetland	Grassland	Woodland
Reserve	35	50	15
Urban Parks	0	60	40
Suburban sector	20	53	27
Periurban sector	5	25	70
Semirural sector	12	6	82
Rural sector	24	66	10

Data analysis

The software STATISTICA PROGRAM was used to find correlations between the plant physiognomical units (wetlands %, grasslands %, woodlands %) and the richness parameters (biodiversity index of native plants, annual plants, exotic plants and birds) in the eleven study areas (the rural sector of the middle of the 20th century was not included).

To highlight similarities and differences among the studied sectors the plant physiognomical units and the richness parameters were taken in account using Principal Components Analysis (PCA) with the software Pcord 3.0. PCA is well known as a tool to (1) reduce the number of variables and (2) detect structure in the relationships between variables (Glavac 1996). The method involves a mathematical procedure that transforms a number of (possibly) correlated variables into a (smaller) number of uncorrelated variables called principal components. In the space it can be viewed as a projection of the observations on to orthogonal axes defined by the original variables. The first axis contains the maximum amount of variation. The second axis accounts the maximum amount of variation orthogonal to the first. Principal components are obtained by projecting the multivariate data vectors on the space spanned by the eigenvectors. The eigenvector associated with the largest eigenvalue has the same direction as the first principal component (axis 1). The eigenvector associated with the second largest eigenvalue determines the direction of the second principal component (axis 2).

Results

Vascular plants and birds richness of the different sectors is shown in table 3. In all sectors the most frequent plants are herbs, which are the typical components of grasslands of the eastern Argentinean pampa. Seven species are natives (*Paspalum dilatatum, Bromus catharticus, Gamochaeta simplicicaulis, Hydrocotyle bonariensis, Hypochaeris microcephala, Stenotaphrum secundatum, Funcus tenuis*), four aliens (*Trifolium repens, Poa annua, Carduus acanthoides, Cirsium vulgare*) and two grow world-wide (*Cynodon dactylon, Sporobolus indicus*). The presence of the exotic self-growing species is considerable in the urban parks (biodiversity index: 14.87) and the semirural area (biodiversity index: 14.22). In contrast to this, rural areas show an important decrease in exotic self-growing species.

In all study areas native birds are the principal component of avian richness. While 250 different bird species were observed in the Natural Reserve Costanera Sur, richness declines to 15 in the urban parks (Parque Lezama, España, Patricios and Chacabuco). Exotic birds like the House Sparrow (*Passer domesticus*) and the Rock Dove (*Columbia livia*) have the highest abundance in the urban parks near the city centre. The avian richness increases little by little towards the semirural sector

Table 3 Richness of vascular plants and birds

	Urban (n:5)					
	Reserve (n:1)	Parks (n:4)	Suburban (n:2)	Periurban (n:1)	Semirural (n:1)	Rural actual (n:2)
Number of plant species	245	124	102	237	334	156
Number of bird species	250	15	30	65	189	75
Exotic sps. %	29	60	51	41	45	22
Spontaneous sps. %	98	41	73	83	77	98
Exotic spontaneous sps. %	27	17	39	15	27	21
Native spontaneous sps. %	71	24	34	68	50	77
Annual spontaneous sps. %	15	14	25	22	15	23

n = number of sampled areas

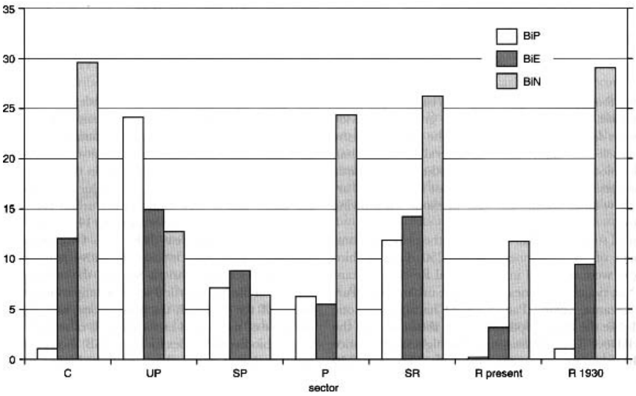

Fig. 5 Biodiversity indices of vascular plants

(Parque Indoamericano, Ribera Sur, Ezeiza and Los Robles) to decline again in the rural sector (Portela and Alsina).

Figure 5 shows the biodiversity indexes of vascular plants for each sector, comparing planted (BiP), exotic spontaneous (BiE) and native spontaneous species (BiN). Planted species are, as a result of afforestation practices on public land, significant in the urban parks and semirural areas. The highest values for self-growing exotic plants were found in the urban parks (BiE: 14.87) and the semirural sector (BiE: 14.22), the lowest in the present rural areas (BiE: 3.17). Outstanding are the high biodiversity indexes of native plants in the Natural Reserve Costanera Sur and the relatively low indexes in the present rural sector. Comparing the rural sector of the past with the present one, an important decrease of native and exotic plants is observed today.

The biodiversity index for birds (BiB) shows the same trend as that of native plants, with the highest value in the urban reserve (BiB: 42.67), a minimum in the urban parks (BiB: 3.89) and values increasing towards the semirural areas (BiB: 29.8). The index drops abruptly in the rural sector (BiB: 4.61–10).

This tendency reveals that agricultural practices have also altered birds' natural habitats.

In the eleven study areas along the urban-to-rural gradient, a strong positive correlation (0.84) between the biodiversity indexes of birds and native plants is recognised (Table 4). A negative correlation (−0.71) between woodlands (F) and grasslands (G) can be inferred, indicating the replacement of herbs and grasses by trees and shrubs.

In table 5 the similarity values for floristic composition between the defined sectors are presented. These values range from 0.2 to 0.46, which represents low and moderate similarity. Highest

Table 4 *Correlation coefficients between site and richness parameters (software STATISTICA PROGRAM)*

	W	G	F	BiN	BiA	BiE
Wetlands (W)	1.00					
Grasslands (G)	−0.39	1.00				
Woodlands (F)	−0.37	0.71	1.00			
Biodiversity index of native plants (BiN)	0.28	−0.57	0.36	1.00		
Biodiversity index of annual plants (BiA)	0.24	0.09	0.09	0.26	1.00	
Biodiversity index of exotic plants (BiE)	−0.24	0.27	0.08	0.03	0.19	1.00
Biodiversity index of birds (BiB)	0.54	−0.52	0.12	0.84	−0.10	0.01

Marked correlations are significant at $p < 0.05$

Table 5 *Similarity Index between areas considering vascular plants (spontaneous species)*

	Reserve	Urban Parks	Suburban sector	Periurban sector	Semirural sector	Rural present
Urban Parks	0.26					
Suburban sector	0.38	0.45				
Periurban sector	0.33	0.46	0.46			
Semirural sector	0.38	0.36	0.38	0.42		
Rural present	0.36	0.22	0.38	0.35	0.36	
Rural 1930	0.24	0.20	0.22	0.30	0.20	0.34

Values in grey represent moderate similarity (0.26–0.50), in white low similarity (0–0.25)

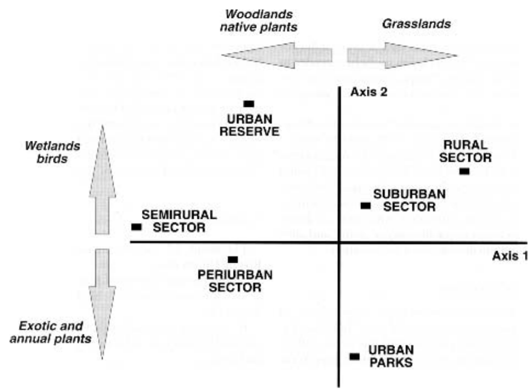

Fig. 6 Ordination of the sampled sectors produced by Principal Component Analysis (PCA)

similarity values were observed between the periurban area and the urban parks and between the periurban and the suburban areas. Dissimilarity is found between the rural area of the middle of the 20th century and urban, suburban and semirural sectors. The similarity index between past and present rural areas is moderate (0.34). The similarities of both rural sectors compared with the periurban sector are also moderate.

Figure 6 shows the Principal Component Analysis ordination of the eleven study areas. Axis 1 explains 42.71% of the variance and grasslands, woodlands and the biodiversity index of native plants are the principal components. They allow the separation of the studied sectors along this axis. The rural and the suburban sectors as well as the urban parks are characterised by the dominance of grassland. On the other hand the urban reserve and the semirural and periurban sectors show a high percentage of woodland and high biodiversity indexes of native plants.

The principal components of the axis 2 are "wetlands" and "the biodiversity index of birds", "the bio-diversity index of exotic" and "the biodiversity index of annual plants". This axis displays 29.16% of the variance. The urban reserve as well as the rural, surburban and semirural sectors show high values of "wetlands and the biodiversity index of birds". Urban parks and periurban sectors are characterised by high values of the "biodiversity index of exotic" and "biodiversity of annual plants".

Discussion

Species richness

In urban biodiversity gradient studies an increase of native plants from the city centre towards the periurban areas is generally recognized (Pickett et al. 2001; Sukopp 1998; Zipperer et al. 1991; Grapow a. Blasi 1998). In the present case the biodiversity indexes of native plants do not conform to these analyses. The Natural Reserve Costanera Sur, situated in the city centre, shows very high values of biodiversity of native plants and birds because its environment, due to its conservation state, is not altered. Therefore, it is suitable to be used as a reference for the comparison with the other sectors.

The high presence of birds in the reserve coincides with the results of Clergeau et al. (2001) who concluded that at regional and local scales, urban bird communities are independent of the bird diversity of adjacent landscapes. In the study conservation area bird richness depends very strongly on local and temporal features like the presence of lagoons, water level and habitat heterogeneity. Excepting the Natural Reserve Costanera Sur, the variation of bird richness from the city centre towards rural areas is consistent with the results of studies conducted in other cities, which also show a decrease towards the city centre (Melles et al. 2003).

On the other hand, all of the calculated biodiversity indexes in the suburban sector are very low. They can be related to the soil conditions of the Parque Indoamericano, which shows poor structure and presence of heavy metal (Hg/Pb) contamination (Kaplanski a. Gonzalez 1997).

Comparing the rural development over the years, today there is an outstanding plant impoverishment of native and also exotic species. Parodi (1930) found eight grasses and four herbs, which had been abundant in the Rolling Pampa in the first decades of the last century; today only two of them (*Bromus catharticus* and *Lolium multiflorum*) are frequent. Many endemic plants have also disappeared. In this sector of the pampa an adverse effect of intensive agriculture – particularly since 1970 – is described by many authors (Senigalesi 1991; Bertonatti a. Corcuera 2000; Casas et al. 2000). They observed that habitat fragmentation, soil erosion and changes in the water balance of the soils led to a decline in the abundance and diversity of native species and the invasion of exotic plants.

All biodiversity indexes of plants in the present rural areas are lower than the values observed in the urban parks. This indicates for the study area, that the effects of urbanisation on the plants have a lower influence than agriculture. This conclusion might be extended to the rest of the Rolling Pampa, considering that the analysed areas are representative for the pampean ecoregion.

Along the studied gradient the "intermediate disturbance hypothesis" (Connell 1978) could be verified for the semirural and periurban areas. According to this author, size and nature of anthropogenic and natural events influence species diversity, with a peak frequently being found where the disturbances are intermediate. The results are also coincident with those of Blair (1996), who found major diversity of native and alien species at intermediate levels of urbanisation.

Woodland encroachment

The results of the present study show absence of trees in present and former rural areas, but considerable tree cover in the other sectors, due to the anthropogenic afforestations, especially of Eucalyptus. At the end of the 19th century, the Eucalyptus was brought from Australia to Argentina.

Today, in the periurban areas of the Rolling Pampa, a spontaneous increase in the number of shrubs and trees can also be observed. Woodland encroachment is a process which is taking place in many prairie-savannah regions (Nowak et al. 1996). An aggressive North American tree, *Gleditsia triacanthos* (Honey locust), covers about 40% (277.52 ha) of the sampled periurban sector. This information was taken from the supervised classification of the Landsat ETM+-Image of this area. In North America the honey locust shows a wide natural range and its distribution increased through anthropogenic changes in land-use like pasture clearance and road construction (Blair 1990).

In the periurban areas of Buenos Aires this tree grows rapidly as a dense thicket at the beginning. Finally it forms a monophytic forest. Once planted as an ornamental element, it has invaded the pampa and other regions along streams since the second half of the 20th century (Morello a. Matteucci 2001; Marco a. Paez 2000). Over the years it has displaced the previous native grasslands. The honey locust tree is also an invader in Queensland (Australia) and was declared a plant that has to be eliminated (LAND PROTECTION 2006).

It is possible that climate and land-use changes influence invasion of the honey locust. Murphy et al. (1999) described a decrease of the winter chilling occurring in central Buenos Aires and its suburbs for the period 1911–1998. This trend was also observed by Damario a. Pascale (1995) for the rest of the country. The precipitation rate is actually more than 200–250 mm higher than it was at the beginning of the 20th century (Sierra et al. 1994). The absence or decrease of burning practices to clean grasslands, as were practiced in the past (Frangi et al. 1980), allows the *Gleditsia triacanthos* to expand, it is well known that in the plant succession fire is a selective force for the maintenance or backward motion to seral stages dominated by grasses (Morello 1980; Frangi et al. 1980).

Conclusions

The results of the present study confirm for the Rolling Pampa that:

A. the present cultivation practices are more dangerous for conservation of biodiversity than urban sprawl and

B. the urban renewal projects designed for nature can lead to adequate habitats to protect native plants and birds.

The implementation of further conservation areas in the Rolling Pampa together with more adequate agricultural practices could improve the current situation, rehabilitating and restoring native communities.

A reduction of mowing in urban to periurban areas, especially along the highways, could allow native plants to re-establish themselves and could help to prevent a further decline in richness. Also a policy of invasive plants control, as practised in Australia, has to be considered.

To increase the presence of birds in the urban and suburban parks it is advisable to enhance the presence of small lakes and ponds and to create a diversity of vegetation structure with tree and shrub layers within the grasslands. A good example is the urban Natural Reserve Costanera Sur, which contains 250 taxa of plants and birds respectively, most of them are native species. Compared to a range of urban, suburban and periurban areas this man-made wetland, covering only about 0.45 % of Buenos Aires and its surroundings, contains more than twice the amount of plants than the urban parks and suburbs, protecting a great part of the pampas diversity.

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Does Differential Access to Protein Influence Differences in Timing of Breeding of Florida Scrub-Jays *(Aphelocoma coerulescens)* in Suburban and Wildland Habitats?

Stephan J. Schoech and Reed Bowman

Abstract —Timing of breeding in Florida Scrub-Jays (*Aphelocoma coerulescens*) varies both within and between years. Social status and breeding experience may explain much of the within-year variation, but the availability of certain foods may partially explain between-year patterns. Scrub-jays in suburban habitats with access to unlimited human-provided foods breed earlier and with less between-year variation in timing of breeding than jays in wildland habitats. We hypothesized that those differences in timing of breeding result from access to human-provided foods in the suburban site. Human-provided food may influence timing of breeding by improving the overall body condition of females, or it may influence breeding by providing nutrients essential for breeding. If condition mediated, breeding females in the two habitats should differ in certain physiological parameters relative to time before egg laying and calendar date. If the effect is not related to body condition, we expect differences in pre-breeding females relative to calendar date, but not in relation to time before egg laying. To test those predictions, we measured plasma levels of total protein, calcium, luteinizing hormone, and estradiol. We also measured variables associated with body condition—body mass, a size-corrected condition index, and total body lipids. Most variables tended to increase with both days before laying and calendar date, except total body lipids, which decreased. Suburban females had higher levels of plasma protein relative to both days before egg laying and calendar date than female breeders in the wildland habitat. Luteinizing hormone differed between sites relative to calendar date but not days before laying. Our data suggest that suburban scrub-jays with access to predictable sources of high-quality human-provided foods accumulate endogenous protein that can be used to breed earlier. *Received 25 January 2002, accepted 14 June 2003*.

Keywords: Florida Scrub-Jay · timing of breading · supplemental food · diet · bird feeding · reproductive physiology

One of the most important decisions made by seasonally breeding animals is when to initiate their reproductive effort in a given year. Reproduction should occur when environmental factors are suitable (i.e. when resources are available to meet the energetic and nutritional demands of the mother and her growing young). To time reproduction to coincide with resources, animals rely on environmental cues that reliably predict favorable conditions. Many temperate-zone birds use an initial predictive cue, such as photoperiod, as the primary predictor of the onset of favorable conditions, but numerous other supplemental cues are known to play a role in fine-tuning the timing of breeding within the temporal window when environmental conditions are favorable for reproduction (for reviews see Farner 1985, Wingfield and Kenagy 1991, Wingfield et al. 1992, Wingfield and

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Farner 1993). Nonphotic cues may include temperature, rainfall, humidity, food abundance or food type (i. e. changes in availability of different types of food), and social environment (Wingfield 1983, Wingfield et al. 1994).

Many of those cues, especially climatic variables, are likely to affect relatively large spatial areas in the same manner. Therefore, birds within the same population, or even adjacent populations, will experience a similar set of cues and, as a result, initiate breeding at approximately the same time. Many temperate-zone birds display a relatively high degree of within-population breeding synchrony. Although photoperiod remains constant between years, climatic conditions often vary markedly, and that may lead to between-year variation in the onset of reproduction. Other cues, such as availability of food and social and demographic factors, can be expected to vary locally leading to both between- and within-population variation in timing of breeding.

In natural habitats, Florida Scrub-Jays (*Aphelocoma coerulescens*) exhibit considerable within- and between-year variation in clutch initiation date (see Woolfenden and Fitzpatrick 1984, 1990, 1996; Schoech 1996). However, scrub-jays that occur in a suburban matrix breed earlier than those in natural habitats and show relatively little between-year variation in clutch initiation dates (Bowman et al. 1998). Scrub-jays in suburban habitats have access to numerous sources of human-provided foods, including bird seed, pet foods, food waste, and peanuts provided specifically for scrub-jays by human residents that are unavailable to scrub-jays in natural habitats. Numerous studies, including two on Florida Scrub-Jays (Schoech 1996, Reynolds et al. 2003), have shown that access to supplemental food can advance laying date in birds (for review see Meijer and Drent 1999).

Over 30% of the diet of breeding female scrub-jays in suburban habitats during the one to two months preceding reproduction consists of human-provided foods (Fleischer et al. 2003). Those observations led us to conclude that access to human-provided food by suburban scrub-jays may be an important factor in the between-population differences in the timing of breeding. We hypothesized that year-round access to human-provided foods could improve the overall body condition of scrub-jays, thereby providing them with sufficient endogenous resources to initiate breeding shortly after stimulus by an initial predictive cue (i.e. to recrudesce their gonads and form ova following stimulation of the hypothalamo–pituitary–gonadal [HPG] axis). Jays in wildlands would have to wait for local resources to increase before initiating breeding. Alternatively, access to human-provided foods could provide jays with enough exogenous resources, regardless of their endogenous resources, so that once an initial predictive cue such as increasing daylength informed a female that the time to breed was approaching, breeding was unimpeded by limited local resources. If the endogenous resources hypothesis is correct, suburban females should have more endogenous resources than wildland females long before initial predictive cues occur and thus certain physiological measures of prebreeding females should differ between the two populations relative to both calendar date and days before egg laying (Fig. 1; H1). If the exogenous resource hypothesis is correct, then birds in both habitats are in similar physiological condition when initial predictive cues occur, but suburban females begin improving their body condition immediately because local resources are not limiting. In that case, physiological differences between females in the two habitats should exist relative only to calendar date (Fig. 1; H_0); in both populations, breeding should occur shortly after exogenous resources are sufficient to fuel the costs of reproduction.

To differentiate between those hypotheses, we examined several variables predicted to reflect the differences in food availability, and presumably diet, between the two populations. During the two months immediately preceding the breeding season we compared the following in female breeder Florida Scrub-Jays in the two populations: (1) total plasma protein, a measure of protein intake and an indicator of body condition; (2) total body lipids, another measure of condition; (3) body mass; (4) a morphometric index of body condition based on mass corrected for body size; (5) plasma levels of calcium, an essential component for eggshell production and a number of physiological processes; and plasma levels of two reproductive hormones, namely (6) luteinizing hormone (LH), and (7) 17β-estradiol (E_2).

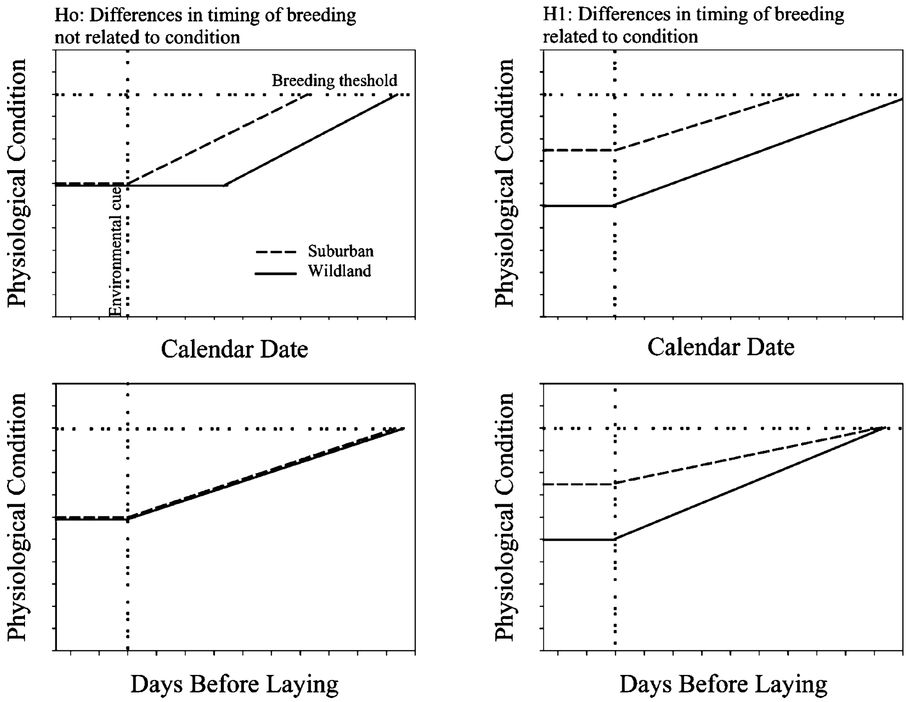

Fig. 1 Hypothetical trends in various physiological parameters relative to days before egg laying and calendar date in two Florida Scrub-Jay populations, one in suburban habitats that consistently breeds earlier than a population in wildland habitats. The two hypotheses are Ho (physiological parameters do not differ between sites and are unrelated to differences in timing of breeding; therefore, they should differ only in respect to calendar date and not days before laying) and H1 (physiological parameters differ between sites and are related to differences in timing of breeding; therefore, they should differ in respect to both calendar date and days before laying.)

Methods

Study population.—Florida Scrub-Jays were studied in a natural scrub preserve at Archbold Biological Station (hereafter wildlands") (27°10′N, 81°21′W, elevation 38–68 m) and at a nearby (<8 km) suburban development in Highlands County, Florida, during the 1998 breeding season. All scrub-jays in both populations were banded with a unique combination of color and federal aluminum bands. The sex, breeding status (breeder or nonbreeding helper), and group association of all scrub-jays in both populations were known from ongoing long-term studies. All nests were located during the nest-building stage or incubation (for more information on the wildland population, see Schoech et al. 1991, 1996; Mumme 1992; Schoech 1996; for further details on the suburban population see Bowman et al. 1998, Bowman and Woolfenden 2001).

Capture and blood sampling.—Beginning in mid-January, scrub-jays from the two populations were captured in Potter traps baited with peanuts. Traps were monitored continuously and each scrub-jay was removed within seconds of capture. Only female breeders were used and individuals were sampled only once between 21 January and 21 April 1998. To control for diel fluctuations in the variables of interest, all scrub-jays were captured between 0700 and 1100 hours EST. Immediately following removal from the trap, a blood sample was collected from the brachial vein after venipuncture with a 26 gauge needle. Samples were stored on ice until returned to the laboratory (within 1–4 h). Samples were centrifuged, and the plasma harvested for later assay. A small volume (20 μL) of plasma was retained to evaluate plasma protein levels (see below). The remaining volume was frozen and stored at −20°C until shipped to Indiana University for assay of plasma levels of calcium and estradiol or to Princeton University for assay of luteinizing hormone.

Immediately after blood samples were taken, scrub-jays were anaesthetized and total body lipids were determined (see below for details). While recovering from the anaesthetic, individuals were weighed to the nearest 0.1 g with an Avinet (Dryden, New York) spring balance, and several morphological variables, including wing-chord and tail length, a series of bill measurements, head-breadth, and overall head plus bill length, were measured to the nearest 0.1 mm. Birds were released at their capture site after the effects of anaesthesia had completely abated, usually within 0.5 h of initial capture. All procedures were sanctioned by the Bloomington Institutional Animal Care and Use Committee of Indiana University.

Physiological parameters.—Plasma protein concentration was measured with a handheld clinical refractometer (Model A 300 CL, Atago Company, Kirkland, Washington) that allows determination of the amount of dissolved solute in a small volume of plasma (20 μL), based upon the degree to which light passing through the sample is refracted. To validate protein estimates using refractometry, protein levels were also determined for 10 samples using the Biuret method of protein determination (Sigma Total Protein diagnostic kit, procedure 541; Sigma-Aldrich Corporation, St. Louis, Missouri). Protein levels determined by the two methods were similar (linear regression, $r^2 = 0.69$, $F = 17.3$, $\mathrm{df} = 8$, $P < 0.01$).

Immediately after blood samples were taken, body lipids were assessed with the total body electrical conductivity (TOBEC) method. This allows determination of an animal's lean mass and from this, total body lipids can be estimated (Kenagy and Barnes 1988, Walsberg 1988, Roby 1991, Schoech 1996). Accurate use of the instrument requires that subjects are positioned uniformly and kept in the same position for the duration of the scanning procedure. Therefore, all scrub-jays were anaesthetized with Metophane (inhaled) and held in a nylon stocking during the procedure.

When calibrated, that method is useful for within-species comparisons of relative body lipid content, although the technique has been criticized (see Morton et al. 1991, Asch and Roby 1995). Calibration of the instrument for a given species necessitates that on one occasion a number of individuals be scanned, killed, and total body fat content measured directly with a lipid extraction technique (e.g. chloroform or ether in a Soxhlet apparatus). For this study, an equation derived from the congeneric Western Scrub-Jay (*A. californica*) was used to estimate total body lipid in Florida Scrub-Jays (see Schoech 1996 for calibration, validation, and methodological details).

Body mass divided by a linear measurement is often used as an index of condition. Previously, discriminant function analysis found that overall head plus bill length was the best predictor of sex and, therefore, body size in this visually monomorphic species (S.J. Schoech unpubl. data). Accordingly, body mass was divided by that linear measurement as an index of body condition.

Determination of plasma calcium concentration was made with a kit obtained from Sigma Diagnostics (procedure 587). Calcium reacts in a dose dependent fashion with o-cresolphtalein complexone to produce a red complex. The intensity of the resulting color of the solution is directly proportional to the calcium concentration in the plasma sample. The color intensity was determined with a spectrophotometer with absorbance set at 575 nm.

Luteinizing hormone was assayed in the laboratory of Dr. T. Hahn at Princeton University. The assay is a post-precipitation double antibody RIA that uses purified chicken LH as a standard and rabbit-reared antisera against LH (Follett et al. 1972, 1975; Sharp et al. 1987). Components were provided by Dr. P. Sharp of the Agricultural Research Council, Roslyn, Scotland. Radiolabeling of LH with ^{125}I was done using the chloramine-T method. All samples were run in a single assay in duplicate with volumes that ranged from 10 to 20 μL and averaged 18.9 μL. Intra-assay coefficient of variation was 5.5%.

Plasma levels of the sex steroid hormone estradiol were measured in the laboratory of Dr. E. Ketterson at Indiana University following transport on dry ice from the field site in Florida. This competitive binding radioimmunoassay has been conducted in the Ketterson laboratory for several years (Ketterson et al. 1992; Schoech et al. 1998, 1999). Further details on the methods and reliability criteria can be found elsewhere (see Wingfield and Farner 1975, Ball and Wingfield 1987, Schoech et al. 1991).

In brief, assay of E_2 was conducted in the following manner. Approximately 2,000 counts $\min^{-1}$ of radiolabeled E_2 were added to the unknown sample to allow calculation of the percentage of hormone recovered following extraction and column chromatography (see below). Following overnight equilibration of the labeled hormone with the plasma sample, steroids were extracted from the aqueous phase using anhydrous diethyl ether. Snap freezing of the ether and aqueous mixture in acetone super-chilled with dry ice allows the steroid-containing organic phase to be easily decanted. Samples were then dried down under nitrogen gas and reconstituted in an ethyl acetate and iso-octane mixture (10:90 by volume). That solution was then forced into celite-glycol packed minicolumns with nitrogen gas. Steroids were separated from one another by adding ethyl acetate and iso-octane mixtures of increasing polarity that were moved through the columns with nitrogen gas. Upon collection of the desired fraction, samples contained within the elute were dried, reconstituted in buffer, and subjected to a standard competitive-binding assay.

All estradiol samples were run in a single assay. Mean plasma volume used was 178 μL; recovery averaged 79.2% and intra-assay variation was 6.6%. Tritiated E_2 was obtained from New England Nuclear Research Products (Boston, Massachusetts), standard from Sigma, and antibody from Arnel Products (New York, New York).

Data analyses.—Between-population differences in timing of first-clutch initiation were determined using a t-test. To distinguish between our alternative hypotheses, all physiological and morphometric measures between the two populations were compared relative to first-clutch initiation date (days before laying) and calendar date. Regression was used to test data for linearity; most data were nonlinear but were successfully linearized by plotting the natural log of each variable. Analyses of covariance (ANCOVA) were then used on the transformed data with site (suburbs or wildlands) as the factor, days before laying or calendar date as the covariate, and physiological or morphological measures as the dependent variable to test whether the measures examined differed between sites or across time. In a few instances, natural-log transformation failed to linearize data, but ANCOVA were performed on the nonlinear data. ANCOVA can be fairly robust to violations of assumptions, especially experimental designs with heterogenous slopes (Klockars and Beretvas 2001). Our data met all other assumptions, our sample sizes were relatively large, and the power of our statistical tests exceeded 0.8; thus we felt that analytical approach was justified. Interactions between independent factors were tested for, and where no interaction existed, data were reanalyzed without the interaction term. When no interaction existed, data reported are for models without the interaction term. The overall fit ($adj R^2$) for each model is included. All analyses were conducted with SPSS for WINDOWS (SPSS 1999).

Results

In total, we obtained blood samples and measurements from 35 and 46 females from the suburban and wildland populations, respectively. Of those, 32 females in the suburbs and 36 females in the wildlands eventually laid eggs. Samples from the suburban population were collected between 21 January and 25 March and from 88 days to 1 day before egg laying. In the wildlands, samples were obtained between 29 January and 21 April and from 90 days to 1 day before egg laying.

Timing of reproduction.—Florida Scrub-Jays in suburban habitats initiated first clutches significantly earlier than jays in the natural habitat (Student's t-test, $t = -6.52$, $P < 0.001$, $n = 68$; Fig. 2). Of the 32 females that laid eggs in the suburbs, the mean laying date of the first egg in the first clutch was 24 March ±2.3 (SE) days, and laying dates of first eggs ranged from 5 March through 23 April. In contrast, of the 36 females that laid eggs in the natural habitat, the mean laying

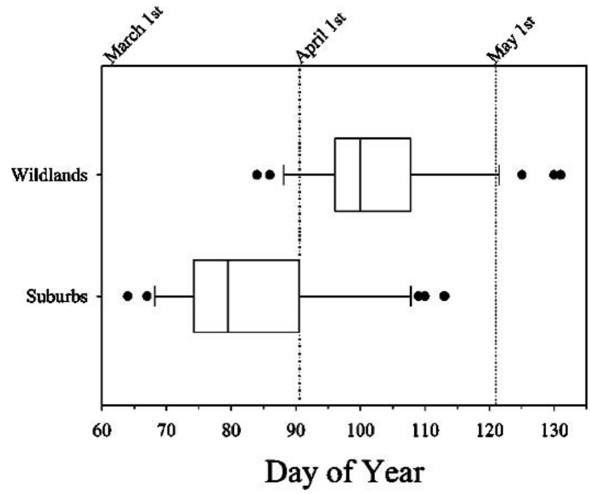

Fig. 2 Horizontal box plot of the timing of firstlaid clutches of the Florida Scrub-Jay populations in suburban and wildland habitats. Vertical lines within the box represent the median laying date. The box represents the middle 50% of the population and error bars represent the first 10 and 90% of the populations. Filled circles represent outliers that began laying very early or very late

date was 13 April ±1.95 (SE) days, and laying dates of first eggs ranged from 25 March to 11 May. Although dates of breeding differed, the distribution of laying dates within each population was similar. In each population, breeding was relatively synchronous with 50% of all female breeders laying within 15 days of the first egg laid in each population (Fig. 2).

Plasma protein.—Plasma protein levels increased as laying date neared ($F = 57.4$, df = 1 and 66, $P < 0.001$) and differed by site ($F = 5.0$, df = 1 and 66, $P = 0.03$). In addition, a significant interaction existed ($F = 5.9$, df = 1 and 63, $P = 0.02$; Fig 3A). The corrected model was highly significant ($adj\ R^2 = 0.49$, $P < 0.001$). Plasma protein levels were higher in the suburbs long before egg laying but increased more slowly than in the wildlands. Plasma protein levels increased with calendar date ($F = 25.7$, df = 1 and 80, $P < 0.001$) and were significantly higher in the suburbs than in the wildlands ($F = 4.4$, df = 1 and 80, $P = 0.04$). No interaction between site and calendar date existed ($F = 1.2$, df = 1 and 78, $P = 0.27$; Fig. 4A). The corrected model was significant ($adj\ R^2 = 0.23$, $P < 0.001$).

Several previous publications have equated plasma protein levels with overall body condition (Dawson and Bortolotti 1997, Ots et al. 1998). We, therefore, examined the relationship between plasma protein levels and body condition index. Plasma protein levels were significantly correlated with the body condition index in both populations (Pearson correlation: natural habitat, $r = 0.52$, $P = 0.01$; suburban habitat, $r = 0.37$, $P = 0.04$).

Total body lipids.—Total body lipids decreased as laying date neared ($F = 4.0$, df = 1 and 66, $P = 0.05$), but did not differ by site ($F = 0.0$, df = 1 and 66, $P = 0.99$), nor was there a significant interaction ($F = 0.5$, df = 1 and 63, $P = 0.83$; Fig. 3B). However, the corrected model was not significant ($adj\ R^2 = 0.03$, $P = 0.14$). We found no significant variation in total body lipids relative to calendar date ($F = 0.6$, df = 1 and 80, $P = 0.43$), site ($F = 0.1$, df = 1 and 80, $P = 0.83$), nor was there an interaction ($F = 1.3$, df = 1 and 77, $P = 0.27$; Fig 4B). The corrected model was not significant ($adj\ R^2 = -0.02$, $P = 0.73$)

Body condition and mass.—Body condition increased as laying date neared ($F = 46.8$, df = 1 and 64, $P < 0.001$) but did not differ between site ($F = 0.6$, df = 1 and 66, $P = 0.43$) nor was there a significant interaction ($F = 1.8$, df = 1 and 61, $P = 0.19$; Fig. 3C). The corrected model was significant ($adj\ R^2 = 0.41$, $P < 0.001$). Body condition increased with calendar date ($F = 8.2$, df = 1 and 78, $P = 0.01$) but did not differ by site ($F = 0.3$, df = 1 and 78, $P = 0.60$) nor was there a significant interaction (F = 0.1, df = 1 and 75, $P = 0.73$; Fig. 4C). The corrected model was significant ($adj\ R^2 = 0.17$, $P = 0.02$).

Body mass increased as laying date neared ($F = 50.8$, df = 1 and 66, $P < 0.001$) but did not differ by site ($F = 2.1$, df = 1 and 66, $P = 0.15$) nor was there a significant interaction

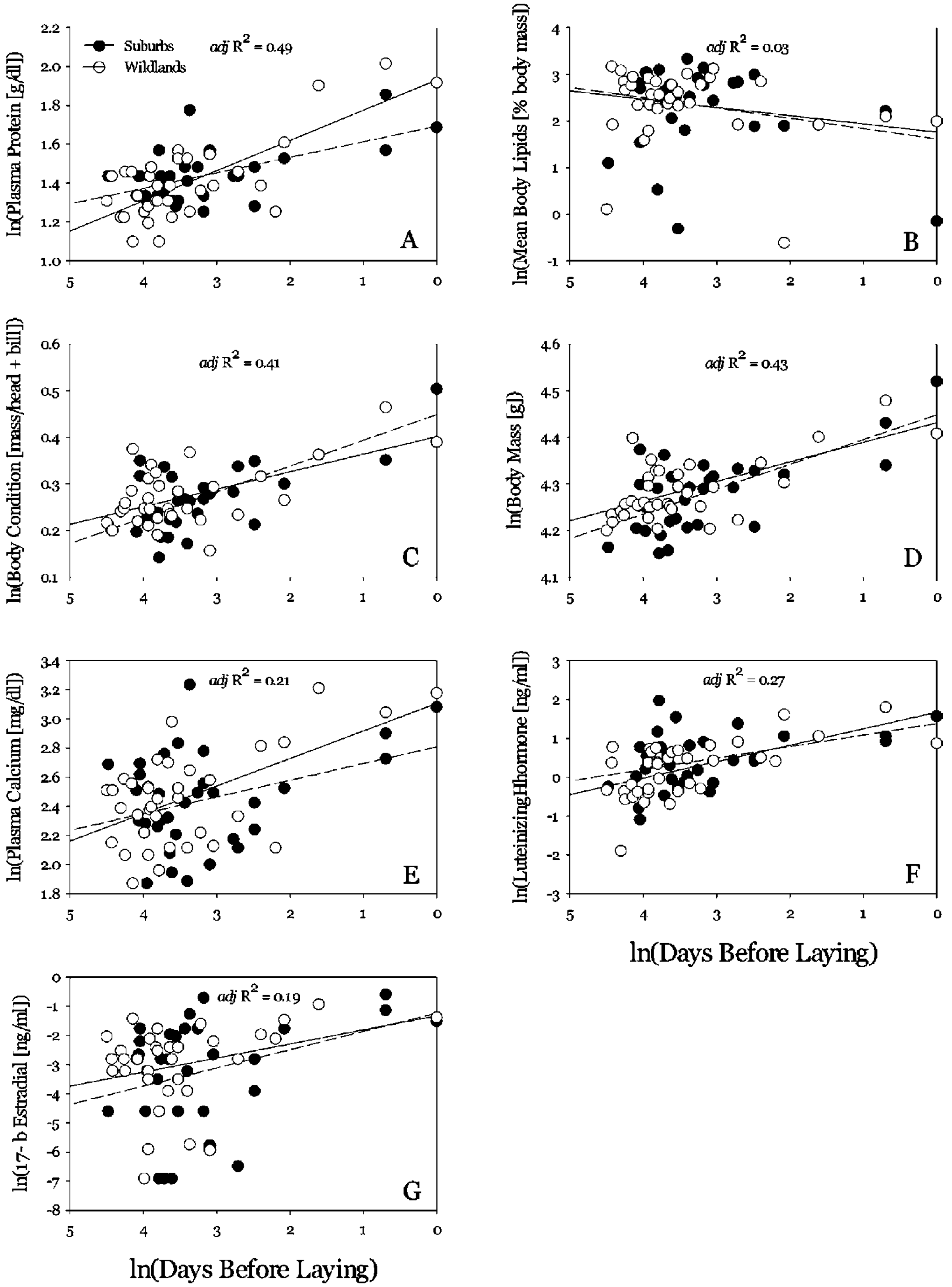

Fig. 3 A between-population comparison of female Florida Scrub-Jays relative to days before egg laying and (A) plasma protein levels, (B) total body lipids, (C) a size-corrected condition index, (D) body mass, (E) plasma calcium levels, (F) plasma luteinizing hormone levels, and (G) plasma estradiol levels. Suburban females are represented by closed circles and wildland females are represented by open circles. The dashed line represents the best-fit linear function for the suburban data and the solid line represents the same for wildland females

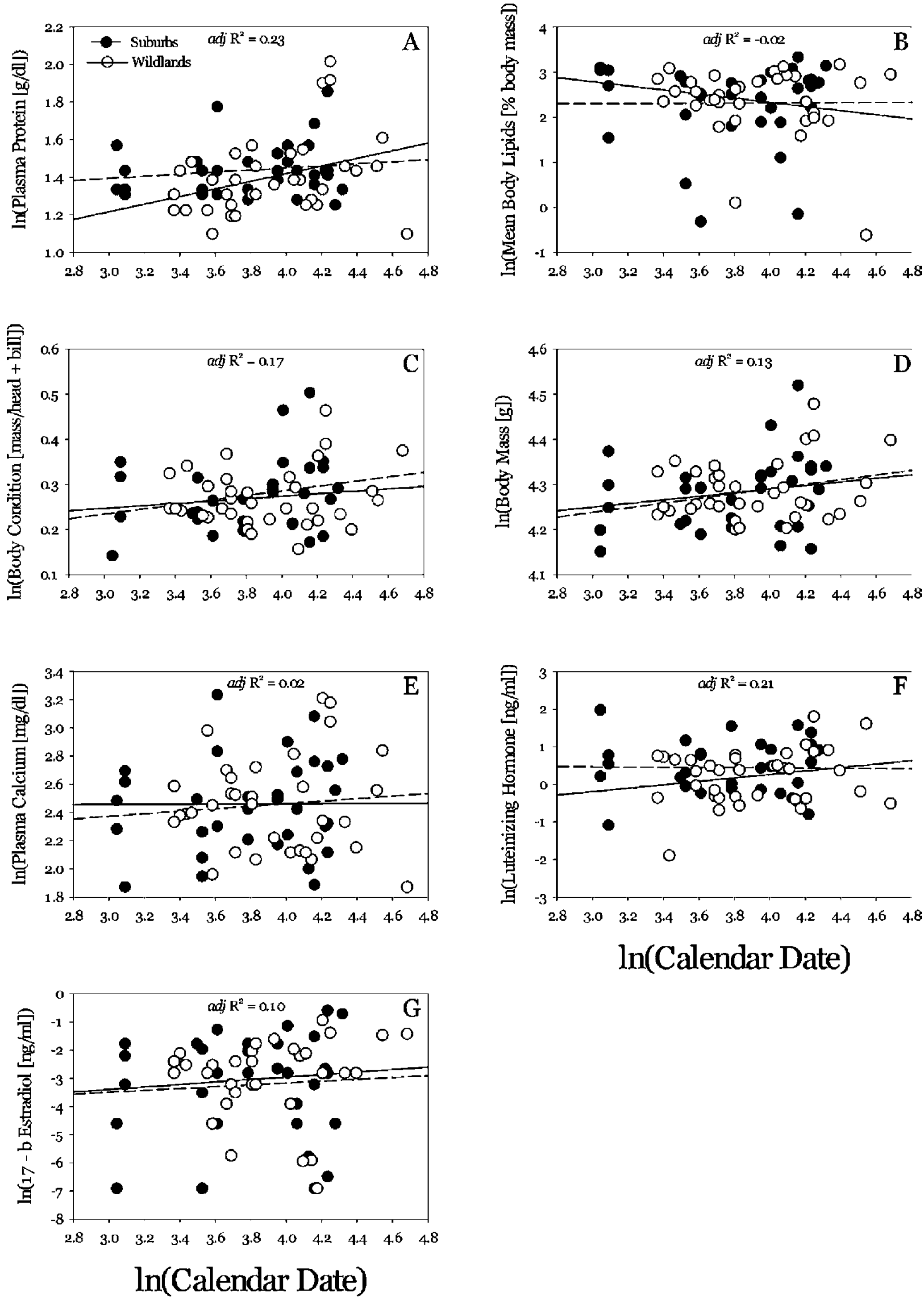

Fig. 4 A between-population comparison of female Florida Scrub-Jays relative to calendar date and (A) plasma protein levels, (B) total body lipids, (C) a size-corrected condition index, (D) body mass, (E) plasma calcium levels, (F) plasma luteinizing hormone levels, and (G) plasma estradiol levels. Suburban females are represented by closed circles and wildland females are represented by open circles. The dashed line represents the best-fit linear function for data from suburban females and the solid line represents the same for wildland females

($F = 0.7$, df $= 1$ and 63, $P = 0.41$; Fig. 3D). The corrected model was significant ($adj\ R^2 = 0.43$, $P < 0.001$). Body mass increased with calendar date ($F = 13.4$, df $= 1$ and 80, $P < 0.001$) but did not differ by site ($F = 0.4$, df $= 1$ and 80, $P = 0.52$) nor was there a significant interaction ($F = 0.4$, df $= 1$ and 77, $P = 0.52$; Fig. 4D). The corrected model was significant ($adj\ R^2 = 0.13$, $P = 0.02$)

Plasma calcium.—Plasma calcium increased as laying date neared ($F = 19.0$, df $= 1$ and 67, $P < 0.001$) but did not differ by site ($F = 0.6$, df $= 1$ and 67, $P = 0.46$) nor was there a significant interaction ($F = 1.1$, df $= 1$ and 64, $P = 0.29$; Fig. 3E). The corrected model was significant ($adj\ R^2 = 0.21$, $P < 0.001$). Plasma calcium increased with calendar date ($F = 4.1$, df $= 1$ and 80, $P = 0.04$) but did not differ by site ($F = 0.3$, df $= 1$ and 80, $P = 0.57$) nor was there a significant interaction ($F = 1.1$, df $= 1$ and 78, $P = 0.31$; Fig. 4E). The corrected model was not significant ($adj\ R^2 = 0.02$, $P = 0.14$).

Luteinizing hormone.—Luteinizing hormone increased as laying date neared ($F = 24.3$, df $= 1$ and 66, $P < 0.001$) but did not differ by site ($F = 0.7$, df $= 1$ and 66, $P = 0.40$) nor was there a significant interaction ($F = 0.3$, df $= 1$ and 63, $P = 0.76$; Fig. 3F). The corrected model was significant ($adj\ R^2 = 0.27$, $P < 0.001$). Luteinizing hormone increased with calendar date ($F = 21.0$, df $= 1$ and 80, $P < 0.001$) and was significantly higher in the suburbs ($F = 6.7$, df 1 and 80, $P = 0.01$), but there was no significant interaction ($F = 0.4$, df $= 1$ and 77, $P = 0.53$; Fig. 4F). The corrected model was significant ($adj\ R^2 = 0.21$, $P < 0.001$).

17β-Estradiol.—Estradiol increased as laying date neared ($F = 7.9$, df $= 1$ and 63, $P = 0.01$) but did not differ by site ($F = 1.0$, df $= 1$ and 63, $P = 0.33$) nor was there a significant interaction ($F = 0.1$, df $= 1$ and 60, $P = 0.72$; Fig. 3G). The corrected model was significant ($adj\ R^2 = 0.19$, $P = 0.02$). We found no significant variation in estradiol relative to calendar date ($F = 2.7$, df $= 1$ and 77, $P = 0.10$), site ($F = 0.5$, df $= 1$ and 77, $P = 0.49$), or their interaction ($F = 0.2$, df $= 1$ and 74, $P = 0.67$; Fig 4G). The corrected model was not significant ($adj\ R^2 = 0.10$, $P = 0.25$).

Discussion

We found that scrub-jays in the suburbs nested earlier than jays in wildlands and that plasma levels of protein differed between sites relative to both days before egg laying and calendar date. At least for plasma protein, those patterns are consistent with our hypothesis that in suburban habitats human-provided foods improve the condition of adult scrub-jays, providing them with additional endogenous resources that enable them to breed earlier. However, we found no other condition-related differences between sites.

Few supplementation studies have attempted to examine the relative importance of different nutrients on the timing of breeding. Of those, all have attempted to compare a high protein diet with a high energy diet, and although supplemented birds began breeding earlier than controls in all cases, the specific effects of different nutrients could not be determined (Bolton et al. 1992, Nager et al. 1997, Ramsay and Houston 1997). Human-provided foods make up 30% of the diet of prebreeding females in suburbs and of that, peanuts constitute 30% (Fleischer et al. 2003). With the exception of peanuts, which are relatively rich in proteins (Karasov 1990), we have few data on the nutritional composition of the human-provided foods consumed by suburban scrub-jays. In contrast, adult females in wildlands depend heavily on acorns cached during the previous fall (Woolfenden and Fitzpatrick 1984, 1990). They may augment that diet with arthropods and small vertebrates, but both of those food sources can be scarce in winter. Although acorns also are relatively rich in proteins, tannins in acorns may reduce the ability of scrub-jays to assimilate proteins (Koenig 1991, Fleck and Woolfenden 1997). Thus, it seems reasonable to postulate that jays in wildlands might be protein limited during years in which environmental conditions delay or decrease the availability of protein-rich food sources, such as arthropods; whereas jays in suburbs have predictable, essentially *ad libitum*, access to high-protein foods.

In both populations, plasma protein increased with calendar date and as laying neared; however, the rate of increase was slower in suburbs. In many birds, elevated levels of plasma protein as laying approaches reflects an increase in vitel-logenin (White 1991). This protein is produced in the liver in response to endocrine signals (primarily estradiol), transported to the ovary via the bloodstream, and incorporated into the yolk to be mobilized later as an essential nutrient by the developing embryo (Jackson et al. 1977, Ho 1991, Deeley et al. 1993, Carey 1996). It may be that plasma protein levels that exist at the onset of vitellogenesis influence the rate of additional protein production; thus suburban birds would likely add protein at a slower rate (Figs. 3A and 4A).

We also found that LH was higher in the suburbs relative to calendar date, but not days before laying. That pattern suggests that site-specific differences in LH are a result of differences in timing of breeding rather than a cause of the difference. In many seasonally breeding animals, environmental changes cause an increase in hypothalamic secretion of gonadotropin-releasing hormone (GnRH) that results in increased pituitary release of LH that, in turn, induces increases in other reproductive hormones (for reviews see Wingfield and Kenagy 1991, Wingfield and Farner 1993). Both field and laboratory studies confirm that changes in environmental conditions that precede conditions favorable for reproduction precipitate those endocrine changes (for reviews see Immelmann 1971, 1973; Farner and Follett 1979; Wingfield and Kenagy 1991; Wingfield et al. 1993, 2000; Hahn et al. 1997). Dietary restrictions are well known to negatively affect the reproductive axis (see Tanabe et al. 1981, Hocking et al. 1987, Katanbaf et al. 1989, Wade and Schneider 1992, Hocking 1996, Hocking and Bernard 1998, Renema et al. 1999). In wildlands, during years in which peak food availability is delayed, those endocrine changes may be dampened, delaying gonadal recrudescence and oogenesis and, in turn, delaying breeding (Balthazart 1983, Wingfield and Farner 1993, Schoech and Lipar 1997). We suggest that the temporal differences in LH between the two sites simply reflect the differences in timing of breeding between the two sites that result from differential access to protein-rich foods.

Surprisingly, we found few differences in condition-related or other micronutrient measures between sites relative to calendar date, even though many of those measures tended to increase during the season. This suggests that those variables, although associated with breeding, have a relatively small influence on timing of breeding. Some researchers have postulated that plasma protein levels reflect overall body condition. For example, Ots et al. (1998) concluded that urban-dwelling male Great Tits (*Parus major*), which had higher plasma protein levels than nearby rural birds, were in better condition. Similarly, Dawson and Bortolotti (1997) suggested that plasma protein values can be used as an index of condition in American Kestrels (*Falco sparverius*). Our data show that plasma protein levels were correlated with body condition, but body condition *per se* did not appear to be affected by food supplementation nor appear to be related to the differences in timing of breeding between sites. It is possible that other environmental conditions in suburban habitats also affect body condition, independently of plasma protein levels.

It is somewhat surprising that we found no between-population differences in plasma calcium levels or total body lipids. In the suburbs, calcium is easily available; all roads are built upon a crushed shell base. Calcium is likely highly limited in the low pH environments typical of wildland scrub. Although natural calcium sources may be scarce in scrub habitats, birds may know where to find it when necessary during the breeding season. It may only be when natural sources of calcium are reduced through anthropogenic effects that reproduction is limited by calcium (Graveland et al. 1994). Schoech (1996) found that scrub-jays provided supplemental food in wildlands bred earlier and had higher levels of total body lipids than did control birds. Peanuts are relatively rich in both protein and lipids (Karasov 1990); thus it is somewhat surprising that we did not see a similar pattern in suburban jays.

Equally intriguing is our finding that total body lipids appeared to decline prior to egg laying in both populations (see Fig. 3B), although our model explained relatively little of the variation in the data. Whereas some avian species rely on endogenous reserves, most passerines appear to depend on exogenous resources through increased food intake to fuel reproduction (Perrins 1970, 1996). That apparent dichotomy in strategies to fuel reproduction, termed capital versus income strategies (Drent and Daan 1980, Meijer and Drent 1999), may be better viewed as a continuum between the two extremes of total reliance on either endogenous or exogenous resources. For example, Williams and Ternan (1999) found that female Zebra Finches (*Taeniopygia guttata*) reduce locomotor activity rather than increase food intake during oogenesis, and it is likely that other species meet part of the increased energetic or nutritional demands of oogenesis by reducing their daily energetic expenditures by reallocating time spent in various activities. Prebreeding female scrub-jays in suburban habitats reduce the proportion of time they spend foraging while increasing time perched (Fleischer et al. 2003), which suggests that when provided with supplemental food, they too reduce and reallocate their daily energy expenditure. Similarly, food-supplemented wildland jays spend less time foraging than unsupplemented jays (S. J. Schoech and R. Bowman unpubl. data).

Given the proximity of these two populations, it seems unlikely that the difference in timing of breeding could be explained by differences in any of the seasonally variable abiotic environmental factors used by birds to time reproduction (e.g. photoperiod, temperature, rainfall, or relative humidity), though suburban light pollution and heat island effects could be important and should be considered. However, observations that the phenology of oak leaf-out does not differ between sites (R. Bowman unpubl. data) are inconsistent with the hypothesis that temperature differences exist between the suburban and wildland study sites. Suburban habitats are lighter at night than the wildlands (R. Bowman and S. J. Schoech unpubl. data). Although constant dim light can affect biological rhythms in birds (Kumar et al. 2000), there is little information on whether low levels of nighttime light accelerate the seasonal physiological changes that occur in breeding birds.

The data presented here support the hypothesis that differential access to exogenous protein sources between suburban and natural habitats underlies the observed between-population differences in timing of breeding. Variation of plasma protein explained a much higher proportion of variation in breeding date than did models for other variables; however, it seems likely that other environmental effects, in addition to food, may influence timing of reproduction. For example, human-provided foods are spatially and temporally predictable, and as a result, suburban scrub-jays are much more efficient foragers than are jays in wildland habitats. Foraging efficiency may be a perceptual cue that jays use to predict future levels of resource abundance. In tits, capture rates of lepidopteran larva likely provide a cue for initiating breeding because rising capture rates accurately predict the increase in resource availability (Perrins 1991). However, human-provided foods are unlikely to be a reliable cue of future foods that are suitable for nestlings, and as a result, the advancement of reproduction might have negative effects on the fitness of both parents and offspring, as has been observed in Blue Tits (*Parus caeruleus*; Nilsson 1994) and European Coots (*Fulica atra*; Brinkhof 1995). Clearly, further research is needed to distinguish between those competing hypotheses, but our data suggest a role for protein in the decision of when a female initiates her clutch.

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Creating a Homogeneous Avifauna

Robert B. Blair

Abstract I compared birds on urban gradients in two ecoregions of the United States by censusing summer resident bird populations at six sites in central California's coastal chaparral and southwest Ohio's eastern broadleaf forest. These sites represented comparable gradients of urban land-use which ranged from relatively undisturbed to highly developed and included biological preserves, recreational areas, golf courses, residential neighborhoods, office parks (or apartment complexes), and business districts. Species richness and Shannon diversity peaked at moderately disturbed sites and were significantly correlated between comparable land-use types in the two ecoregions. Bird abundance and biomass peaked at moderately disturbed sites as well but were not significantly correlated between ecoregions. The pre-development bird species (assumed to be those found at the most undisturbed sites) dropped out gradually as the sites became more urban and the number of remaining species was significantly correlated between ecoregions with only three (CA) and one (OH) species remaining in the most urban sites. Taxonomically, the bird communities at the least urbanized sites (the preserves and recreational areas) were very different with an average Jaccard's index of species similarity of 0.065 while the most disturbed sites (the business district, apartment complexes/office park, residential areas and golf courses) had an average similarity of 0.185. The species assemblages along the gradient shifted gradually, demonstrating local extinction of and local invasion by different species as the sites become more urban.

Keywords: Conservation · diversity · extinction · gradient · homogenization · invasion · urbanization

1 Introduction

"The earth's biota is being homogenized rapidly as human activities increasingly introduce species outside their natural range." (Lodge 1993)

"Replacement of native endemic species with opportunistic and human commensal species could ultimately produce a relatively homogenized biosphere." (McKinney 1998)

"Global ecological homogenization, with commensurate loss of species, is the logical endpoint of the anthropogenic spread of exotics." (Lodge et al. 1998)

The world's biota is becoming homogenized at an unprecedented rate (Vermeij 1991, Vitousek et al. 1996, McKinney 1998) and statements such as those listed above reflect a shifting concern in the field of conservation biology. Human activities are aiding the dispersal and range expansion of

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tens of thousands of species (Lodge 1993), leading some scientists to wonder if we are creating a world of "cockroaches, pigeons, and starlings" (P. Ehrlich pers. comm.).

Ecologists studying the phenomenon of homogenization have focused on identifying characteristics common to invading species or invaded habitats (Reichard et al. 2001). Traits of invasive species include high dispersal rates, single-parent or vegetative reproduction, high genetic variability, phenotypic plasticity, large native range, eurytopy, polyphagy, and human commensalism (Lodge 1993). Attributes of invasible habitats include climatic similarity to the native habitat of the invading species, early successional stage, low species diversity, low predator richness, and recent disturbance (Lodge 1993). However, many exceptions to these traits exist and predicting which species successfully will invade or which habitats will be invaded is an inexact science (for brief introductions to the invasion literature, see Lodge 1993, Vitousek et al. 1996, Hobbs 1988).

Many of the traits of invasible habitats are the result of changes in the landscape brought about by human land-use and urbanization. However, few ecologists have asked if analogous communities in different ecoregions are actually becoming more homogeneous (i.e., similar) due to human land-uses. In other words, biotic homogenization is assumed to result from biological invasion, but this assumption rarely has been tested explicitly. In this study, I examine the avifauna of two ecoregions to test whether one type of human activity—urbanization—leads to more homogeneous biotic communities.

The establishment of non-indigenous bird species is peculiarly linked with humans. During the 1800s, western European settlers introduced many species throughout the world because they wanted birds from their homelands in their new environs. The American Acclimatization Society is infamous for its persistent attempts to introduce all of the species mentioned in Shakespeare's writings to New York City's Central Park and for its most notorious introduction, the European Starling (*Sturnus vulgaris*) (Wetmore 1964, Ehrlich 1988). North America's population traces from a release by the Society of 80 birds in 1890 and 40 more in 1891 (Wood 1924). A similar group on the West Coast unsuccessfully tried to establish starlings in the City Park of Portland, Oregon with releases in 1889, 1890, and 1892 (Long 1982). Currently, many non-indigenous species are becoming established through the intentional or inadvertent release of pets (Vanbael 1996, Garrett 1999). Less direct, but still human-induced, is the range expansion following manipulation of the landscape due to urbanization (Mills et al. 1989).

Case (1996) took a global perspective on the establishment of invasive bird species by examining introductions and extinctions on islands throughout the world. He found that the best correlate of successful invasion was the number of native species that had gone extinct on that island during the previous 3000 years. He interpreted the number of extinctions as a reflection of the direct and indirect effects of human activity, including habitat destruction, and invasion by deleterious organisms such as exotic predators, herbivores, and parasites. He also surmised that the correlation between introductions (a.k.a. invasions) and extinctions arose because native birds are usually more common to (or even restricted to) native habitats while introduced birds occupy open and disturbed habitats. In addition, Case suggested that conversion of land to urban and agricultural uses decreases habitat availability for native species and increases habitat availability for introduced species.

In this study, I examine the idea that urbanization leads to the local extinction of species and invasion by different species and, consequently, creates more homogeneous bird communities. Specifically, I extend previous research on an urban gradient in California's coastal chaparral ecoregion by comparing it to a similar gradient in Ohio's eastern broadleaf forest ecoregion (sensu Bailey 1994). In California, I found that urbanization affects not only the distribution and abundance of individual species of birds, but also community measures such as species richness, Shannon diversity, bird abundance, and bird biomass (Blair 1996, Blair 1999). Here, I examine (1) if the effects of urbanization on the avian community are similar in such disparate ecoregions, and (2) if increasing urbanization leads to an increasingly homogeneous avifauna.

2 Methods

2.1 Field Sites

The sites chosen for this study represent forms of development typical in an urban-suburban matrix and comprise the full gradient of human land-use intensity ranging from relatively undisturbed wildland to highly developed urban areas. In each ecoregion, the sites include a biological preserve, recreational area, golf course, residential neighborhood, office park (California) or apartment complex (Ohio), and business district.

In California, the six sites were located within a circle of 3 km radius centered at Stanford University near Palo Alto, California USA (37°20’ N, 122°15’ W; typical elevation $< 100\,\mathrm{m}$). These sites were judged to be as ecologically similar as possible prior to development. Assessment of historical ecological similarity was based on the presence of coast live oak (*Quercus agrifolia*), valley oak (*Q. lobata*) or blue oak (*Q. douglasii*) trees greater than 100 years of age and on accounts of predevelopment vegetation (Cooper 1926).

The California sites (Figs. 1 and 2) include:

(1) a biological preserve—The Jasper Ridge Biological Preserve is 512 ha property which has been used for biological research since the 1890s. Prior to 1974, portions of the preserve were leased for cattle grazing and public recreation. Currently, access is limited to researchers and visitors on docent-led tours. This site can be considered wildland with none of it being developed.

(2) an open-space recreational area—The Stanford University foothills preserve was heavily grazed until 1989 and currently is used by joggers, dog-walkers, hikers, and equestrians. Motorized access is limited to maintenance vehicles. This site, too, could be considered wildland because it has no development, though it has a history of human manipulation.

(3) a golf course—The Stanford University Golf Course borders the open-space recreation area and was originally developed in 1930. At the smallest scale, this site would also be considered wildland because it has few permanent structures, no permanent residents, and $< 4\%$ coverage in buildings and pavement. However, at a more appropriate scale, I view it as exurban or suburban.

(4) a residential area—This area consists of single-family detached houses which were built mostly between 1930 and 1960. The larger residential zone that includes this site shares borders with the open-space recreational area and the office park. This site is distinctly suburban with $\sim 42\%$ of the area covered in buildings and pavement.

(5) an office park—The Stanford Research Park was created in the mid-1950s. The development of the specific study site began in 1965 and consists primarily of low-rise office buildings, parking lots, and landscaped areas. The office park borders the residential area. This site can be considered borderline urban with approximately 66 percent of the area covered in buildings and pavement. However, it is lushly landscaped.

(6) a business district—Downtown Palo Alto is separated from Stanford University buildings by an approximately 1 km wide band of eucalyptus and oak woodland. This site is distinctly urban with 86% of the area covered in buildings and pavement.

In Ohio, the six sites were located within a 10 km radius of Miami University, Oxford, Ohio USA (39°30’ N, 84°45’ W; typical elevation $\sim$290 m). These sites were judged to be as ecologically similar as possible prior to development and originally consisted of climax beech-maple forest according to historical maps and reference materials (Gering 1999).

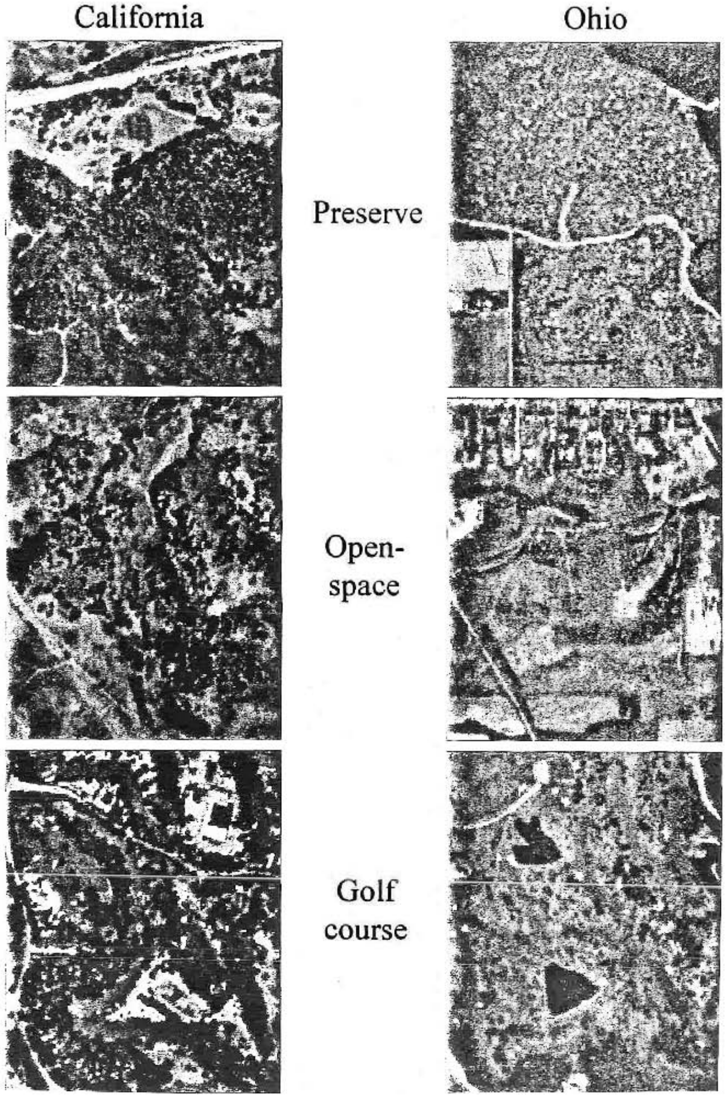

Fig. 1 Aerial views of the preserves, open-spaces, and golf courses in California and Ohio. Images were taken in August 1998 in California, in March 1994 in Ohio, and cover ~50 ha

The sites (Figs. 1 and 2) include:

(1) a nature preserve—Hueston Woods Nature Preserve is a 67 ha woodlot comprised of mature beech-maple forest with the exception of a narrow service road, a small parking area, and moderate-use dirt footpaths. A surrounding state park (975 ha) consists of young and mature deciduous forest, old fields, infrequent stands of conifers, and a 250 ha lake (Ray and Vankat 1982). This site can be considered wildland with little development.

(2) a recreational area—Peffer Memorial Park was purchased by Miami University as two separate parcels of pasture land and agricultural field in 1955 and 1966. A large portion of the 80 ha recreational park is managed by Miami University as a multiple-use trail system for residents of the local community. Currently, the vegetation is predominantly secondary

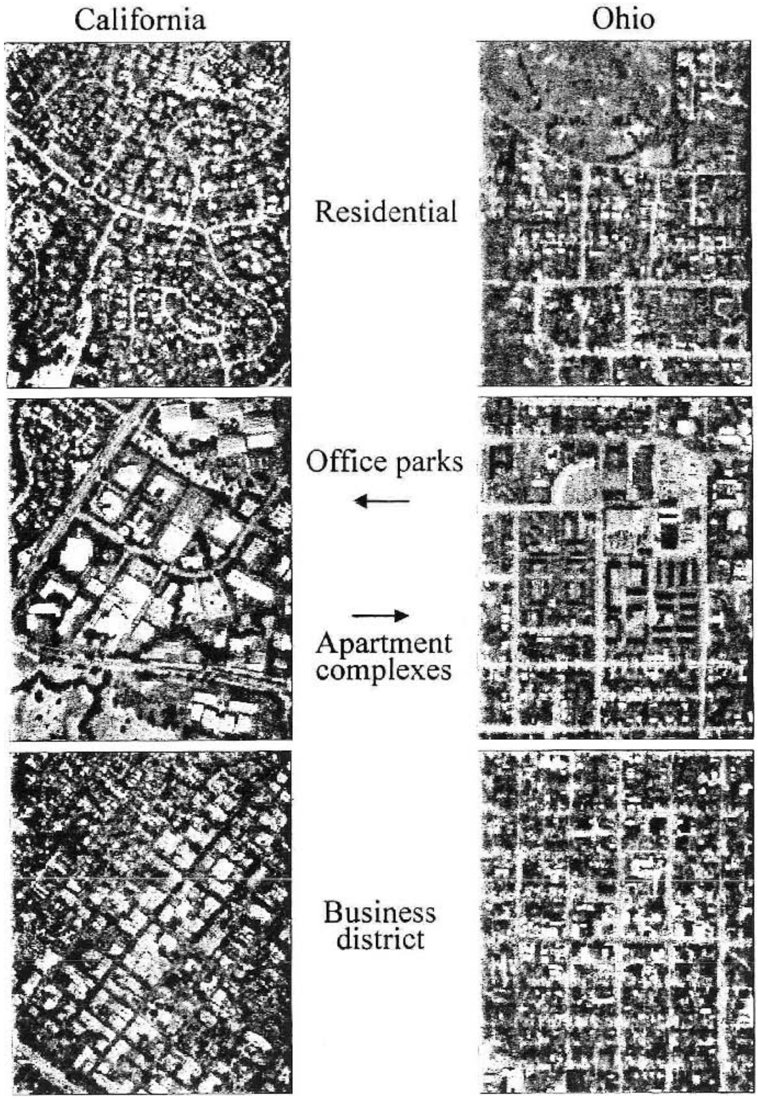

Fig. 2 Aerial views of residential areas, apartment complexes/office parks, and business districts in California and Ohio. Images were taken in August 1998 in California, in March 1994 in Ohio, and cover ∼50 ha

growth of low stature. This site, too, can be considered wildland with no development, though it has a history of being substantially manipulated by humans.

(3) a golf course—The 102 ha Hueston Woods Golf Course borders the nature preserve and was established in 1980. The course accommodates 10,000 players between March and November. Rough consists primarily of native trees (e.g., maple, beech) and grasses. The site has $< 2\%$ of its immediate area covered with buildings or pavement, but the surrounding landscape dictates that it should be considered exurban or suburban.

(4) a residential district—This residential community contains mostly detached, single-family houses built between 1960 and 1974. Dense lawns, gardens, shade trees, and native and ornamental flora are frequently located around the houses (Beissinger 1982). This site can be considered suburban with $\sim 32\%$ of its area covered with buildings and pavement.

(5) apartment developments—This area consists of multi-level apartment complexes and intervening parking lots with isolated trees and small-scale landscaping. The development of apartment complexes occurred between 1960 and 1980. We conducted our surveys in the apartment complexes rather than an office park, as used in California, because no office parks exist near Oxford, Ohio. The building height, density, layout, parking, and plantings are comparable to that of an office park. This site can be considered marginally urban with approximately 61 percent of its area covered with buildings and pavement.
(6) a business district—This is a 4×2 block city area consisting primarily of low-rise office buildings, parking lots, and one-half block of landscaped urban park. Initial construction of these properties began in the early to mid 1800s. This site is distinctly urban with $\sim$81% of the area covered in buildings and pavement.

2.2 Ordering of Gradients

I used two methods to rank sites from most natural to most urban. The first method was the Delphi Technique which can be used when hard data are lacking, but expertise in a system is available (Zuboy 1981, Blair 1994, Blair 1996). In California, faculty, staff, and students ($n = 14$) working with the Center for Conservation Biology at Stanford University were asked to rank the sites independently and anonymously from most natural to most urban, and to give the criteria they used. The results of the ranking were then distributed to the participants and the process was repeated until the group reached consensus on the ordering.

In Ohio, faculty and graduate students ($n = 12$) in the Department of Zoology at Miami University ranked the sites. The initial survey did not yield a group consensus, so the results were redistributed to participants along with measurements of canopy cover ($\mathrm{m}^2\,\mathrm{ha}^{-1}$) and tree basal area ($\mathrm{m}^2\,\mathrm{ha}^{-1}$) from each site. The participants again ranked the sites, taking into account the site-specific data as well as the results of the initial survey.

The second method was to order the sites in accordance with percent of area covered by buildings, pavement, lawn, grassland, and trees or shrubs. This produced an identical ranking to that of the Delphi Technique. Alberti (2001) discuss more quantitative techniques for measuring degree of urbanization.

2.3 Land-cover Quantification

For each study site, I visually estimated the area covered by buildings, pavement, lawn, grassland, and trees or shrubs in an approximately 50 m radius (depending on the particular scale of the map or photograph) centered at each survey point and then converted the total amount to percentage of site covered. In California, I estimated the cover of each land-use type from recent aerial photographs provided by Stanford University, the City of Palo Alto, and Jasper Ridge Biological Preserve. In Ohio, I used ortho-digitally corrected land-use maps available from the City of Oxford and aerial photographs available from the Department of Geography at Miami University.

2.4 Bird Surveys

I estimated the relative abundance of all perching or singing birds during peak breeding season using variable circular-plots at 16 survey points in a 4×4 matrix within each site where each point was at least 100 m from its nearest neighbor (Reynolds et al. 1980). In Hueston Woods and Peffer Memorial Park (both in Ohio), sampling points were located along foot trails instead of in a 4×4 matrix.

In California, points were surveyed a total of eight times in June and July, 1992, and four times in June, 1993. The open-space site was visited only six times in 1992 because it burned on 10 July. In Ohio, points were surveyed a total of eight times in June and July 1996, and four times in June 1997. Daily surveys began at dawn and continued until all 16 points at a site were covered, $\sim 2\,\mathrm{hrs}$. Each point was visited for five minutes. This method results in a relative abundance that estimates density (birds ha^{-1}) of all species within each site (see Blair 1996).

2.5 Community Measures

Species richness is the number of species seen at each site and is directly comparable among sites because sampling effort and area were equal at all sites. Species diversity was calculated with the Shannon Index (Shannon and Weaver 1949, Magurran 1988), which incorporates both species richness and evenness.

The abundance for all birds is the sum of the abundances for all species within a site. The biomass for all birds is bird abundance multiplied by average mass of each species as reported in Dunning (1993). Rock Doves (*Columba livia*) were omitted from the California biomass data because they overwhelmed all other species (Blair 1996). The assemblages of species found at the biological preserves in California and Ohio were taken as the minimally-disturbed standard (Karr and Chu 1998), or the best representatives of the bird communities that existed in the areas prior to development. Percent of woodland species in more urban sites is based on this minimally-disturbed standard.

2.6 Validation of Sites

I conducted validation surveys to assess whether the assemblages of birds at sites included in this study were typical of sites with similar land-use. In each ecoregion, I surveyed 24 additional sites that were within 20 km of the study sites. The set of validation sites included four different business districts, office parks (CA) and apartment complexes (OH), residential areas, golf courses, open-space reserves which were heavily used for recreation, and open-space reserves which were infrequently used by hikers (because no additional biological preserves were available). In California, I visited the original study sites and the 24 validation sites for one day each in July 1993 using the same bird sampling protocols as used in the main portion of this study. In Ohio, an assistant or I visited these sites in June and July of 1997. To test whether these sites were typical of other sites of land-use within each ecoregion, I performed a cluster analysis using the Goodman-Kruskal gamma correlation coefficient on the count of individuals within each species (not densities) of all birds observed. This distance metric is recommended for rank order or ordinal scales (SYSTAT 1992).

2.7 Comparisons Between Ecoregions

I compared species richness, diversity, abundance, biomass, and percent of woodland species between the two ecoregions with Pearson correlations (SYSTAT 1992). I used two methods to determine if increasing urbanization led to a more homogeneous avifauna. First, I conducted a land-use type by land-use type comparison (e.g., golf courses in both ecoregions) using Jaccard's Index of Similarity, which measures percent species overlap between two sites (Magurran 1988). Jaccard's Index may range from 0 (indicating no species overlap) to I (indicating complete species overlap). Second, I performed a multivariate cluster analysis using normalized percent disagreement on the average daily density of all species in all of the 12 formal study sites, a metric that is useful when there is high β-diversity and, consequently, many empty cells (Jongman et al. 1995, SYSTAT 1992).

3 Results

3.1 Ordering the Urban Gradient

The Delphi Technique led to similar orderings of the sites along the gradient in both ecoregions. In California, the ranking of the sites in order of increasing urbanization was biological preserve, open-space recreational area, golf course, residential area, office park, and business district. In Ohio, the ranking of the sites was nearly identical except that exactly half of the respondents ranked the apartment complexes and half ranked the business district as most urban. However, land-cover data (see below) suggested that the business district is the more urban (Blair 1996, Gering 1999).

The assessments of land-cover revealed unimodal distributions of percentage area covered by buildings, pavement, lawn, grassland, open water, and trees and shrubs along the urbanization gradient (Figs. 3 and 4). Cluster analysis using these measures generally grouped each land-use type across ecoregions. For example, the residential areas in California and Ohio were more similar to one another according to cover characteristics than to other sites (Fig. 5). The major exception was the open-space reserve in California. This location had large areas of grassland, a cover type

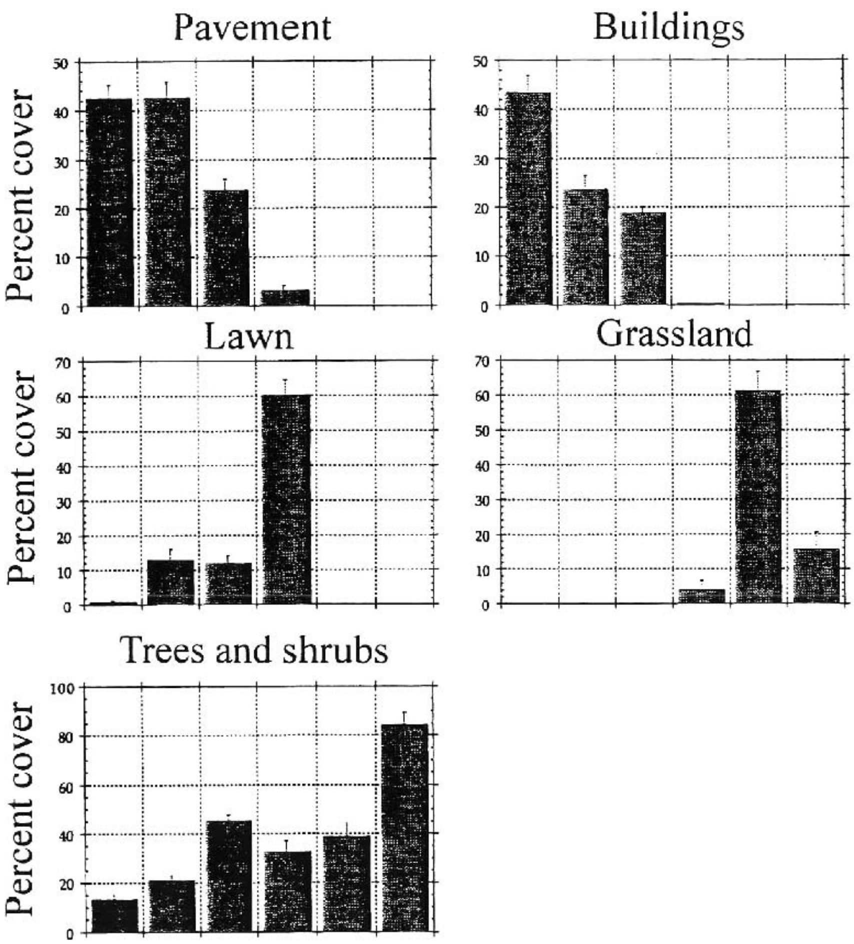

Fig. 3 Percent area (± SE) covered by pavement, buildings, lawn, grassland, and trees and shrubs in a California preserve, open-space recreational area, golf course, residential area, office park, and business district (in order from left to right)

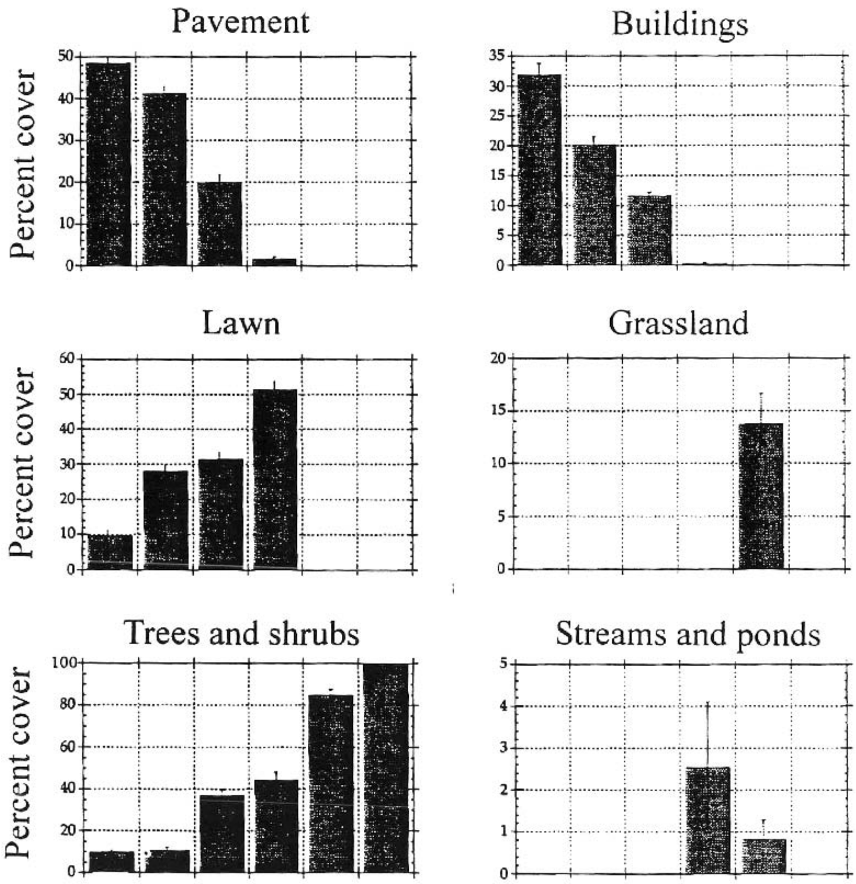

Fig. 4 Percent area (± SE) covered by pavement, building, lawn, grassland, and trees and shrubs, and open water in an Ohio preserve, open-space recreational area, golf course, residential area, apartment complexes, and business district (in order from left to right)

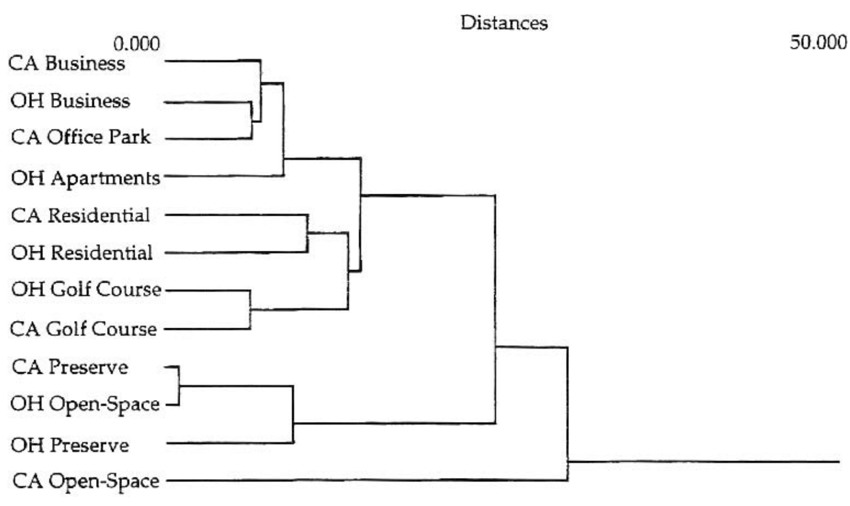

Fig. 5 Cluster analysis of sites in California and Ohio based on percent area covered by pavement, buildings, lawn, grassland, trees and shrubs and the metric of Euclidean distance

largely unique to that site. The distinct placement of the open-space reserve in California forced the open-space reserve in Ohio to group with the preserves.

3.2 Birds Along the Gradient

The abundances of all bird species varied to some degree across the gradients (Figs. 6 and 7). Of the 40 bird species in the California survey, 31 had continuous, unimodal distributions (within the limits of 1 SE) of abundance across the urban gradient. In other words, the estimated densities of these species were highest at one end of the gradient and decreased gradually as urbanization increased or decreased. Of the 44 species encountered in Ohio, 38 had unimodal distributions within the limits of 1 SE.

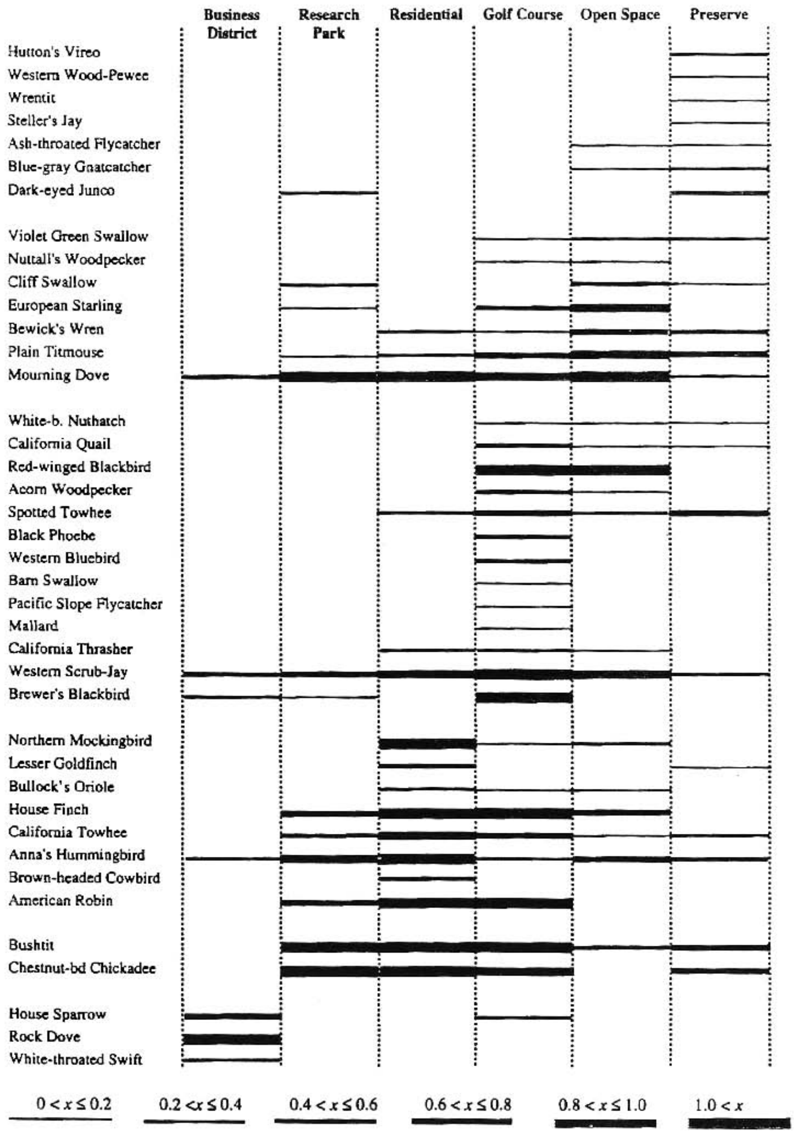

Fig. 6 Distribution and abundance of summer resident birds in California. Line width represents birds ha^{-1}

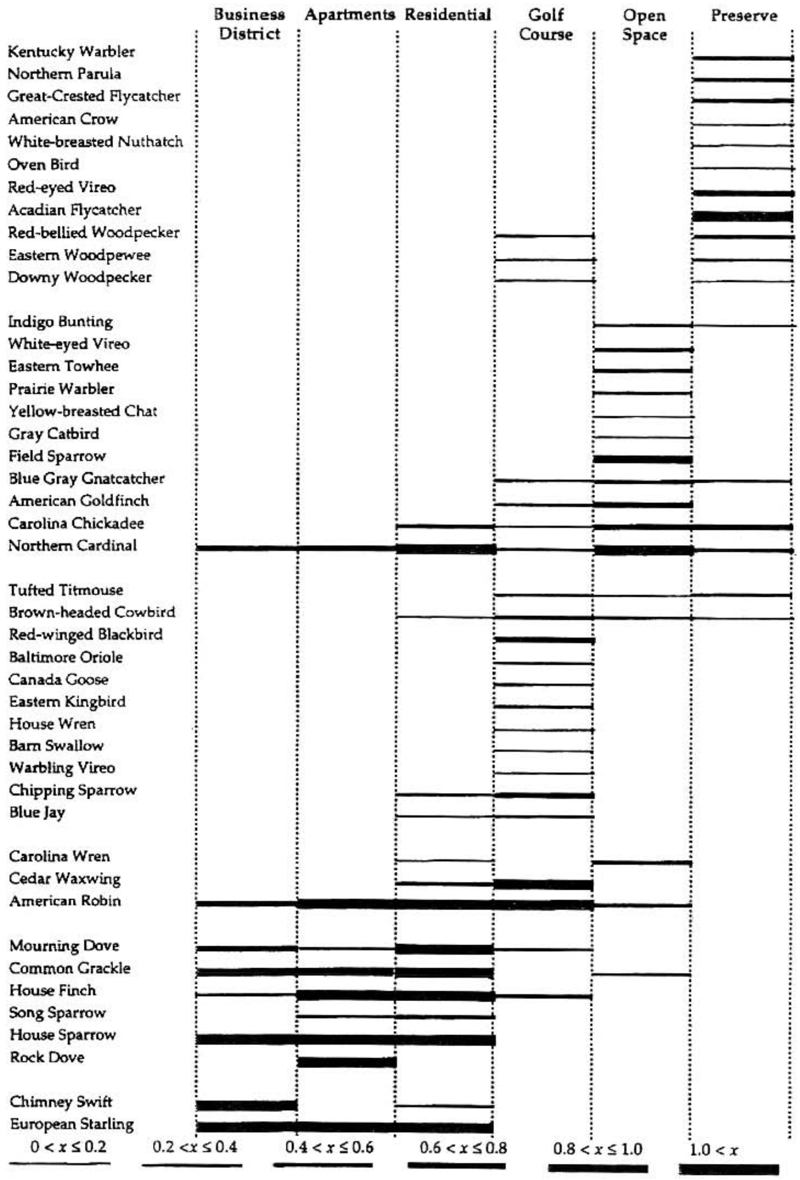

Fig. 7 Distribution and abundance of summer resident birds in Ohio. Line width represents birds ha^{-1}

3.3 Comparison of Community Measures

Species richness ($n = 6$, $r = 0.959$, $P = 0.002$), Shannon diversity ($n = 6$, $r = 0.826$, $P = 0.043$), and the percent of woodland species remaining ($n = 6$, $r = 0.913$, $P = 0.011$) were significantly correlated between similar land-use types in the two ecoregions (Fig. 8). Bird abundance ($n = 6$, $r = 0.488$ $P = 0.326$) was not significantly correlated between similar land-use types in California and Ohio. However, abundances in both California and Ohio displayed similar, unimodal patterns of high levels at intermediate sites along the gradient and lower levels at the more urban or more natural sites. If sites were shifted by one degree of urbanization along the gradient (i.e., pairing Ohio business district and California office park, Ohio apartment and California residential, Ohio

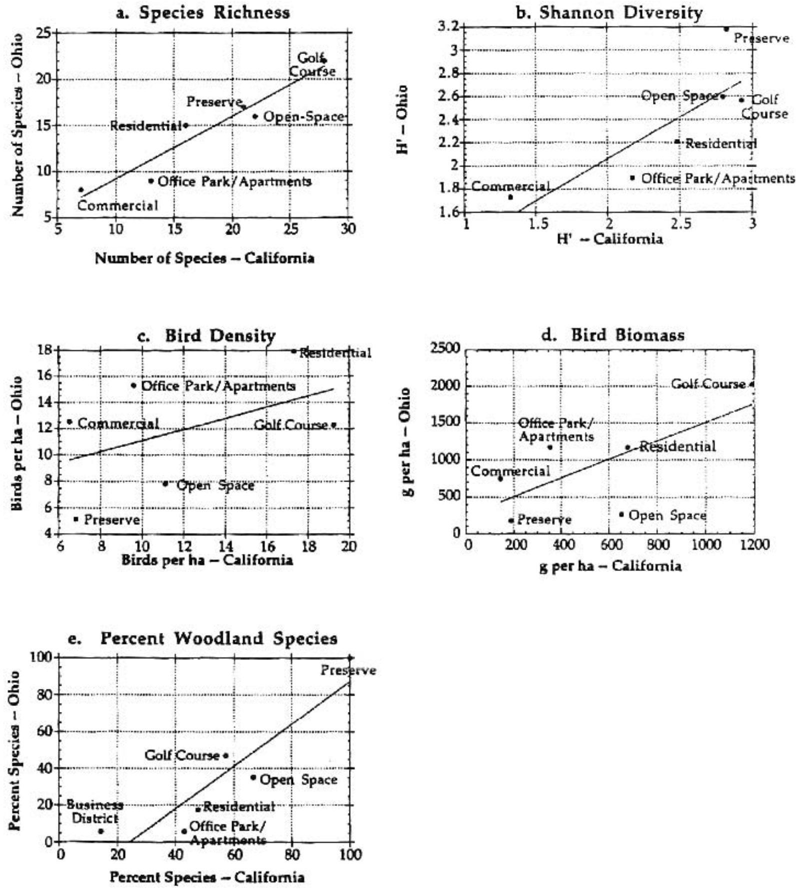

Fig. 8 Comparison between California and Ohio of community measures (a) species richness ($n = 6$, $r = 0.959$, $P = 0.002$), (b) Shannon diversity ($n = 6$, $r = 0.826$, $P = 0.043$), (c) bird abundance (number of individuals ha^{-1}) ($n = 6$, $r = 0.488$, $P = 0.326$), (d) bird biomass (g ha^{-1}) ($n = 6$, $r = 0.717$, $P = 0.109$), and (e) percent of woodland species ($n = 6$, $r = 0.913$, $P = 0.011$). Lines were generated by least squares regressions

residential and California golf course, Ohio golf course and Calfornia open space, Ohio open space and California preserve) then the correlation was significant ($n = 5$, $r = 0.955$, $P = 0.012$). Bird biomass ($n = 6$, $r = 0.717$, $P = 0.109$) was not significantly correlated between similar land-use types in California and Ohio though both states displayed similar, unimodal patterns of high levels at intermediate sites along the gradient and lower levels at the more urban or more natural sites.

3.4 Homogenization of Avifauna

Species similarities between the formal study sites in California and Ohio exhibited a distinct break midway along the urbanization gradient, more specifically between golf courses and open-space (Fig. 9). These results suggest that the land-use types fall into two groups: relatively urbanized (business district, office park/apartment complexes, residential, and golf courses) with an average of 0.185 and relatively undeveloped (the open-space reserve and biological preserve) with an average of 0.065. Instead of increasing monotonically with urbanization, it appears that species similarity jumps once the landscape is entirely manipulated.

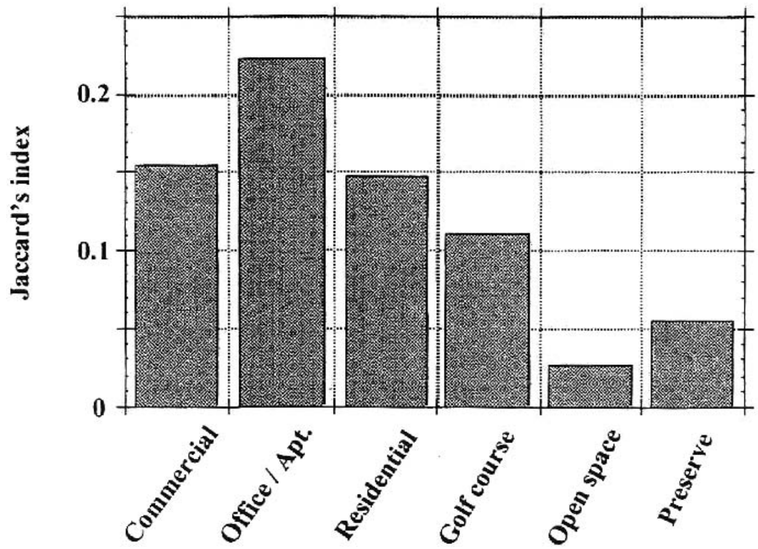

Fig. 9 Jaccard's index of similarity suggests that species overlap for the sites in California and Ohio is, on average, three times higher in the land-use types that are entirely manipulated by humans (business district, research park/apartment complex, residential area, and gold course) than in those sites that are less manipulated (open space/reserves and biological preserves)

The cluster analysis similarly revealed this pattern; more natural sites were relatively unique while more urban sites were more similar to one another (Fig. 10). Based on the abundance of all species, the dendrogram shows that the biological preserves, open-space reserves, and golf courses fall out in highly differentiated groups based on their ecoregion while the residential areas, office park/apartment complexes, and business districts fall into one cluster regardless of ecoregion.

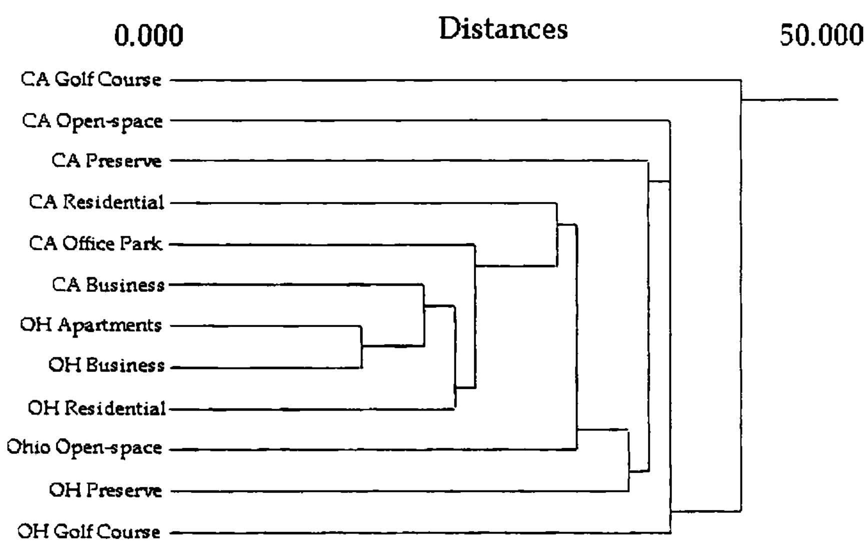

Fig. 10 Cluster analysis of sites based on abundance of all bird species using the distance metric of normalized percent disagreement. The office park in California and the apartments in Ohio are considered equivalent land-uses in this study

3.5 Validation of Sites

In each ecoregion, the species assemblage at the six formal study sites was representative of the species found at validation sites with similar levels of development. In California, all of the business districts, all of the office parks, all of the golf courses, all of the preserves, four of the open-space recreational areas, and four of the residential sites were grouped by cluster analysis into distinct subsets. One of the residential sites and one of the open-space recreational areas (not the formal study sites) grouped away from other sites within its category (Fig. 11). In Ohio, the pattern was similar but less clear because the sites display a strong trend of nestedness (i.e., species in relatively depauperate sites tended to be subsets of those in relatively rich sites; Fig. 12; Bolger et al. 1991, Atmar and Patterson 1993). The golf courses, preserves, and open-space reserves formed distinct,

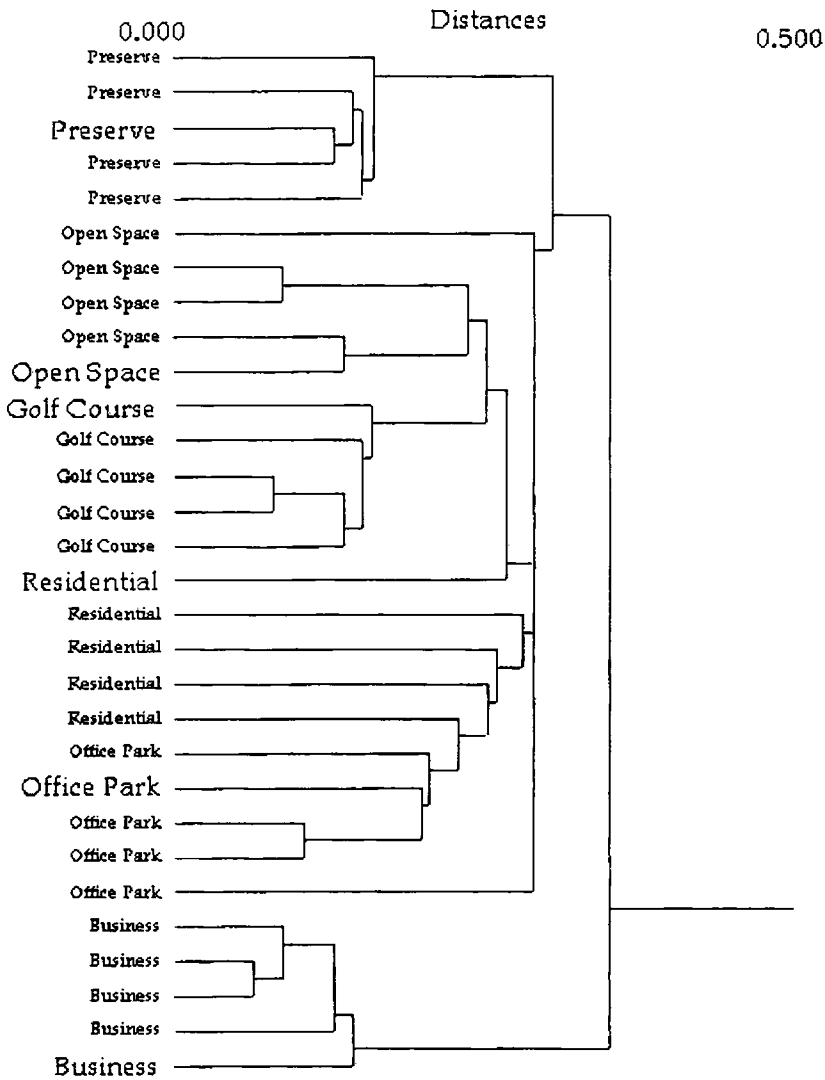

Fig. 11 Cluster analysis comparing formal study sites in California (in larger font) and 24 other sites (four of each land-use type) based upon counts from 1 day surveys of all sites using the distance metric of Goodman-Kruskal gamma correlation coefficient

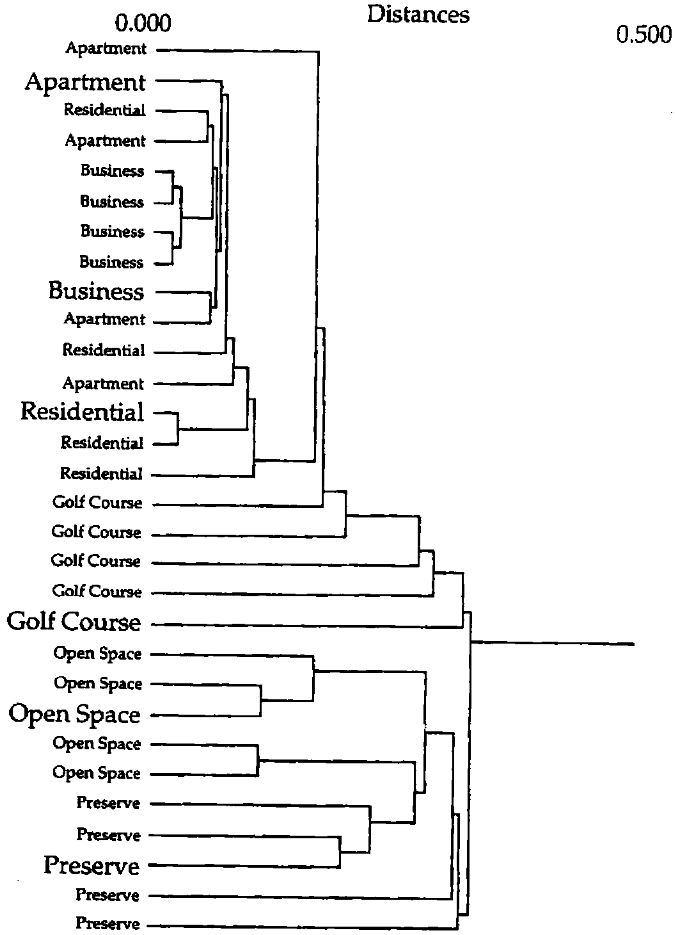

Fig. 12 Cluster analysis comparing formal study sites in Ohio (in larger font) and 24 other sites (four of each land-use type) based upon counts from 1 day surveys of all sites using the distance metric of Goodman-Kruskal gamma correlation coefficient

though nested, groups. The remaining three land-use types were nested as well. The apartment complexes, however, bridged the transition between residential areas and the business districts. In other words, some apartment complexes are very "residential" while others are more "commercial." Also, the Oxford business district was more similar to some of the apartment complexes than most business districts. These results suggest that, in general, the formal study sites in both California and Ohio are more similar in both species composition and individual species abundances to sites of similar land-use in their ecoregion. It also suggests that none of the formal study sites are wildly anomalous.

4 Discussion

Humans have the unique ability to shape the landscape according to their vision. If it is too wet, we drain it. If it is too dry, we irrigate. If it is too hot, we plant trees for shade and construct buildings to air condition. If it is too cold, we plant trees as windbreaks and construct buildings to heat. If it doesn't look like "home", we alter the flora to make it look more familiar. If it doesn't sound like "home", we introduce songbirds with which we are familiar.

This ability and these desires have resulted in a wholesale conversion of the landscape in urban and suburban areas into a comforting and surreal homogeneity throughout the United States. One could argue that you could drop a blindfolded person outside of a fast-food restaurant in Portland, Oregon; Portland, Maine; or Portland, Ohio; remove the blindfold and they would be hard pressed to identify the state in which they had been dropped. This "sameness" is created through uniformity in building types and scale, transportation patterns, and even ornamental plantings (Clay 1994). This conversion of the landscape—urbanization—also has its effect on the fauna.

Urbanization affects the distribution and abundance of birds (Marzluff 2001). My study demonstrates that the avifauna changes in a predictable pattern as land is converted from forested wildland to rural to suburban to urban, whether in California or Ohio. Species richness initially increases but then drops below the level of that of the wildland. Shannon diversity, bird abundance, and total bird biomass all follow similar patterns. The species found in undisturbed woodland are gradually extirpated until virtually none of those species exist in the most urban areas. This study also demonstrates that these patterns are identical in such disparate environments as California's coastal chaparral and Ohio's eastern broadleaf forest.

This similarity in pattern is somewhat surprising because of the environmental differences, both climatic and economic, between the ecoregions. The coastal chaparral has a Mediterranean climate with mild temperatures that rarely dip below freezing or rise above 30°C. The region has a rainy season between November and April and has virtually no precipitation between May and October. Palo Alto is in the heart of the Silicon Valley, a megalopolis that extends 100 km from San Francisco to San Jose. The area has experienced an unprecedented building boom since the early 1970s.

The eastern broadleaf forest presents an interesting contrast. It experiences a different climate with the temperature frequently remaining below freezing in winter, and often rising above 30°C in summer. It has moderate precipitation throughout the year with the highest levels in May and June. Oxford is more than 50 km from any major urban area and its population has remained relatively stable at 8,000 permanent residents and 16,000 students since the early 1970s.

Clergeau et al. (1998) also conducted a comparison of avifauna along urban gradients between two cities with differing climates. They surveyed the birds of Quebec City, Canada with its cold inland temperate climate and Rennes, France with its warm Atlantic temperate climate. They found that the summer bird fauna responded to urbanization similarly in both cities, suggesting that pronounced climatic differences do not mask the effect of urbanization on the avian communities. In fact, it appears that urbanization overwhelms the environmental differences. Thus, both Clergeau et al. and this study demonstrate that avifaunas in different ecoregions respond similarly to urbanization.

It is worth noting that Clergeau et al. found that three species (House Sparrow [*Passer domesticus*], European Starling, and Rock Dove), as a group, dominated the avifauna at most sites in both cities in both summer and winter. This pattern is reflected in this study in that this same group of species dominated the two most urban sites in both California and Ohio.

The similarity of response by the avifauna in this study to increasing urbanization (e.g., species richness, Shannon diversity, abundance, deletion of woodland species) also leads to an increasingly more homogeneous avifauna. In the simplest terms, the most natural sites (the preserves and open-space reserves) in California and Ohio had an average overlap in species of ∼7%. In contrast, the most developed sites (the business districts, office parks/apartments, residential areas, and golf courses) had an average overlap in species of ∼19% percent—nearly three times the rate of the most natural sites.

The more nuanced, multivariate cluster analysis which compares all sites to all others reveals that the most natural sites along the gradient (the preserves, open-space reserves, and golf courses) divide along ecoregions implying that the avifauna of all three land-use types are fairly unique to Ohio or California. In contrast, those sites that are more developed (the business districts, office park/apartments, and residential areas) coalesce regardless of ecoregion. In other words, the assemblage of species is relatively unique to one or the other ecoregion in natural sites but many of the same species are found in urban sites irrespective of ecoregion.

These comparisons suggest that the more humans manipulate the environment, the more similar the bird communities become. However, it also points to regional differences in the entire fauna. For example, White-throated Swifts (*Aeronautes saxatalis*) are found in downtown Palo Alto while Chimney Swifts (*Chaetura pelagica*) are found in uptown Oxford. This highlights that the two are ecologically similar but they cannot be directly compared with indices of species overlap. In the entire study, 40 species were regularly surveyed in California and 44 species were regularly found in Ohio. Only 13 species were common to both regions and this leads to a total species overlap between California and Ohio of $\sim$18%. Consequently, though urbanization does not lead to identical bird communities between the ecoregions, it does lead to more similar communities. This similarity, but not identity, is presumably due to a combination of environmental and dispersal limitations on many species.

Urbanization is not a single environmental parameter such as temperature, pH, or geographic orientation. Rather, urbanization is a layered, patchy network of environmental parameters that play out at many different scales (Alberti 2001, Hostetler 2001). At the territory level, urbanization may affect physical environmental parameters and the dispersion of resources. At the population level, urbanization may affect the viability and longevity of isolated populations (Bolger et al. 1991). At the landscape level, urbanization involves changes in patch size, configuration, connectivity, and the amount of edge (Marzluff 2001, Miller et al. 2001). Urbanization may also affect ecological processes (Vitousek et al. 1996; Bolger et al. 2001). For example, predation rates on artificial nests decline monotonically along the wildland to urban gradient in Ohio (Gering 1999). Here, I suggest that urbanization may affect two other ecological processes in addition to predation: local extinction and invasion.

In this study, woodland species, or those representing the minimally disturbed standard, disappeared along a gradient of increasing urbanization (Fig. 8e, also see Blair 1996). The most sensitive species, or those that go locally extinct with the degree of change that occurs from the preserve to the open-space areas, include Hutton's Vireo (*Vireo huttoni*), Western Wood-Pewee (*Contopus sordidulus*), Steller's Jay (*Cyanocitta stelleri*) and Wrentit (*Chamea fasciata*) in California and Kentucky Warbler (*Opornis formosus*), Northern Parula (*Parula americana*), Great-Crested Flycatcher (*Myiarchus crinitus*), American Crow (*Corvus brachyrhynchos*), White-breasted Nuthatch (*Sitta carolinensis*), Ovenbird (*Seiurus aurocapilus*), Red-eyed Vireo (*Vireo olivaceus*), and Acadian Flycatcher (*Empidonax virescens*) in Ohio. Judging from these lists and other studies (Stenberg 1988, Germain et al. 1998), small incremental shifts along the urbanization gradient may have large effects on the avifauna.

If these species were evenly dispersed across the landscapes prior to European settlement, then this is evidence that habitat conversion associated with urbanization has caused the local extinction of these species. Conversely, species that are restricted to more urbanized sites can be assumed to be local invaders that have been able to exploit environments changed by human settlement and urbanization.

The processes of extinction and invasion happen repeatedly along the urban gradient. For example, in California, the Ash-Throated Flycatcher (*Myiarchus cinarascens*) goes extinct locally with urbanization. It is found at its highest abundance in the preserve, to a lesser degree in the open-space reserve, and not at all in more urbanized sites. In contrast, the Western Scrub-Jay (*Aphelocoma californica*) increases in abundance until the middle of the gradient (the golf course) but then declines as sites become even more urbanized. The jay can be considered both an invader (capitalizing on urbanization) but also threatened by more intense urbanization. Finally, Rock Doves are found only at the most urbanized site and are an example of a particularly successful invader aided by urbanization. Notice the shift in species abundances as your eye travels down Figs. 6 and 7. This pattern of urban avoiding, suburban adaptable, and urban exploiting species is found repeatedly in both California and Ohio.

This study suggests that human manipulation of the landscape is leading to both extinction and invasion on a very local scale. Local extinction and invasion are the patterns that Hobbs (1988) highlight when they argue that the extinction debate should be broadened beyond the extinction of species. They argue that population extinctions and invasions (or, in their terms, deletions and additions) should be addressed as a threat to biodiversity because species extinction is simply the final endpoint of many population extinctions.

5 Research Needs

Future research along the lines of this study should focus on the effects of urbanization at different levels of biological organization. First, at the landscape level, it would be useful to explore how urbanization affects the heterogeneity of the landscape and how landscape heterogeneity is linked to faunal heterogeneity.

Second, at the level of this study, that of the species assemblage or community, it would be useful to develop theory and gather empirical data on the upper limits of similarity between ecoregions. Is a tripling of species overlap from 0.065 in natural areas to 0.185 in urban areas a notable increase in homogeneity?

Third, at the population level, it would be useful to explore the changes in the demographics of certain species along an urban gradient. This study assumes that a high abundance of birds is "good" and that a high density of a particular species indicates better habitat for that species. This may not be the case because reproduction and abundance may not necessarily be linked (Van Horne 1983, Vickery et al. 1992).

Finally, this study must be taken with the warning that it is correlative. It should be fairly easy to test empirically the ideas presented herein by measuring the changes in avifauna as areas continue to undergo suburban- and urbanization.

6 Policy and Management Implications

Determining what forces have led to the current patterns of urban development in the United States and the consequent homogenization of the landscape is far from simple. Current land-use patterns in the United States can be attributed to many factors. These range from historical, local attributes such as the proximity to transportation and potable water to current, global, economic conditions such as the effects of the North American Free Trade Agreement on the United States manufacturing base. The landscapes included in this study largely have been urbanized since 1945 and the automobile has been one of the strongest factors to influence development patterns in these areas. The outcome of these development decisions has been suburban sprawl. For example, the metropolitan area nearest to the Ohio study sites, Cincinnati, grew 12% between 1990 and 1996 while its population increased by only 2% (Sierraclub 1998).

This study has strong implications for management at two different scales: the individual preserve and the geographical region. At the preserve level, managers should note that species richness is not necessarily a good measure of the "wildness" of a site and that species composition is more important in making management decisions. In both ecoregions, the minimally disturbed standard sites (the preserves) had fewer species than slightly more disturbed sites such as the golf courses. This indicates that preserve managers should target the species found in the minimally disturbed standard (i.e., the woodland species) and not on overriding community measures such as species richness, Shannon diversity, or total bird abundance as their metrics for management "success." The second lesson that may be derived from this study is that the effects of urbanization are similar in broadly comparable ecoregions, (e.g., those with forest cover). This implies that the results of other studies, such as those that appear in this volume, may be applied across ecoregions and, with caution, to each manager's particular situations.

At the regional level, this study illustrates two valuable principles for land-use planners. First, any level of urbanization affects native biodiversity. The open-space reserves in California and Ohio do not support all of the species that are found in the preserves and, more intensely developed sites support even fewer species. This implies that urbanization should be concentrated when possible. Second, this study shows that increasingly more intense land-use leads to increasingly more similar avifauna (i.e., that current land-use patterns are creating a homogeneous avifauna). It also shows that this homogenization occurs at relatively less severe uses including golf courses and residential sites.

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Towards a Mechanistic Understanding of Urbanization's Impacts on Fish

Christian Wolter

Keywords: fish diversity · urban fishes · urban stream · urban watershed · dispersal

Introduction

Human population is increasing at a rate of 1.8% per year and urbanization is a global trend. In 2005, there were 3.2 billion urban residents worldwide, representing 49% of the global population (UN 2006). The urban population is projected to increase to 4 billion in 2018 and to more than 5 billion in 2030 (UN 2006). However, in the developed regions 75% of the population lived in urban settlements in 2005. This proportion is expected to increase in Europe from 72% in 2005 to 78% in 2030 and in North America from 81% at present to 87% in 2030 (UN 2006). Urban areas cover only 2.4% on the terrestrial surface of Earth, but their average population density is 52 times that of rural areas (MA 2005). Except coastal areas and island states, the highest average urban population density was found along inland waters with 817 people per km^2 (MA 2005). Indeed, most people believe that urban environmental conditions are deteriorating, and the condition of urban waters is high on their list of worries. Features of urbanization have been reviewed by Paul & Meyer (2001): impervious surface covers, alteration of drainage density and flow dynamics, decreasing groundwater renewing and sediment supply, and increases in surface runoff, water temperature, pollutants, and nutrients. The cumulative effect of various human activities in urban areas profoundly influence urban waters and their biota, either directly by channel modification and habitat degradation or indirectly by land use change and runoff (Booth et al. 2004). The percentage of impervious surface cover has been commonly suggested as the best single predictor of the response of stream biota to urbanization (e.g. Karr & Chu 2000, Allan 2004, Booth et al. 2004, Miltner et al. 2004). However, the threshold values for demonstrated significant biological degradations at the catchment level of 10–15% total impervious area are much below the commonly observed $> 50\%$ impervious cover in metropolitan areas at the regional level (Booth et al. 2004). Karr & Chu (2000) considered biological communities as irreparably damaged if the impervious cover within a watershed ranged between 25–60%. Urbanization is highly positively correlated with both the endangerment of native and the invasion of non-native fish within watersheds and thus, considered as major cause of biotic homogenization (Marchetti et al. 2006). Urbanization tends to favour the persistence of relatively few intolerant, generalist native species, the introduction and establishment of widespread non-natives, and the extinction and extirpation of specialized, intolerant native species (Marchetti et al. 2006).

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But how do urbanized areas and impervious covers impact fish and promote non-native species? What are the basic mechanisms? Non-linear relations between impervious cover and biological communities have been observed, in particular if instream habitats or hydrodynamics were also considered. For example, in Ohio watersheds at sites with relatively undeveloped riparian buffers, the biological integrity was maintained despite high levels of urban land use (Miltner et al. 2004). Accordingly, habitat destruction caused by urban development seemed the primary force driving common species to decline due to resource limitations and enabling different or new species to increase when benefiting from the changed habitats. To analyse this question, between 1992 and 2002 the fish assemblages of 27 Federal waterways have been extensively studied: more than 470 sites were surveyed, 2,100 samples collected, and 336,500 fish recorded representing 35 species. This data set revealed substantial findings on environmental factors structuring local fish assemblages (Wolter & Vilcinskas 1997a, 1998a, Wolter 2000, 2003), environmental pressures (Wolter 2001a, Arlinghaus et al. 2002), urban gradients (Wolter 1997b, 2000, Wolter 1999a), the impact of urbanization on fish abundance (Wolter & Vilcinskas 1996, Wolter 2001b) and population dynamics (Wolter 1998, 1999b, Wolter & Vilcinskas 1998b), and the ecological performance of species (Wolter & Vilcinskas 1997b, Arlinghaus & Wolter 2003).

This paper refers briefly to three main results of the mentioned studies to illustrate the specifics of fish communities in urban waters for the example of the water system of Berlin, Germany, where 3.5 million people reside. Detailed information on methods used, sampling design, environmental variables, species lists etc. are presented in the original papers.

Restricted Structural Diversity Limits Fish Diversity

Fish of urban rivers are generally exposed to higher thresholds of multiple disturbances and in particular to the cumulative effects of altered hydrology and geomorphology. Waters in urban areas are especially embanked, regulated, and their channels fixed, due to an extraordinary high amount of land use in the surroundings, resulting in substantial simplifications of the riverine habitats with monotonous bank structures and few tributary refuges. In the urban waterways studied on average 88% (± 21% standard deviation) of the total bank lines were covered by artificial embankments like rip-rap or sheet pile wall, in contrast to 72% (± 30%) in the rural waterways. In addition, the average number as well as the structural diversity of tributaries was lower in the urban waterways.

The observed fish assemblage patterns differed significantly between the 13 urban and 14 rural waterways surveyed (Fig. 1, Table 2; data from Wolter & Vilcinskas 2000). In urban waters the number of species recorded was generally lower and six native and three non-native fish species were not detected. Typical floodplain species (bitterling, crucian carp, weatherfish, ten-spined stickleback) as well as typical riverine species (chub, zope, stone loach) with more specified habitat requirements disappeared from the urban fish species pool. Dropping abundances and disappearance of intolerant species start immediately with the urbanization of watersheds and result in declining fish diversity (e.g. Boët et al. 1999, Wolter et al. 2000, Paul & Meyer 2001, Wolter et al. 2003). The urban fish communities were dominated by two species, roach and perch accounting for 70% of all fish on average, whilst most of the species were rare, contributing less than 1% each. The observed species inventories ranged from 6–21 in urban and 11–28 in rural waterways. Both the mean number of species and the species diversity (estimated as Shannon's diversity index H') were significantly higher (*Students t* statistics, $p < 0.05$) in rural waters, despite a high amount of habitat destruction in rural waterways. However, urbanization did not impact intolerant species with sensitive habitat requirements only, like chub or pike, but also the abundance of roach, one of the most common eurytopic species in Europe. Roach itself is considered as indicator for environmental degradation by eutrophication (e.g. Oberdorff & Hughes 1992, Oberdorff et al. 1993, Carrel &

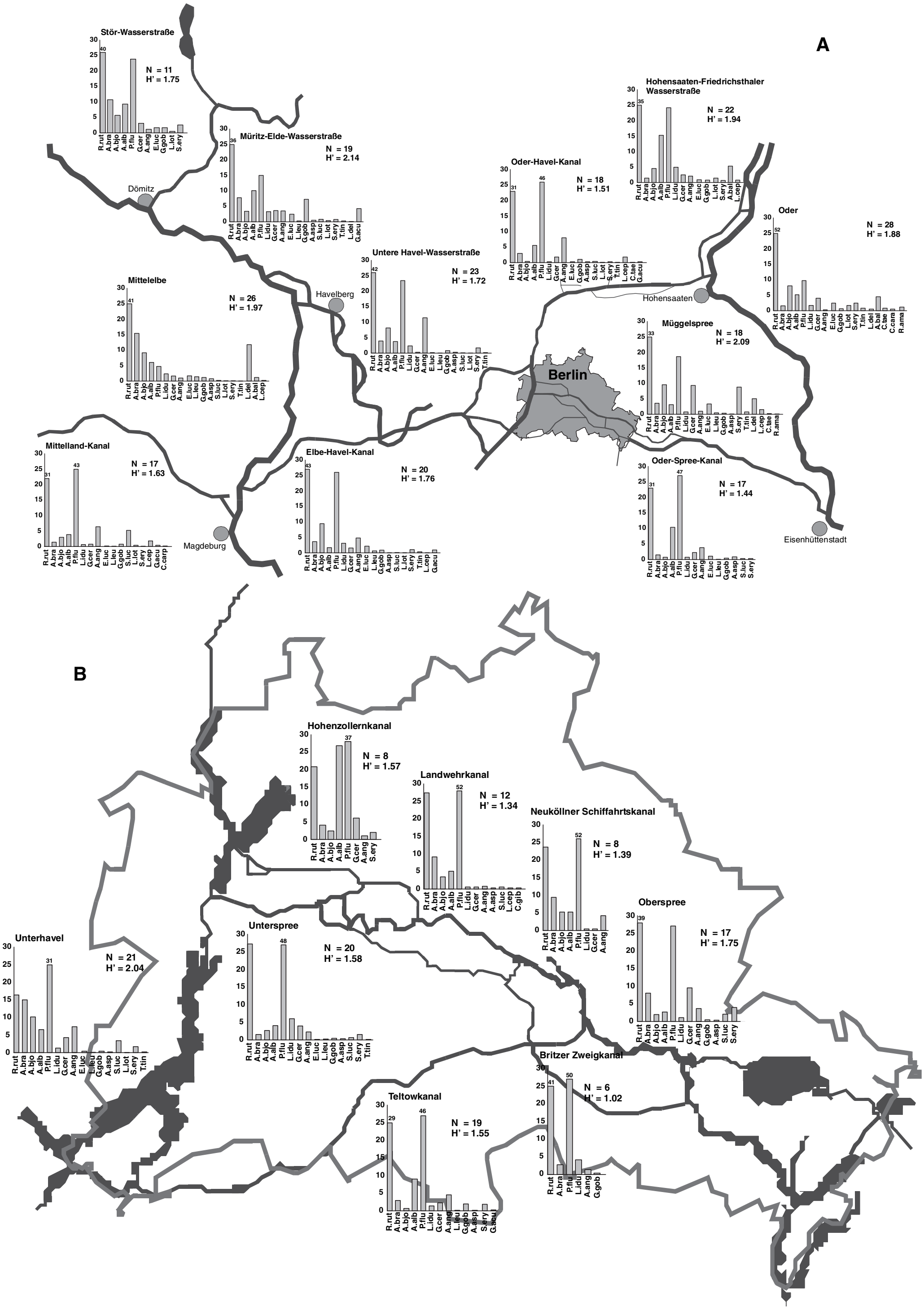

Fig. 1 Relative abundance (%) of the most common fish, species number (N), and Shannon's species diversity (H') in rural (A) and urban (B) waterways surveyed in Berlin and in the NE lowlands in Germany (species abbreviations in Table 2)

Table 1 Main hydro-morphological characteristics of the urban (U) and rural (R) waterways studied (min. = minimum, subm. = submerged, tribs. = tributaries)

Waterway	Length (km)	Min. width (m)	Mean Depth (m)	Mean flow velocity (ms^{-1})	Subm. Macrophytes	Artificial embankment (%)	No. of Tribs. per km	Environment
Britzer Zweigkanal (BZK)	3.4	27.5	2.7	< 0.05	none	100.0	0.00	U
Charlottenburger Verbindungskanal (CVK)	1.7	14.5	2.7	0	none	100.0	0.00	U
Dahme Wasserstraÿe (DaW)	25.0	21.0	1.5	< 0.05	none	74.2	0.36	U
Gosener Kanal (GoK)	2.8	35.0	3.0	< 0.1	none	100.0	0.00	U
Hohenzollernkanal (HZK)	8.2	53.7	3.3	< 0.05	none	100.0	0.36	U
Landwehrkanal (LWK)	10.7	23.0	1.8	0	none	100.0	0.19	U
Neuköllner Schifffahrtskanal (NSK)	4.1	25.0	2.0	0	none	100.0	0.00	U
Oberhavel (OHa)	28.5	54.0	3.5	< 0.1	none	68.1	0.77	U
Oberspree (OSp)	15.0	30.0	2.0	< 0.05	none	96.3	0.38	U
Teltowkanal (TeK)	37.8	27.5	2.7	< 0.05	none	87.6	0.14	U
Unterhavel (UHa)	16.4	46.0	1.5	< 0.1	rare	24.4	0.30	U
Unterspree (USp)	17.8	30.0	2.0	0.1	none	95.2	0.82	U
Westhafenkanal (WHK)	3.1	46.6	3.8	0	none	100.0	0.32	U
Elbe-Havel-Kanal (EHK)	56.8	40.0	3.0	< 0.1	none	93.6	0.32	R
Havelkanal (HvK)	34.9	34.0	3.0	< 0.05	none	99.2	0.20	R
Havel-Oder-Wasserstraÿe (HOW)	135.0	23.0	2.6	< 0.05	rare	89.7	0.28	R
Hohensaaten-Friedrichstaler-Wasserstraÿe (HFW)	42.5	20.0	2.4	< 0.05	none	87.6	0.14	R
Mittelelbe (Elbe)	486.0	200.0	2.9	1.0	rare	10.2	1.07	R
Mittellandkanal (MLK)	321.3	37.0	3.5	< 0.1	none	99.2	0.10	R
Müggelspree (MSp)	35.4	25.0	2.0	0.5	frequent	8.6	7.63	R
Müritz-Elde-Wasserstraÿe (MEW)	120.8	20.0	1.5	0.1	frequent	80.3	0.51	R
Oder (Od)	161.0	200.0	2.2	1.0	none	44.7	0.26	R
Oder-Havel-Kanal (OHK)	49.3	34.0	3.0	< 0.05	frequent	98.8	0.14	R
Oder-Spree-Kanal (OSK)	85.1	28.0	2.0	0.05	none	98.4	0.34	R
Stör-Wasserstraÿe (StW)	19.7	17.0	2.0	0	frequent	87.0	0.18	R
Untere Havel-Wasserstraÿe (UHW)	148.5	46.0	1.8	< 0.1	frequent	50.5	0.69	R
Westoder (WOd)	66.5	43.5	1.8	0.5	none	67.1	0.24	R

Table 2 Means (± SD) of relative abundance (%) and main fish assemblage characters in urban and rural inland waterways (N = number of waterways, [a] single records, significance level: *p < 0.05, **p < 0.01)

Scientific name	Common name	Urban (N = 13)	Rural (N = 14)
Anguilla anguilla	eel	4.3 ± 3.0	3.8 ± 3.2
Abramis ballerus	zope		3.7 ± 1.8
Abramis bjoerkna	silver bream	4.1 ± 3.5	6.0 ± 3.6
Abramis brama *	common bream	10.3 ± 8.4	4.5 ± 4.2
Alburnus alburnus	bleak	6.9 ± 6.5	8.0 ± 5.2
Aspius aspius	asp	0.3 ± 0.1	0.4 ± 0.4
Carassius carassius	crucian carp		0.2 ± 0.2
Carassius gibelio	prussian carp	0.2 ± 0.1	0.04 ± 0.03
Ctenopharyngodon idella	grass carp		0.005[a]
Cyprinus carpio	common carp	0.03 ± 0.01	0.1 ± 0.1
Gobio gobio	gudgeon	0.5 ± 0.5	1.5 ± 1.7
Hypophthalmichthys nobilis	bighead		0.01[a]
Leucaspius delineatus	sunbleak	0.1 ± 0.05	2.9 ± 4.4
Leuciscus cephalus *	chub	0.2[a]	1.0 ± 0.9
Leuciscus idus	ide	1.6 ± 1.7	2.0 ± 1.5
Leuciscus leuciscus	dace	0.1 ± 0.1	0.3 ± 0.4
Rhodeus amarus	bitterling		0.5 ± 0.5
Rutilus rutilus **	roach	29.5 ± 8.4	39.2 ± 7.0
Scardinius erythrophthalmus	rudd	1.6 ± 1.1	1.7 ± 2.2
Tinca tinca	tench	0.2 ± 0.2	0.3 ± 0.3
Barbatula barbatula	stone loach		0.04[a]
Cobitis taenia	spined loach	0.02 ± 0.01	0.2 ± 0.3
Misgurnus fossilis	weatherfish		0.1 ± 0.04
Silurus glanis	wels	0.03 ± 0.01	0.02 ± 0.01
Esox lucius **	pike	0.3 ± 0.2	1.4 ± 1.1
Oncorhynchus mykiss	rainbow trout		0.04[a]
Lota lota	burbot	0.04 ± 0.04	0.7 ± 0.7
Gasterosteus aculeatus	3-sp. stickleback	0.2 ± 0.1	1.0 ± 1.4
Pungitius pungitius	10-sp. stickleback		0.06[a]
Gymnocephalus cernuus	ruffe	4.2 ± 3.8	2.7 ± 2.2
Perca fluviatilis *	perch	37.7 ± 14.1	24.2 ± 13.2
Sander lucioperca	zander	1.1 ± 1.0	0.8 ± 1.5
Mean number of species *		13.7 ± 4.6	18.8 ± 4.8
Species diversity H′*		1.56 ± 0.24	1.77 ± 0.25
Evenness		0.61 ± 0.07	0.61 ± 0.08
Community Dominance Index		70.0 ± 11.3	64.5 ± 8.6
Total number of individuals		34,828	97,563
Total number of species		23	32

Rivier 1996), but was much (*Students t* statistics, $p < 0.01$) less abundant in the urban than rural waters (Table 2).

These comparisons might be biased by the naturally higher fish species inventories of the large regulated rivers. Thus, we compared the canal fish assemblages separately. In both, urban and rural areas these canals have been artificially constructed according to the same standardized guidelines regarding the navigation-induced physical forces, embankment stability, vessel's drought, canal width, depth, and profile. However, the different land use patterns resulted in slightly lower percentages of artificial embankments, especially of sheet pile walls, and in a higher amount of instream habitats relevant to fish like macrophyte cover, as well as in higher numbers of tributaries in the rural canals (Table 1). These structural differences seemed relevant to fish, because the urban canals showed much (*Students t*, $p < 0.01$) lower mean species number and species diversity than rural canals (Table 3). These findings corresponded very well with those presented in Table 2, and underlined the impact of the heavily reduced structural diversity resulting from urbanization on fishes.

Table 3 Means (± SD) of main fish assemblage characters of artificial navigation canals in urban and rural environments (N = number of canals, significance level: *p < 0.05, **p < 0.01)

Metric	Urban canals (N = 8)	Rural canals (N = 9)
Mean number of species **	11.1 ± 3.8	17.1 ± 3.6
Species diversity H′*	1.43 ± 0.21	1.72 ± 0.27
Evenness	0.61 ± 0.08	0.61 ± 0.08
Community Dominance Index	74.4 ± 10.4	66.8 ± 9.1
Total number of individuals	9,559	29,982
Total number of species	21	28

The particular influence of available tributaries on the fish communities was studied in more detail by comparing 56 sampling sites at the mouth of tributaries within 88 linear stretches (Wolter 2001b). Mean fish species number (10.8 ± 2.09, standard deviation), species diversity ($H' = 1.78 \pm 0.32$), and catch per unit effort CPUE (fish/100 m = 73.3 ± 29.2) were significantly higher (*Students*

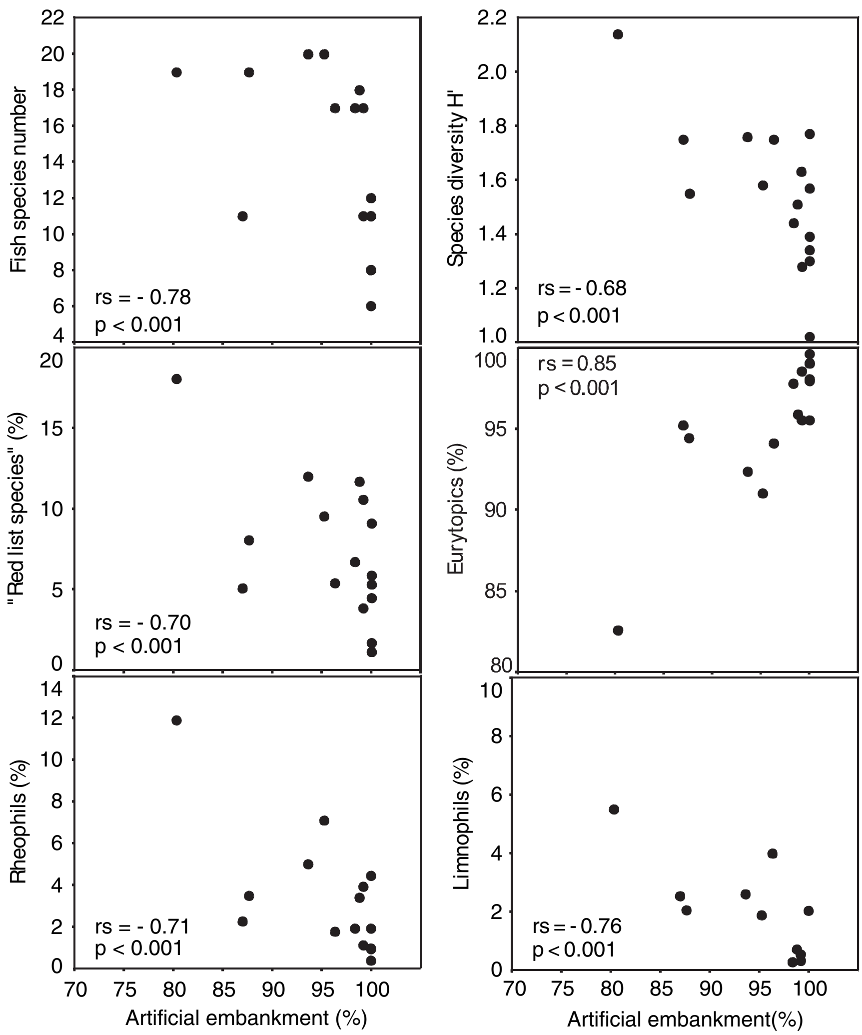

Fig. 2 Spearman rank correlations (coefficient rs) between the percentage of artificial embankment (riprap or sheet pile wall) and various fish community measures (number of waterways N = 19). Redrawn from Wolter (2001b)

t, $p < 0.01$) at tributary sites than at linear canal reaches (fish species number $= 5.7 \pm 2.12$, $H' = 1.19 \pm 0.27$, and CPUE $= 27.3 \pm 8.1$). These findings correspond very well with the observed inverse correlation between distance from tributaries and abundance of eurytopic fish in the free flowing section of the Danube River (Hirzinger et al. 2004), and the importance of off-channel habitats as refuges for fish (Copp 1997). Similarly, ecotone diversity, i.e. diverse instream habitat structures along the banks, has been identified as essential for high fish diversity (Copp 1997, Wolter 2001b, Arlinghaus et al. 2002, Hirzinger et al. 2004). In the Seine River basin a general decrease of specialized fish species has been observed resulting from homogenization of littoral habitats (Boët et al. 1999). Even in lakes the residential development and alterations of littoral habitats caused decreased growth rates and productivity of fish stocks (Schindler et al. 2000) and altered the spatial distribution and aggregation of fish (Scheuerell & Schindler 2004).

Finally, the percentage of artificial embankment requires special consideration as one of the main features of waterways. Except for the large regulated rivers, the proportion of artificial embankments was rather high and typically increased 70–80%, with low differences between rural and urban waterways. However, 80% artificial embankments seemed to be a threshold value, and the gradual increase up to complete embankment significantly impacted fish assemblages. Even the final 10% of the total bank line, if remaining natural or if covered by artificial embankments were reflected in highly significant fish-faunistic differences (Fig. 2, from Wolter 2001b). In waterways with "only" 90% of the shore lines embanked, the observed fish species numbers, species diversity and proportions of rheophilic, limnophilic, as well as threatened fish were significantly higher compared to the completely embanked waterways. The dominance of the most tolerant, eurytopic species significantly increased with shoreline degradation, especially the dominance of perch.

Shore Line Degradation Causes Community Dominance of Perch

Perch (*Perca fluviatilis*) was one of the most widespread fish species in the waters surveyed (Wolter & Vilcinskas 1997a, Wolter et al. 2003). In contrast to the majority of species, perch abundance increased substantially in urban waters (Table 2). Within both rural and urban waters there was a shift of perch abundance along a gradient of artificial embankment, involving a change from the numerical dominance of roach to the dominance of perch in the fish assemblages (Wolter & Vilcinskas 1997b, 1998b). In waterways with predominately natural shorelines, perch were significantly (*Students t*, $p < 0.05$) less abundant (mean 15.43% ± 8.22% standard deviation) than were roach (38.87% ± 4.14%) and other fish species (45.69% ± 8.62%). Contrary to this, the mean perch abundance (42.24% ± 8.83%) exceeded that of roach (32.75% ± 7.82%, $p > 0.05$) and other species (25.01% ± 7.64%, $p < 0.05$) in waterways with predominately artificial shorelines (Fig. 3).

Similar observations were made with respect to total fish biomass: in the more natural waterways, roach was dominating (22.36% ± 5.39%), while the relative biomass of perch was 6.79% ± 3.04%. In contrast, in the more artificial waterways perch became the dominant fish (26.54% ± 2.81%), whilst the mean roach biomass dropped to 19% ± 9.09%. Which factors favoured perch? All waterways investigated were polytrophic to hypertrophic, i.e. the nutrient conditions were favourable to cyprinids, especially roach (Persson et al. 1991, Oberdorff & Hughes 1992, Carrel & Rivier 1996). The competitive superiority of cyprinids involves roach outcompeting juvenile perch (Persson & Greenberg 1990). Limited food resources would favour roach too (Persson & Greenberg 1990, Bergmann & Greenberg 1994), and the only structure competitively favouring perch, submerged vegetation, is lacking in urban waterways: (Persson & Greenberg 1990). Thus, in contrast to the perch dominance observed, a numerical dominance of roach had to be expected.

Roach and perch are the most environmentally tolerant, common and widespread fish species in the waterways studied. Both species require neither specific spawning habitats nor substrata

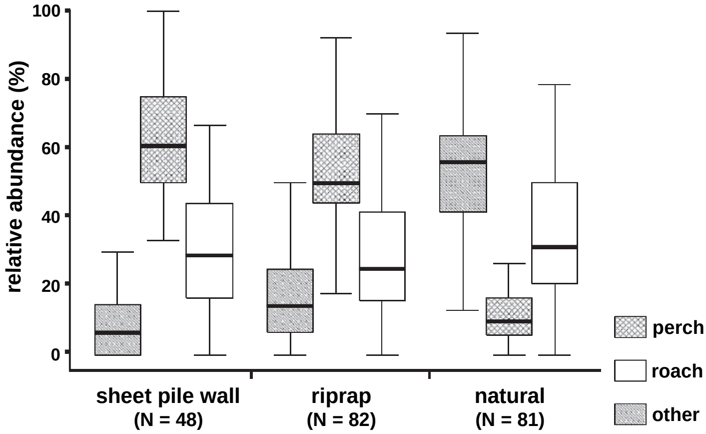

Fig. 3 Occurrence of perch, roach and other species in relation to the type of embankment. The boxes represent 50% of all observations and the whiskers 90%. The thick line marks the median and N the number of sites. Figure redrawn from Wolter & Vilcinskas (1997b)

or hydraulic conditions, and their spawning has been commonly observed. The main difference between both species, or more general between percids and other species, is an ontogenetic habitat shift: larval perch shift their habitat to the pelagic zone immediately after hatching and shift back to the littoral zone at a size range of 11–30 mm depending on the predation pressure in the pelagic (Byström et al. 2003). In contrast, larvae of other species remain in the littoral and essentially depend on shallow, slow flowing nursing habitats. Accordingly, habitat degradations in the littoral zones impacts the recruitment of all fish species with shoreline-bounded larvae. Species with pelagic larvae remain substantially less impacted and thus, the latter become numerical dominant. This pattern might favour perch over other species such as roach under anthropogenic degradations in waterways and urban waters, which has led to the suggestion of perch as an indicator species for structural degradation in regulated rivers and canals (Wolter & Vilcinskas 1997b). This hypothesis has been confirmed empirically by a study of juvenile fish recruitment in a canal, where the availability of littoral habitats was restricted due to commercial navigation (Arlinghaus et al. 2002). The recruitment of juvenile fish in the littoral zone was restricted to bays, oxbows and tributaries due to the high navigation-induced currents in the main channel preventing small juveniles from maintaining shallow low flowing nursing habitats along the banks (Wolter & Arlinghaus 2003, Wolter et al. 2004).

Urbanized Waters Hinder Migrations and Limit Gene Flow

The third example comprises species exchange and gene flow within urban water systems and canals. In selected waterways numerous sites have been surveyed to investigate fish migrations within canals, the suitability of canals as migration routes for fish as well as the barrier effect of navigable locks (Wolter & Vilcinskas 1998b); and population genetic studies were performed to characterize habitat fragmentation and genetic isolation by migration barriers and urbanized river segments (Wolter 1998, 1999b).

Presence-absence and relative abundance data of fish species suggested a barrier effect of extended, nearly still, monotonous watercourses (Wolter & Vilcinskas 1998b). They were less attractive for fish movements and consequently inhibited directional fish migrations. Correspondingly, observed fish invasions proceeded much faster in natural river systems compared to canals. For example, the tubenose goby (*Proterorhinus marmoratus*), a fish species native to the Danube River and recently invading the Rhine River, needed eight years to pass the 171 km long Main-Danube canal (Schadt 2000), but only three years to reach the 895 km distant Rhine Delta in the Netherlands (Tien et al. 2003). However, the barrier effect of such monotonous canals with negligible low flow velocity seemed to be species-specific. While rheophilic specimens terminated their migrations on average at maximum distances of 6–8 km (range 0.5–15 km), no obstructing effects could be observed for eurytopic species, like bleak, common bream, silver bream, roach, and perch. However, possible barrier effects might be hidden by the widespread distribution of eurytopic fish.

Therefore, population genetic analyses have been performed to detect limited gene flow as an indication of restricted fish migrations. Four widespread, eurytopic cyprinids were selected for a population genetic study: common bream, silver bream, roach, and rudd. At the same nine sites, a minimum of 30 specimens each was collected of all four species (details in Wolter 1998, 1999b). Two sites were situated in the urban part of Berlin (Fig. 4), and two reference sites enclose a 155 km long free flowing stretch of the Oder River without any barriers. Thirteen enzyme systems coded by 25 loci were analysed and revealed a high genetic variability of the species examined (details in Wolter 1998, 1999b). The following mean values ($\pm$ standard error) of intraspecific genetic variability were calculated: common bream (9 subpopulations) percentage of polymorphic $P_{95} = 21.8 \pm 2.1\%$, average observed heterozygosity $H_{obs} = 0.098 \pm 0.010$, average expected heterozygosity $H_{exp} = 0.083 \pm 0.008$; silver bream (9) $P_{95} = 17.8 \pm 1.2\%$, $H_{obs} = 0.082 \pm 0.006$, $H_{exp} = 0.074 \pm 0.006$; roach (9) $P_{95} = 22.2 \pm 2.6\%$, $H_{obs} = 0.091 \pm 0.010$, $H_{exp} = 0.086 \pm 0.008$; and rudd (7) $P_{95} = 17.1 \pm 2.1\%$, $H_{obs} = 0.086 \pm 0.009$, $H_{exp} = 0.079 \pm 0.010$.

The unbiased genetic distances according to Nei (1978) between sites ranged from 0.000–0.037 in common bream, 0.001–0.054 in silver bream, 0.000–0.040 in roach and 0.000–0.044 in rudd. Within rivers between 8.1% (common bream) and 17.4% (rudd) of the genetic variability was attributable to differences between subpopulations, while more than 80% of the observed total genetic variability was due to individual variability within subpopulations. Summarising the results of non-hierarchical F-statistics, the theoretical gene flow between neighbouring samples was restricted in a species-specific manner. The samples from the reference river behaved as one panmictic unit inhabiting a 155 km long Oder river stretch. In contrast, the population genetic structure of the subpopulations from the two urban sites indicated a considerable habitat fragmentation.

In general, two populations are considered as isolated or panmictic respectively, if the theoretical gene flow between them is less than one or more than four effective migrants, i.e. immigrating

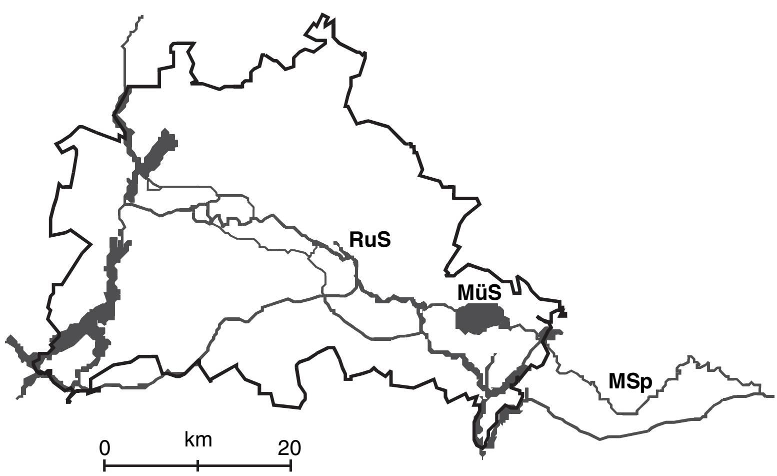

Fig. 4 Location of the sampling sites in two urban flushed lakes, Rummelsburger (RuS) and Müggelsee (MüS) and in the Spree River before entering Berlin

Table 4 Estimated number of effective migrants per generation between neighbouring sites (in parentheses: geographical distance in km / number of weirs between). For sites see Fig. 4

Species	MSp / MüS	MüS / RuS
	(13 / 0)	(13 / 0)
Common bream	5.2	6.3
Silver bream	1.8	10.6
Roach	6.9	3.7
Rudd	0.8	3.0

active spawners per generation (Slatkin & Barton 1989). In the cyprinid species studied, the criteria for a panmictic population will be fulfilled at one to two effective migrants per year only, due to the generation intervals of 2–3 years in roach, 3–4 years in silver bream and rudd, and 4–5 years in common bream. The calculated theoretical gene flow indicated significantly reduced migrations between the urban, channelized river stretch and the more natural Spree River before entering Berlin (MSp, compare Fig. 4) in silver bream and rudd (Table 4).

The value observed in rudd was the absolutely lowest theoretical gene flow of all: 0.8 effective migrants per generation roughly correspond to one spawner every five years entering the urban stretch from upstream rural parts. In addition, roach and rudd subpopulations showed a substantially reduced gene flow in the urban stretch between the sites MüS and RuS (Table 4, for sites see Fig. 4). The observed gene flow pattern did not correspond with geographical distances between sites. The 13 km long channelized river stretch between MüS and RuS restricted the theoretical gene flow similarly to the 155 km long reference stretch in the Oder (3.6 effective migrants per generation between the most upstream and downstream ends in common bream, 3.3 in roach, 10.6 in silver bream). This confirms the conclusions from presence-absence data by Wolter & Vilcinskas (1998b) suggesting a barrier effect of canals and channelized urbanized water stretches. Similar observations were reported by Guinand et al. (1996) from a study of chub in the Rhône basin. They found a strong correlation between genetic and geographic distance in chub from the natural parts of the Rhône River, but no correlation between both distances in chub from the regulated lower Rhône. Mean heterozygosity should increase downstream with absolutely increasing populations of spawners. In contrast, at the most downstream situated site in the urban area of Berlin (RuS) the lowest genetic variability was detected in all four species: in common bream $P_{95} = 16\%$, $H_{obs} = 0.087$; in silver bream $P_{95} = 12\%$, $H_{obs} = 0.049$; in roach $P_{95} = 16\%$, $H_{obs} = 0.057$; and in rudd $P_{95} = 12\%$, $H_{obs} = 0.067$. This loss of genetic diversity was interpreted as an effect of urbanization resulting from both, limited recruitment success and related genetic bottlenecks in urban waters, and restricted individuals exchange. The population genetic studies underlined the barrier effects for fish migrations caused by urbanized water bodies.

Conclusions

In developed urban areas with improved waste water treatments, the water quality is rarely the limiting factor for fish abundance and distribution, but the structural degradation of essential habitats is centrally important (Wolter et al. 2003). Essential habitat structures for fish become bottlenecks and limit fish spawning, recruitment or productivity and therefore, habitat degradation has to be considered as one principal factor of how urbanization impacts fish. The effects on fish assemblages becomes further intensified by altered migration abilities restricting individuals exchange as well as the accessibility of compensatory habitats in the watershed.

It has been argued, that at places where urban development is virtually complete and biological condition at its worst, rehabilitation efforts are unlikely to much improve biological condition

(Booth et al. 2004). However, the findings mentioned above give opposite evidence. If at a very high level of artificial embankment a further reduction of the remaining 10% structured habitats causes a significant decline of fish, then we may also infer that the rehabilitation of 10–20% of the bank line might significantly improve fish abundance and diversity. At least modest improvements seem fully achievable. Improvements in heavily degraded areas can also reduce downstream effects and rehabilitate downstream reaches.

Research needs to both assess the quantity of structural diversity and habitat patterns required to sustain a productive and diverse fish community, and identify the most efficient, technical solutions and measures to improve habitat quality of urban waters for fish by meeting their multiple human uses and social services.

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Bat Activity in an Urban Landscape: Patterns at the Landscape and Microhabitat Scale

Stanley D. Gehrt and James E. Chelsvig

Abstract Relatively little attention has been devoted to the urban ecology of bats (Chiroptera) despite their ecological importance. Although previous studies have indicated that urbanization has a negative effect on the abundance of bats and bat activity, this relationship may differ among regions. We monitored bat activity during 1997–1999 in 15–20 natural areas distributed across a 3500-km^2 area spanning the Chicago metropolitan area in northeastern Illinois. Our objectives were to elucidate relationships between landscape and microhabitat characteristics and bat activity. Bat activity was correlated with visual estimates of abundance. Among adjacent land-use classes, industrial/commercial use was positively related to bat activity in 1997 and 1999, and the predominant rural land use, agriculture, was negatively associated with bat activity in 1998. Proportion of woodland habitat within study areas was positively related to bat activity in every year. There was a positive relationship between agricultural land use and relative use of water sites in each year. Microhabitat analyses revealed that distance between trees was positively related to bat activity in woodlands, and in open habitats there was a relatively strong, negative relationship between distance from the edge and bat activity. Within open habitats, mowed areas had more bat activity than agricultural areas. Our landscape results suggest that the relationship between urbanization and bats may be related to context. Heterogeneous urban landscapes may represent islands of habitat for some bats within larger landscapes dominated by intensive agriculture, such as much of the Midwest.

Keywords: bats · Chiroptera · Illinois (USA) · landscape · microhabitat · urban ecology.

Introduction

The process of urbanization, or development of rural or natural habitats, dramatically transforms landscapes (Forman and Godron 1986) and the composition of flora and fauna (Gilbert 1989). Urbanization is generally considered to have a negative effect on natural ecosystem processes, and is a leading cause of endangerment for species in the continental United States (Czech and Krausman 1997). Urbanization fragments natural areas into patches that are disjunct and dispersed in a human-dominated landscape (Forman and Godron 1986, Wilcove et al. 1986). Eventually, landscape scale effects may outweigh local effects on the distribution and abundance of wildlife in natural areas within an urban matrix. In addition to their value as refuges and importance to conservation,

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habitat fragments provide an opportunity to address larger ecological principles such as landscape or gradient effects of urbanization (McDonnell and Pickett 1990).

Landscape effects, in the form of adjacent urbanization, appear to influence the abundance of some bird species in habitat fragments in coastal southern California (Soulé et al. 1988, Bolger et al. 1997). The importance of adjacent natural habitat, or proximity to rural areas, for conservation of native species in urban landscapes also has been illustrated for terrestrial fauna, such as ground arthropods (Davis 1978), lizards (Germaine et al. 1998), and mammals (Dickman 1987, Dickman and Doncaster 1989). However, some studies of avian communities have found that some bird species tend to colonize urban habitats regardless of the surrounding landscape and levels of urbanization (Thompson et al. 1993, Clergeau et al. 2001). These contrasting results suggest landscape effects on certain wildlife species may vary by metropolitan area, or more likely may be scale dependent (Forman and Godron 1986, Clergeau et al. 2001). Also, flight may allow some species to navigate urban landscapes and exploit habitat patches that are otherwise isolated (Gilbert 1989).

The effects of urbanization on bats, particularly on a landscape scale, are poorly understood in North America. Previous research has suggested urbanization may be detrimental to bats (Geggie and Fenton 1985, Kurta and Teramino 1992), with the resulting inference that urbanization results in dramatic declines in diversity and abundance of bats (Pierson 1998). Nevertheless, bats occur in cities, as indicated by reports from nuisance wildlife professionals and observations of use of streetlights by bats (Geggie and Fenton 1985, Brigham et al. 1989), and some of the habitat requirements of bats (e.g., roost sites and water) are present in urban areas (Everette et al. 2001). In addition, the effects of urbanization on bats may be context-specific (Fenton 1997); i.e., it may depend on the condition of the rural habitat and the pattern of development.

We report on spatial patterns of bat activity in patches of natural areas exposed to an extreme gradient of urbanization, namely the Chicago region in northeastern Illinois. This region is a large urban metropolis composed of $\sim 8 \times 10^6$ human residents spanning all or portions of six counties in northeastern Illinois. An important feature of this region is the number of preserved habitat fragments protected from development, most of which are county forest preserves. For example, Cook County, Illinois, contains $\sim 5 \times 10^6$ people and the city of Chicago, yet 11% of the land area in the county is composed of county forest preserves that are permanently protected from development. These forest preserves are assorted patches of natural areas scattered across the landscape, and because of their prominence in the Chicago landscape, there is considerable interest in determining the role of open spaces in the conservation of biodiversity in the area. This dichotomy of intense urban development and patches of protected areas also provides an excellent opportunity to assess the relative effects of each on bat activity in an urban landscape.

To better understand the complex relationship between urbanization and biodiversity and the general urban ecology of bats, we addressed the following objectives. First, we addressed the hypothesis that urbanization has a negative effect on bat abundance by predicting a negative relationship between adjacent urbanization and bat activity within habitat fragments in a heavily urbanized landscape. Our assumption was that spatial patterns of bat activity vary at the landscape scale along an extreme urban gradient. Second, we determined if there is a relationship between relative habitat use by bats and surrounding landscape characteristics because some surveys attempting to elucidate bat activity over a large spatial scale were conducted in only one habitat type (Kurta and Teramino 1992), and may have missed shifts in habitat use in different landscapes.

Finally, we test the prediction that microhabitat characteristics have an important effect on bat activity irrespective of landscape characteristics. The possibility exists that in a heavily urbanized landscape, anthropogenic effects from adjacent areas may confound the influence of local habitats on bat activity within study areas (Kurta and Teramino 1992). Alternatively, the marked vagility of bats may mitigate some negative aspects of urban landscapes and allow individuals to exploit favorable microhabitats within habitat patches. In a landscape-scale study in Great Britain (Walsh and Harris 1996), microhabitat effects were relatively important for explaining spatial patterns of bat activity. In any event, patterns of microhabitat use may help explain patterns of bat activity at a larger spatial scale.

Study Area and Methods

Study area

The Chicago metropolitan area, including > 260 municipalities, extends across all or part of six counties (Cook, DuPage, Kane, Lake, McHenry, Will) in northeastern Illinois and has a cumulative human population $> 8 \times 10^6$ people. General land cover in 1997 for this region was estimated to be 33% agriculture, 30% urban, 16% natural areas, and 21% unassociated vegetation (Wang and Moskovits 2001). Natural areas (including savannas, woodlands, grasslands, and wetlands) were highly fragmented, first by agriculture in the early 1800s, and more recently through urbanization. The extensive process of urbanization has produced a dynamic landscape in these counties, especially recently. During the 25-year period between 1972 and 1997, urban land increased 49%, natural areas decreased 21%, and the greatest loss was agricultural lands with a 37% decrease (Wang and Moskovits 2001). Of natural habitats, woodlands have the greatest land cover, with most of this habitat occurring as an extensive patchwork of fragments within a 15–45 km band around the urban core, and decreasing appreciably outside this band in the rural landscape dominated by agriculture (Wang and Moskovits 2001).

Bat monitoring

We selected 15 study areas in Cook and Kane Counties for bat monitoring in 1997, and added five additional study areas in McHenry County in 1998 and 1999. The 20 study areas occurred within an overall 3500-km^2 area extending from Indiana to Wisconsin borders (Fig. 1). Study areas were arbitrarily selected based on landscape characteristics, dispersion across the Chicago region, logistical constraints, and safety for equipment and personnel. Our goals were to obtain a sample from areas exposed to different levels of urbanization and to have areas with a variety of characteristics such as fragment size and habitat composition. All study areas were disjunct from each other, with the exception of two study areas that were located within a large continuous complex of forest preserves.

We used a stratified random procedure to select 10 permanent monitoring sites within each study area. We randomly located sites within three general habitat types: four sites were located in open areas, four in woodlands, and two near water. All sites within a forest preserve were separated by $> 100\,\mathrm{m}$. Woodland and open monitoring sites were located $> 50\,\mathrm{m}$ from the edge of the habitat (including water), and woodland sites often were situated on a bike trail or other small pathway. Water sites were located $< 10\,\mathrm{m}$ from the edge of a lake or stream. In all but a few cases ($< 5\%$) we used the same locations for monitoring sites each year. Known bat colonies occurred in two study areas, but no monitoring sites were located near these colonies.

At all monitoring sites, we recorded bat calls with broadband ultrasonic bat detectors (AnaBat II; Titley Electronics, Ballina, New South Wales, Australia) to survey relative activity (Thomas 1988, Krusic et al. 1996, Hayes 1997, Humes et al. 1999). At each monitoring site, a bat detector and audiocassette recorder (Optimus model, Radio Shack, Fort Worth, Texas, USA) were placed in a protective box, and the box was hung $\sim 1.5\,\mathrm{m}$ above the ground with a tripod. Detector sensitivity was assessed with an electronic flea collar (KLT Investments, Miami, Florida, USA) and calibrated for each bat detector prior to the field season, and periodic checks were conducted during the season. Each detector was set to a division ratio of 16 and sensitivity of 8.

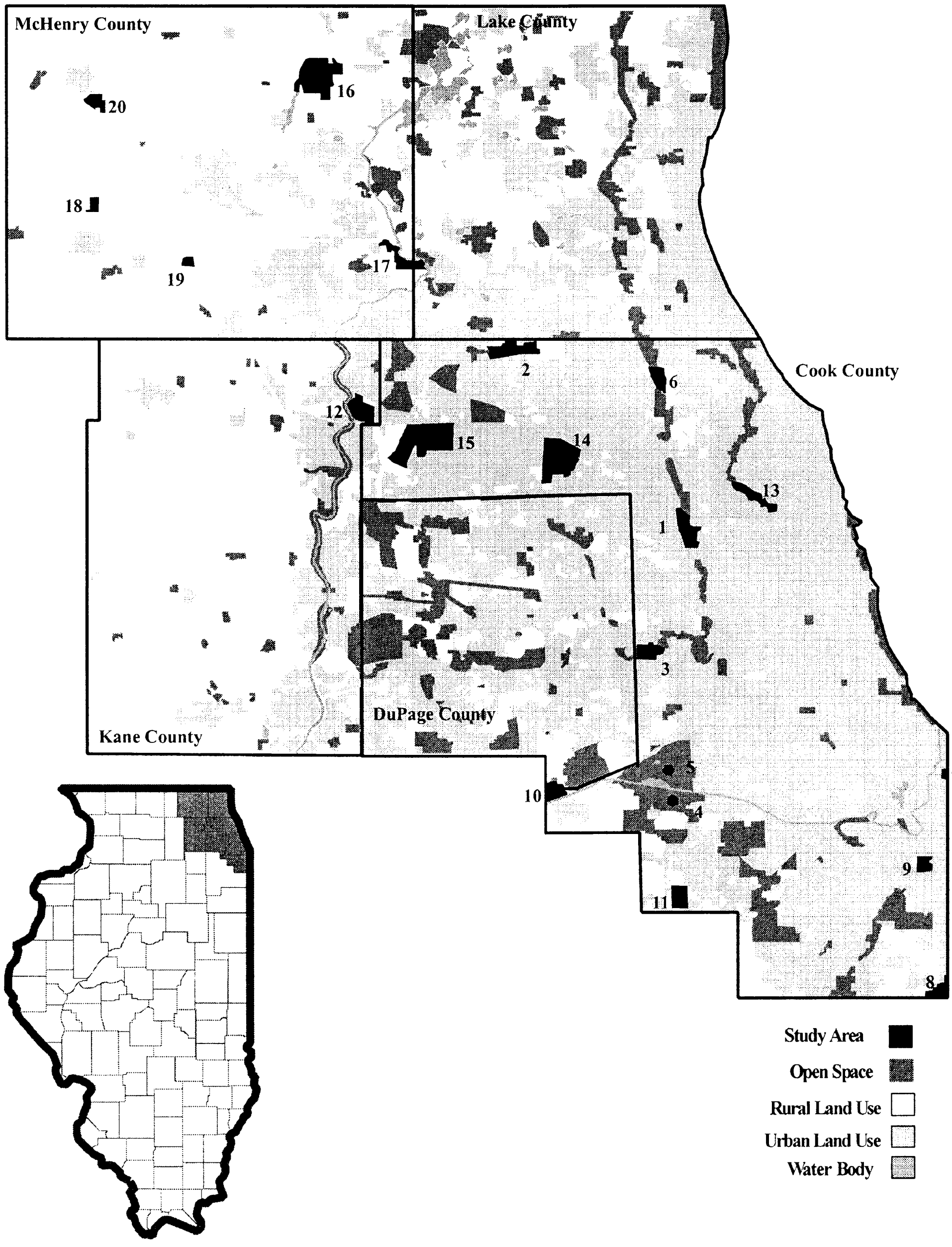

Fig. 1 Distribution of study areas and land use patterns in the Chicago, Illinois, region. The map was generated from unpublished 1995 land-use data obtained from the Northern Illinois Planning Commission. Numbers refer to the study areas in the Appendix.

We monitored bat activity during 6 July–15 October 1997, 6 June–29 September 1998, and 6 June–18 September 1999. Our monitoring protocol was essentially identical in each year. During each monitoring night (3 or 4 nights/week), study areas were selected randomly (without replacement) and bat detectors were placed at five of the 10 monitoring stations for each study area (two woods, two open, and one water site). Monitoring was conducted simultaneously for 2–4 study areas each night, and monitoring periods began at sundown and terminated 3 h later. During a monitoring session, bat detectors were checked each hour to calibrate the recordings (by pressing the calibration button on the detector) and to change batteries or tapes as needed. If precipitation forced us to terminate a monitoring session prematurely, those data were discarded and the study areas were monitored later in the week.

After each of the study areas had been monitored once, we began the process again by randomly reselecting the study areas and monitoring the alternate five sites on each study area. In general, all study areas were monitored within a week from each other for each rotation throughout the season. This monitoring design ensured equal sampling effort among study areas throughout the season.

Tapes were analyzed with an Anabat V zero crossing analysis interface module and the computer programs AnaBat5 (version 5.7) and AnaLook (Titley Electronics). We tabulated passes, or call sequences, composed of two or more calls (White and Gehrt 2001). Calls separated by $> 1\,\mathrm{s}$ were counted as separate passes (Hayes 1997). Occasionally, multiple bats passed over the detector simultaneously and these were considered as multiple passes.

Relative activity was determined for each study area by calculating the mean number of passes per night for each monitoring site, and then obtaining an overall mean for all monitoring sites within a study area. A two-way, fixed analysis of variance (ANOVA) with study area and year as main effects and an interaction term was used to determine if bat activity varied among study areas, and if the pattern of variation was consistent between years. Because the number of study areas differed between 1997 and the following two years, it was necessary to perform two levels of analysis: we removed the additional study areas from 1998 and 1999 and included these reduced data sets for comparisons among all three years. Second, we used the full data sets for 1998 and 1999 to compare bat activity among study areas and between these two years.

As an additional attempt to supplement electronic monitoring and determine the relationship between bat activity and abundance, we recorded conservative visual estimates of bat abundance at monitoring sites during monitoring sessions in 1997. As tapes were calibrated and checked each hour, technicians observed the monitoring site for bat activity. If bats were seen, a minimum number of individuals observed simultaneously was recorded for each monitoring site. If bats were observed at different times, they were not recorded separately unless they were of different sizes. Although visual counts have been used previously to survey bats in urban areas (Gaisler et al. 1998), our visual observations undoubtedly represent underestimates of the actual number of bats occurring at monitoring sites, and represent rough estimates of bat abundance at each site.

Landscape effects

We estimated several metrics to determine if certain landscape features were related to bat activity among study areas (see the Appendix). We used data from the Critical Trends Assessment Project Land Cover 1995 Database of Illinois (Illinois Department of Natural Resources 1996) to identify land use classes adjacent to study areas. The land cover database was derived from Thematic Mapper (TM) satellite imagery from Landsat 4 (ground resolution $28.5 \times 28.5\,\mathrm{m}$) taken from 1992 to 1995. Using ARCVIEW we constructed 2-km boundaries around the borders of each study area and we used the geographic information system (GIS) covermap and USGS 7.5 minute orthophoto maps to identify land-use characteristics and estimate the percentage land use within the boundary. We selected this scale because there was little change in variables at larger intervals (e.g., 5 km), and this scale helped maintain independence among study areas. The original database delineated 19 land cover classes, and we combined these classes into the following land-use types: industrial/commercial, residential, agricultural, and a general open category that includes open space, water bodies, and quarries.

In addition to adjunct land use, we measured the size of the study area, linear distance between the center of the study area and downtown Chicago (i.e., Union Station), linear distance to next nearest open space, average number of mercury-vapor streetlights per kilometer (white lights) on 3–4 adjacent streets, and average number of all types of streetlights per kilometer. We also estimated traffic volume and human population density adjacent to each study area. Estimates of 1997 traffic volume were obtained from the Illinois Department of Transportation, and an average from 3–4 roads nearest the study area (one road in each cardinal direction for which traffic data were available, and $< 0.5\,\mathrm{km}$ from the study area) was calculated for each study area. Some areas only had three adjacent roads. Traffic data represents the average number of vehicles that pass by in 24 hours on adjacent roadways.

U.S. Census Bureau data for 1990 were obtained for each zip code within 2 km of each study area, and the number of residents in the zip code adjacent to each study area was scaled relative to the proportion of the zip code area that was encompassed by the 2-km boundary. Finally, we used aerial photos and 7.5 minute USGS topographic maps to estimate the proportion of the three major habitat types within study areas: woodland, open, and water.

We did not have a priori knowledge of what urban factors might directly affect bats; therefore we did not always select landscape characteristics in the belief there might be a direct effect on bats, but rather some variables (e.g., traffic) were used to obtain an indirect measure of urbanization and human activities. We used multiple linear regression to determine relationships between relative activity of bats and landscape characteristics. We used Pearson correlation matrices to identify correlations among independent variables, and the variance inflation factor to identify multicollinearity among the remaining combinations of variables in the models. Initial model selection was performed with a backwards stepwise procedure. We validated the model selected by the stepwise procedure by using a complete multiple regression analysis on all subset variables. Mallow's Cp was used to evaluate the sum of squares for each subset model against the mean squared error for the full model. Cook's *D* was used to evaluate the effect of outliers and normality and constant variance tests were performed to determine if the data satisfied assumptions of distributions.

Because of high multicollinearity among multiple variables, we analyzed subsamples with different combinations of variables but kept correlated variables in separate models. If a particular variable was consistently selected as important in multiple submodels, it was included in the final model for variable selection. It was necessary to remove traffic and population density from these analyses due to multicollinearity with many variables, and we pooled adjacent habitat variables and quarries into the open class to simplify the analyses. We partitioned the regression analysis by year because additional study areas were added in 1998 and activity levels differed (see *Results*) among years.

We were concerned that multicollinearity prevented certain variables from occurring together in a model, yet a combination of these variables may have provided a better measure of urbanization than each variable individually. Therefore we developed an urban index that combined multiple variables associated with urbanization that also exhibited multicollinearity. The urban index for each study area was calculated as follows:

$$\frac{1}{[(\mathrm{IC} + \mathrm{RD}) \times \mathrm{PD} \times \mathrm{TV}] \times 100}$$

where IC and RD is the proportion of adjacent land use in industrial/commercial use and residential use, respectively, and PD and TV are the relative proportions of human population density (PD) and traffic volume (TV) among all study areas. We log-transformed the urban index to use in a simple linear regression with bat activity as the dependent variable.

A pilot study indicated relative differences in habitat use among study areas. Therefore, for each study area we determined the proportion of the total number of passes by habitat type to determine if relative habitat use of the three general habitat classes was related to types of adjacent land use. Proportions of bat activity were arcsine square-root transformed and we followed the same regression procedures outlined above for each habitat type (i.e., woodland, water, open).

Microhabitat effects

In 1998 and 1999, we measured microhabitat variables at each monitoring site. We did not block microhabitat variables by study area because variations in bat activity and habitat characteristics within study areas were as great or greater than variations between study areas. That monitoring sites were effectively independent was also suggested by results from bat activity recorded during a pilot study in 1996. For woodland sites, we used the point-quarter method to measure the average distance from the monitoring site to the nearest tree and average diameter at breast height (dbh), with a minimum criterion of 4-cm dbh. We visually estimated percent canopy immediately above the monitoring site. Because our sampling did not extend beyond the monitoring site, our woodland metrics pertain only to the site and do not necessarily characterize the overall woodland habitat. Multiple regression was used to determine the relationships between woodland characteristics and bat activity.

We classified each water site as moving or standing water, and classified the width of the water site into one of four classes (<5, 6–10, 11–20, >20 m). The height of vegetation at the edge of the water site was measured with a vertical coverboard marked into four 46-cm heights. Height of vegetation was assessed with regression, and types of water and width classes were compared with ANOVA.

For monitoring sites in open habitats, we recorded height of vegetation, ground cover, and distance to nearest edge (e.g., water, or usually woodland). For each site we identified four sampling points at a distance of 7 m in four directions, and estimated vegetation cover at four 46-cm heights with a vertical coverboard. A 1-m^2 frame was placed at each sampling point to estimate percent ground cover of vegetation. Linear distance from the monitoring site to the nearest woodland or water edge was measured to assess edge effects. Finally, each open site was classified as old field, restored prairie, cropland, or mowed picnic area/ developed area. We used multiple regression to determine relationships of distance-to-edge, ground cover, and vegetation height with bat activity, and ANOVA was used to assess differences in bat activity among classes of open sites.

Results

General bat activity

We monitored for a cumulative total of 6810 detector-hours (one detector for one hour) and 7–9 monitoring nights per study area within each season. In 1997 we monitored each study area for nine nights (2010 detector-hours), and in 1999 each study area was monitored for eight nights (2400 detector-hours). All study areas were monitored for eight nights in 1998, except one was monitored for seven nights and another for nine nights (2400 detector-hours). We recorded 52 818 passes by bats (1997, 8336; 1998, 21201; 1999, 23 281) and the number of passes per study area within a season ranged from 2.0 ± 1.8 to 118.3 ± 128.0 passes/study area (mean + 1 SD, Fig. 2). Number of passes for each monitoring site within a single monitoring period ranged from 0 to 909 passes. Mean number of passes by bats varied among study areas in each year (1997–1999 model, $F = 5.80$, df = 14, 427, $P < 0.001$ 1998–1999 model, $F = 3.74$, df = 19, 280, $P < 0.001$), and patterns of this variation were consistent between years (all $P > 0.1$ for the interaction term of both models).

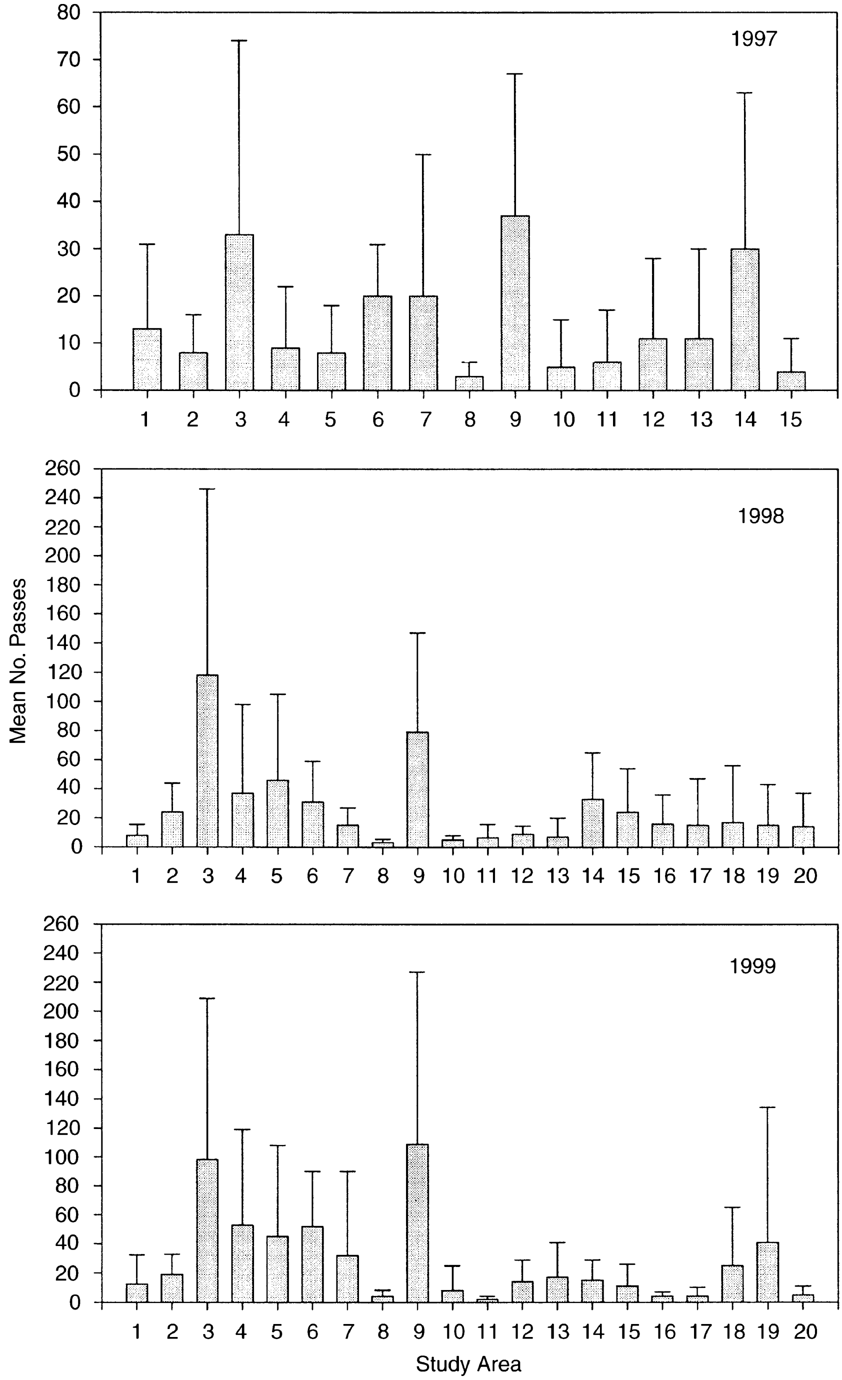

Fig. 2 Frequency distributions of bat activity, expressed as mean (+1 SD) number of passes, for study areas in northeastern Illinois, 1997–1999. The numbers of the study areas correspond to those in Fig. 1.

However, bat activity varied annually ($F = 11.06$, df $= 2, 427$, $P < 0.001$), with less ($P < 0.05$) activity recorded during 1997 than the following two years. Bat activity did not differ ($F = 0.05$, df $= 1, 280$, $P = 0.825$) between 1998 and 1999.

The number of bats simultaneously observed at monitoring sites ranged from 0 to 10 bats. Number of passes for each study area was correlated ($r = 0.80$, $P < 0.001$, $n = 14$) with mean number of bats observed within each study area. When we treated monitoring sites as independent observations, the correlation coefficient between visual observations and number of passes was $r = 0.70$ ($P < 0.001$, $n = 140$). For the three study areas with the greatest activity (numbers 3, 6, and 9, Appendix), correlation coefficients ranged from 0.54 to 0.80 (all $P < 0.001$) between number of passes and number of observed bats at monitoring sites within study areas.

Landscape analysis

Few landscape characteristics, or combinations of characteristics, were related to overall activity among the study areas. Full models and various submodels consistently reduced to 2–3 variables each year (Table 1). A consistent result among all years was a positive relationship between woodland habitat and bat activity. Among adjacent land-use classes, industrial/commercial was significant in 1997 and 1999, with positive relationships with bat activity in both years, and agriculture as a negative coefficient was included in the model for 1998. Industrial/commercial and agriculture classes were never included into a model together because of high autocorrelation between them. Exchanging industrial/commercial use with agricultural use produced a model with $R^2 = 0.53$, $P = 0.002$, $n = 20$ in 1999, with a negative coefficient (−0.425) for agriculture. Similarly, the submodel with the next-highest R^2 (0.44) in 1998 was $Y = 0.146 + 0.482(\text{industrial/commercial}) + 3.367\ (\text{size}) + 1.031$ (woodland). This pattern was not true for 1997 (when five of the study areas with the greatest amount of adjacent agriculture were not monitored); although agriculture consistently had negative coefficients, all models containing agriculture had considerably lower R^2 values (such as $R^2 = 0.47$ for the model with only agriculture and woodland). A positive regression between the urban index and bat activity occurred in 1997 ($R^2 = 0.49$, $P = 0.004$, $n = 15$), but the relationship was not significant in 1998 ($R^2 = 0.04$, $P = 0.424$, $n = 20$) and 1999 ($R^2 = 0.06$, $P = 0.306$, $n = 20$). Thus, a consistent trend in these analyses was a positive relationship between woodland habitat in fragments and bat activity, and also between adjacent urban land use and bat activity.

When we partitioned bat activity by habitat type, there was a positive (all $P < 0.001$) relationship between agricultural land use and relative bat activity at water sites in each year (Fig. 3; 1997, $R = 0.79$, $n = 15$; 1998, $R = 0.82$, $n = 20$; 1999, $R = 0.63$, $n = 20$). No consistent patterns among years occurred between landscape characteristics and relative use by bats of woods and open habitats.

Table 1 Regression models of overall bat activity and landscape characteristics in northeastern Illinois.

				Coefficients			
5-8 Year	N	R^2	P	B_0	B_1	B_2	B_3
1997	15	0.765	< 0.001	−0.188	1.575 IC	0.862 WD	
1998	20	0.452	0.019	0.441	−0.222 AG	0.0003 SIZE	0.942 WD
1999	20	0.596	0.002	0.096	1.214 IC	1.248 WD	

Note: Landscape characteristics were adjacent industrial/commercial use (IC), agricultural use (AG), size of study area (SIZE), and proportion of woodland habitat within the study area (WD).

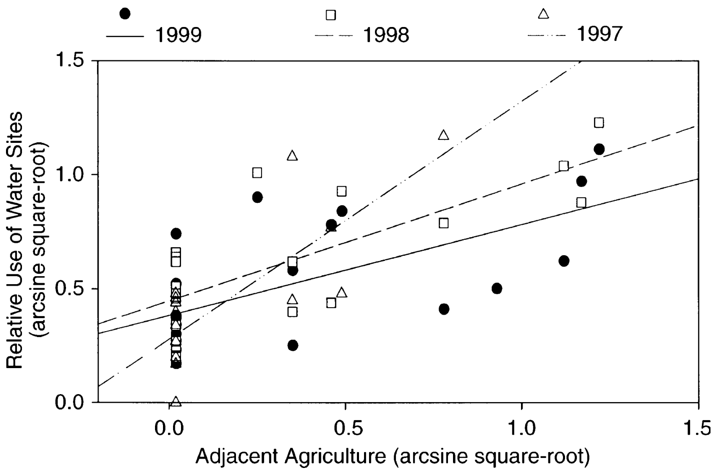

Fig. 3 Relationship between the amount of agricultural land use adjacent to study areas and the relative use of water sites by bats during 1997–1999. Regressions in each year are significant ($P < 0.001$).

Microhabitat analysis

At woodland sites, distance between trees was positively related to bat activity in both years, whereas dbh and canopy cover were not significant in either year (Table 2). For water sites, vegetation height was not related to bat activity (Table 2), nor was width of water site (1998, $P = 0.364$; 1999, $P = 0.658$). There was a difference ($t = -2.157$, df $= 38$, $P = 0.037$) in mean (± SD) bat activity between types of water site in 1998 (standing water: $\bar{x} = 0.944 \pm 0.65$; moving water: $\bar{x} = 0.335 \pm 0.96$). However, there was no difference ($t = -1.406$, df $= 38$, $P = 0.168$) between water types in 1999, and the pattern among means (standing water: $\bar{x} = 1.069 \pm 0.47$; moving water: $\bar{x} = 1.336 \pm 0.66$) contrasted with data from 1998.

In open areas, there was a relatively strong, negative relationship between distance from edge and bat activity in both years (Table 2), and a negative relationship between vegetation height and bat activity in both years. There was no significant relationship between ground cover and bat activity in 1999, but there was a significant relationship in 1998 (Table 2). Additional inspection of regression models suggested the most important variable for explaining variation in bat activity at open

Table 2 Partial-regression coefficients and associated P values for microhabitat variables and bat activity during 1998 and 1999 in northeastern Illinois.

	1998		1999	
Habitat, variable	Coefficient	P	Coefficient	P
Woods				
Distance between trees	0.114	0.004	0.147	0.002
Dbh	0.005	0.811	0.026	0.323
Canopy cover	0.065	0.851	−0.537	0.267
Water				
Ground cover	−0.125	0.714	−0.160	0.314
Open				
Vegetation height	−0.620	0.013	−0.320	0.005
Ground cover	−0.740	0.003	0.099	0.808
Distance from edge	−0.945	< 0.001	−0.921	< 0.001

Notes: Separate models were constructed for each habitat type. See *Methods* for descriptions of the variables. Dbh = diameter at breast height.

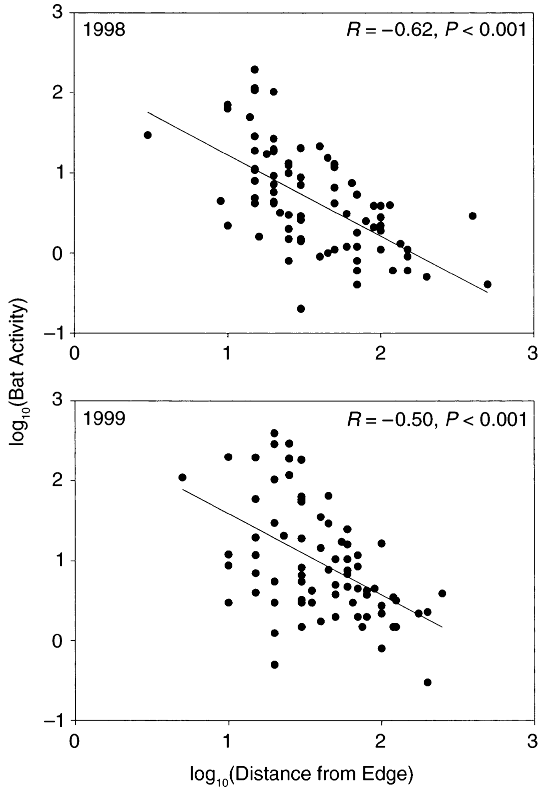

Fig. 4 Regressions between distance from edge (measured in meters) and bat activity (measured in mean number of passes/night) at open sites during monitoring in 1998 and 1999.

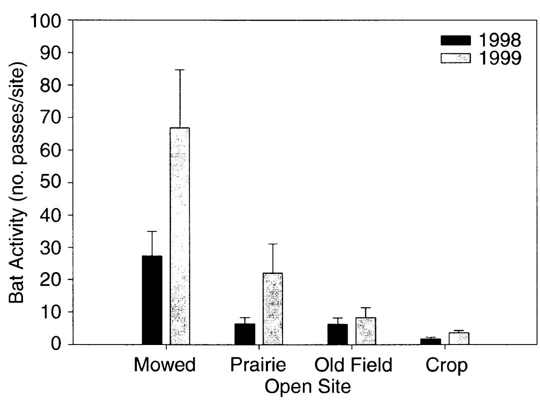

Fig. 5 Distribution of the number of passes by bats (mean +1 SD) by type of open site during monitoring in 1998 and 1999.

sites was distance from edge (Fig. 4). Comparisons of bat activity among open site classes revealed significant differences in both years (1998, $F = 3.85$, df = 3, 68, $P = 0.013$; 1999, $F = 4.43$, df = 3, 68, $P = 0.007$), with more activity in mowed areas than in crop and old fields (Fig. 5).

Discussion

We documented a positive relationship with certain types of urban land use (i.e., industrial/commercial) with bat activity in habitat fragments, in contrast to previous studies. However, because of strong autocorrelation between industrial/commercial and agriculture land-use classes, we were not able to determine the relative importance of each land use type, exclusive of the other, for bat activity. Woodland habitat appeared to be important, with a consistent positive relationship between woodland habitat and bat activity each year. Woodlands are important to a variety of bat species by providing foraging habitat (Grindal and Brigham 1999), roosting habitat (Hayes 2003), or escape cover from predators (Zimmerman and Glanz 2000), and this habitat is relatively rare in the rural landscape of Illinois.

Relatively more activity was near water in rural study areas than in more urban areas. A possible explanation for this shift in habitat use is that in extensively cultivated areas, standing water is relatively rare, or more widely dispersed, compared to more urbanized areas. By the early 1900s, many of the rural areas in Illinois were drained of their wetlands, and streams channelized, for cultivation (Havera et al. 1997). Open habitats frequently had the greatest bat activity among study areas including urban areas, although this was not always the case. Previous studies that focused only on one habitat type such as water (e.g., Kurta and Teramino 1992) may have missed shifts in habitat use by bats in response to land use changes.

Microhabitat effects were relatively more obvious for explaining overall bat activity than landscape effects, and the patterns we observed were consistent with other studies (Krusic et al. 1996, Zimmerman and Glanz 2000). We did not evaluate woodland characteristics beyond the specific monitoring site. Nevertheless, the positive relationship between distance between trees at the monitoring site and bat activity probably reflects the effect habitat “clutter” has on bats (Fenton 1990, Humes et al. 1999). In studies of forest systems, bats avoid structurally cluttered habitats and focus their foraging in less complex habitats presumably for the ease of navigation (Brigham et al. 1997, Grindal and Brigham 1998). Background clutter appeared to be reduced as distance between trees increased, thereby providing opportunities for use by more species of bats (Humes et al. 1999). Numerous studies have reported bats preferentially using edge habitats (see Hayes 2003). Bats use forest edges as foraging areas (Grindal and Brigham 1999) or as travel corridors (Krusic et al. 1996). Insect abundance may be higher at the interface between woodlands and open habitat, or navigation in the pursuit of prey may be easier along edges than woodland interiors (Grindal and Brigham 1999).

One reason why microhabitat effects were relatively strong as compared to landscape effects might be that flight as a mode of locomotion enables bats to easily move through the urban matrix, whereas landscape effects may outweigh microhabitat effects for nonvolant species. Thus, flight may minimize some of the physical effects of urbanization that prove challenging for other wildlife, and allows bats to exploit patches of favorable habitat. For example, roads act as barriers or otherwise negatively impact many species of wildlife such as herptiles and other ground dwelling species (Mader 1984, Fahrig et al. 1995, Findlay and Houlahan 1997), but probably do not hinder movements of bats.

Our results at the landscape scale differ from those of previous studies in North America that have reported a negative relationship between abundance and diversity of bat species, and bat activity, with urbanization (Geggie and Fenton 1985, Kurta and Teramino 1992). Kurta and Teramino (1992) measured community structure of bats in a park in suburban Detroit with mist nets, and concluded that urbanization can decrease species diversity and overall abundance relative to rural areas. However, their sampling of urban sites did not occur in the same year as rural sampling, and annual differences in bat activity such as we documented in our study might have affected their results. Also, their mist netting was restricted to water sites. If we had relied only on mist nets for sampling, our results may have been considerably different as many sites with relatively high bat activity were poor mist-netting locations. In our study, water sites with relatively high bat activity were associated with rural areas. Although we attempted to mist net at many of our monitoring sites with high activity, we typically observed multiple bats detecting and avoiding the nets because of their poor location. Consequently, our low capture success was not indicative of bat activity in the region, and we could not use mist net data for comparisons between areas.

Our results are most similar to an extensive survey of general bat activity in Great Britain (Walsh and Harris 1996), also conducted at a large spatial scale. In that study, microhabitat effects, or habitat at the local level, were relatively more important for explaining patterns in general bat activity than landscape effects. In particular, there were strong positive associations with woodland habitat and edge with bat activity. At the landscape level, there was a consistent negative trend between bat activity and arable land, and a mild positive trend between bat activity and urbanization (Walsh and Harris 1996).

The strong microhabitat effect we observed is similar to large-scale studies of urban bird communities, in which community structures in urban landscapes were independent from areas outside towns (Clergeau et al. 2001). Thus, a strong local scale effect might be typical of species capable of flight, such as birds and insects, which mitigates some landscape effects and allows them to exploit habitat fragments throughout the urban landscape (Gilbert 1989).

The most plausible explanation for the landscape patterns observed in our study (that is, a positive relationship between urbanization and bat distribution and activity), and perhaps explains differences among studies, lies in the landscape characteristics surrounding Chicago, and for most metropolitan areas in the Midwest. Although the ecological principle of context is typically applied to patch analysis (Zipperer et al. 2000), it can also be applied on a larger scale. Metropolitan areas typically resemble urban "islands" surrounded by a larger landscape. In Illinois, the rural landscape is predominantly intensive row-crop agriculture, with $\sim$ 85% of the statewide land area represented by farmland (Illinois Department of Agriculture, *unpublished data* [1999]). Among the counties represented in our study, two relatively rural counties (Kane, McHenry) had 63% and 68% land area in farmland, whereas only 4% of Cook County was farmland (Illinois Department of Agriculture, *unpublished data* [1999]). The agricultural-dominated landscape of Illinois typically has vast areas of monotypic vegetation and a limited number of woodlots and woodland edges. Within our study areas, agricultural habitat received the least amount of use by bats compared to other types of open habitats, and agriculture was the only type of land use to exhibit a negative relationship with bat activity.

There are multiple reasons why urbanized landscapes may be preferred by bats over agricultural landscapes in the Midwest. Given that an interface between habitats such as woods and open space is important to foraging bats (Findley 1993, Krusic et al. 1996, Zimmerman and Glanz 2000), urban areas in some landscapes may be preferred because there is relatively more woodland edge in some urbanized areas than in typical agricultural areas in Illinois. Similarly, there may be more roosting habitat, whether artificial or natural, in urbanized areas than in rural areas (Everette et al. 2001). *Eptesicus fuscus* and *Myotis lucifugus* occur in the Chicago region (S. D. Gehrt and J. E. Chelsvig, *unpublished data*), and both frequently roost in artificial structures such as buildings. Also, in addition to protected woodlands in forest preserves and parks, many older and upper-class communities contain numerous trees and other woody vegetation in yards and parks. Other species (e.g., *L. borealis*) that occur in the Chicago region have been reported to roost in urban areas (Mager and Nelson 2001), including parks near downtown Chicago (Meritt 1989; S. D. Gehrt, *personal observation*).

Agricultural practices such as pesticide use, in addition to agricultural landscape features such as limited woodland habitat, may have deleterious effects on bats. This, in addition to other factors such as lack of roosting habitat, may partially explain why we, and others, have reported lower bat activity in agricultural fields than other open habitat types (Furlonger et al. 1987, Walsh et al. 1995, Fenton 1997).

Finally, some species of bats often concentrate their foraging around mercury vapor lights, such as some street lights because these lights attract flying insects (Rydell 1992, Rydell and Racey 1995, Hickey et al. 1996). We did not conduct monitoring under or near light sources; however, there are more of these types of lights present in the urban landscape (although many have now been replaced with sodium lights) than in the rural landscape. Our measurements of lights were limited to street lights immediately adjacent to study areas, and there were many sources of lights that we were unable to quantify that were available in urbanized areas. Thus, there may be more opportunities to exploit artificial concentrations of insects in urban areas, even if urbanization itself has a negative effect on some insect species (Davis 1978, Frankie and Ehler 1978).

Caveats

A limitation of acoustic monitoring of bats is it does not provide measures of abundance, and caution should be used when inferring abundance from such activity data (Hayes 2000). Acoustic monitoring was necessary in our study as many sites with high levels of activity were not conducive to mist-netting or other types of capture, despite our attempts to do so. An advantage of monitoring in a large metropolis is that the night sky is perpetually lit by artificial lights, which facilitates visual observations and allowed us to determine a strong correlation between activity levels and minimum numbers of bats seen at monitoring sites. Thus, there appeared to be a relationship between minimum abundance and activity levels. However, because we could not identify individuals, our method is not definitive and the association between abundance and activity levels remains tenuous.

Our monitoring protocol, while rigorously designed to minimize as many potential sources of bias as possible, still has limitations. Although we stratified our sampling among habitat types, our general habitat categories within study areas grouped a wide range of habitat characteristics within the general categories. This variation may have affected bat activity and our ability to detect that activity, as we could not control clutter within habitat types (Hayes 2000), and relationships between habitat types and bat activity in urban areas should be investigated further. Also, our analyses elucidated relationships but not causal mechanisms. Thus, although we proposed possible mechanisms, we still have much to learn as to how urbanization affects the distribution of bats across the landscape.

We may have masked species-specific effects by pooling all bat activity into a general analysis. Furlonger et al. (1987) reported species' differences in activity patterns with respect to urbanization in Ontario. However, an analysis of a subsample of the data, in which we partitioned the data by species, revealed similar relationships to urbanization among species or species groups (S. D. Gehrt and J. E. Chelsvig, *unpublished data*).

We have modeled the relationships between bat activity and landscape characteristics as linear functions of urbanization; however, we did not monitor within the extreme urban core (the study areas located closest to downtown Chicago were 18 km away). It is important to note that while we consistently identified models with positive relationships between our measures of urbanization and bat activity, equally important were consistent positive slopes between woodland habitat and bat activity. Because undeveloped area and associated woodland habitat often is dramatically limited within the urban core, a curvilinear relationship between urbanization and bat activity, with an asymptote occurring near the extreme end of urbanization, may be more realistic than our linear models. For example general bat activity was relatively low in the city center of Brno, Czech Republic (Gaisler et al. 1998). However, although we did not monitor bat activity within the urban core, we occasionally captured bats (e.g., *L. borealis*) in downtown Chicago, saw evidence of small colonies in some downtown structures, and others have reported colonies or groups of various species regularly occurring in and near the core (Meritt 1989). Similarly, Everette et al. (2001) observed *E. fuscus* roosting in the urban core of Denver, Colorado, and foraging in a refuge some distance away.

Implications

While this study does not elucidate direct effects of urbanization on bat populations, our results suggest the relationship between urbanization and bat ecology is complex. The nature of this relationship probably depends on context at the macrogeographic scale and the pattern of development within the urban matrix (Fenton 1997). Habitat fragments in the urban matrix have potential value for bats despite the intensity of adjacent development. Given the strong influence of microhabitat effects on patterns of bat activity, local habitat management of urban open spaces may have an important role in the conservation of bats in urban landscapes, particularly in the agriculture-dominated landscape of the Midwest. Similarly, avian species also respond to habitat management at the local scale in urban areas (Clergeau et al. 2001).

Although our results should not be used to advocate development, they do suggest that urban development in the Midwest need not be deleterious to bats if it provides habitat that is otherwise limited in the rural landscape. Habitat fragments within urban landscapes in the Midwest may be important refuges for bats as well as other wildlife, and the conservation of woodlands, regardless of their proximity to the urban core, should be promoted.

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Appendix

A table with a summary of landscape characteristics for each study area in northeastern Illinois (USA) is available online in ESA's Electronic Data Archive: *Ecological Archives* A013-013-A1.

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Urbanization and Spider Diversity: Influences of Human Modification of Habitat Structure and Productivity

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Abstract As a part of the Central Arizona–Phoenix Long-Term Ecological Research project, we determined how land-use alteration influenced spider and harvestman diversity. We sampled spiders in six habitat types (desert parks, urban desert remnants, industrial, agricultural, xeric- and mesic-residential yards) and tested how habitat type and productivity affected spider diversity and abundance. As expected, agricultural fields and mesic yards were more productive than the other, xeric habitats. These more productive habitats were characterized by higher abundances but lower spider diversity and were dominated by *Lycosidae* (wolf spiders), followed by *Linyphiidae* (sheet-web weavers). The increase in wolf spider abundance was positively correlated with habitat productivity and negatively correlated with the abundance of other predatory arthropods that might compete with, or prey upon, wolf spiders.

Temporal changes in productivity affected spider abundance. After an El-Niño winter (May 1998), spider abundance was five times higher than after an extremely dry winter (May 2000). The differences in spider abundance between agricultural fields and the four xeric habitats were profound in 2000 but moderate in 1998, suggesting an interaction between the effects of natural and anthropogenic factors on spider populations. Compared with xeric habitats, the El-Niño effect was less profound in agricultural sites, suggesting that human land modification mollifies seasonal effects. We suggest that habitat structure and productivity alteration may change community structure, as the urban or agricultural habitats favor one or a few preadapted taxa over many others. Incorporation of large fragments of natural habitats into future landscape planning in urban environments may be important for conservation of rich spider communities.

Keywords: Arthropod communities · CAP LTER · diversity · spiders · El-Niño · Lycosidae · Sonoran desert · urbanization.

Introduction

Studying ecological patterns and processes in urban environments is a relatively new direction in ecology (Grimm et al. 2000). The lack of ecological studies in urban environments is especially crucial in the field of conservation biology (Miller and Hobbs 2002). During the last two decades, ecologists have made preliminary forays into urban ecology by studying how biological communities and populations change along urban–rural gradients throughout the world. To date, most studies on wildlife in cities have focused on birds (reviewed by Marzluff et al. 2001). We know much less

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about other vertebrates and almost nothing about the effect of human activities in heavily populated areas upon arthropod communities (McIntyre 2000). Furthermore, the few studies focusing on urbanization and arthropod communities have mainly tested the effect of fragmentation of natural habitats due to urbanization (Miyashita et al. 1998, Bolger et al. 2000, Gibbs and Stantos 2001, Gibb and Hochuli 2002). These studies focused on the changes in arthropod community composition in forest or scrub fragments, ignoring the arthropod communities that inhabit the urban habitat.

Studying arthropod communities in urban environments is important for several reasons. Habitats are becoming increasingly dominated by human-related factors and processes (Grimm et al. 2000, Miller and Hobbs 2002), yet most ecological studies focus on more natural and less human-altered ecosystems. Furthermore, it is critical to understand if different taxa respond to alterations in landscape structure in the same way. For example, urban bird communities increase in total abundance but decrease in species diversity compared to nonurban communities (Marzluff 2001). As the mechanisms are yet unclear (Marzluff et al. 2001), studying patterns in other taxa may help to generate hypotheses about the processes that shape urban wildlife populations. Another reason for studying arthropods in human-managed environments is that many arthropods are important in agriculture and gardens as pests or biological control agents and in medicine (McIntyre 2000). Among the arthropods, spiders are key predators that may also reflect changes in trophic structure in human-altered ecosystems.

Part of the Central Arizona–Phoenix Long-Term Ecological Research (CAP LTER) project is an on-going (since spring 1998) study on ground-dwelling arthropods. The goals of this study are to assess how arthropod communities change in space (between different urban and natural land-use types, or habitats) and time (different seasons and years). McIntyre et al. (2001) summarized the results from the first year of monitoring. Taxonomic richness and abundance were higher in agricultural fields and relatively low in desert sites. Functional groups showed differences in abundance between habitats. Predators and herbivores were most abundant in agricultural sites, whereas omnivores were equally abundant among sites (McIntyre et al. 2001).

In this study, we focus on the predator guild, specifically spiders (order Araneae) and daddy long legs (or harvestmen, order Opilionida). Spiders are abundant and dominant components of the arthropod predatory guild in most communities (Wise 1993), and may be highly influenced by habitat patchiness in general (Whitehouse et al. 2002) and anthropogenic changes in the ecosystem such as urbanization and habitat fragmentation in particular (Miyashita et al. 1998, Bolger et al. 2000). We ask whether spider abundance and diversity are influenced by habitat type within the urban ecosystem and whether differences in habitat productivity affect the spider community.

Productivity is a major environmental axis known to affect species diversity in general (Rosenzweig 1992). Although very productive habitats can support high population densities, for various possible reasons (reviewed by Rosenzweig and Abramsky 1993) their species diversity is often lower than moderately productive environments. The higher productivity in urban environments may be a major factor affecting biological populations (Emlen 1974, Marzluff 2001). Despite these claims, the productivity component and its effect on biological diversity in urban environments have been ignored. Here we test how changes in productivity in both time and space influence spider diversity and abundance. We examine possible bottom-up or top-down mechanisms underlying the observed patterns and suggest future directions for urban landscape planning strategies in terms of conservation of spider diversity.

We used six major habitat types in the greater Phoenix area and sampled spiders each month for three years: desert, urban desert remnants, industrial, agricultural, xeric urban yards, and mesic urban yards. To generate predictions, we placed each habitat on a state space of two major environmental axes: patch size (the result of desert fragmentation) and productivity (the result of desert habitat alteration into residential or agricultural habitats that receive supplemental water). The estimated location of each habitat on the state space and the relationships between the habitats are given in Fig. 1. Based on these criteria we made two predictions:

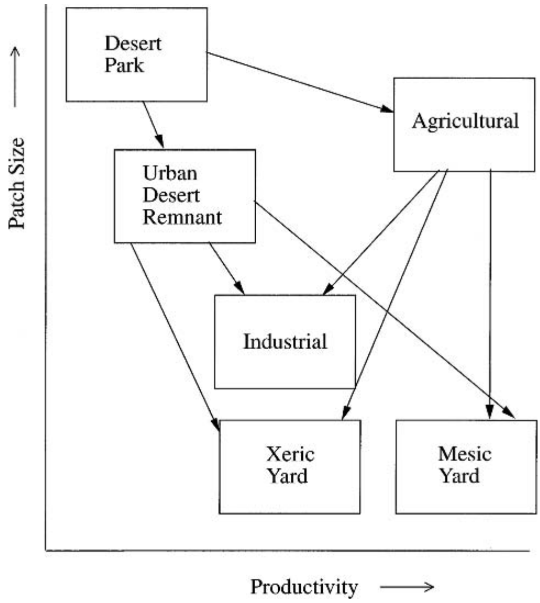

Fig. 1 Schematic showing the estimated location of the six studied habitats along productivity and spatial scale (patch size) axes. Arrows represent development processes in which habitat size, structure, and productivity are being modified

1) Spider diversity should peak in urban desert remnants and industrial sites. Diversity will be lower in: (a) undisturbed desert due to lower productivity, (b) xeric yards due to a decrease in patch size (fragmentation), and (c) mesic yards and agricultural sites due to an increase in productivity.
2) Spider abundances should be greater due to high productivity (a) in mesic yards and agricultural sites in comparison to the other four habitats and (b) during a wet El-Niño year (1998) compared with an unusually dry year (2000).

Methods

Study area

The ground arthropod population project of the CAP LTER has been designed to sample the major land-use types (habitats) in the greater Phoenix area. In the first year, arthropods were sampled in 16 sites classified into four habitats. McIntyre et al. (2001) defined and described in detail the criteria for habitat classification of these habitats. The sites covered a wide geographic range around Phoenix and represented the most abundant forms of land use in the area (McIntyre et al. 2001). The habitats were:

1) Urban desert remnants: characterized by Sonoran desert vegetation and lacking any built structures, these sites represent an early stage of desert fragmentation by urbanization.
2) Agricultural fields: mostly alfalfa fields, representing alteration of the extremely dry desert land into extremely productive and moist habitat.
3) Industrial sites: nonresidential commercial structures or warehouses surrounded by yards with Sonoran desert vegetation subjected to no or very little irrigation.
4) Xeric residential yards: A single-family house surrounded by xeriscaped yard (nonturf, gravel, and a high proportion of Sonoran desert vegetation subjected to irrigation).

   Though industrial sites and xeric yards may undergo changes in habitat structure, they are in most cases characterized by desert vegetation, and therefore can be treated as representing mainly a progressed stage of desert fragmentation. The differences between the industrial sites and xeric yards are usually in fragment size (xeric yards are smaller), isolation (due to fencing in xeric yards), and productivity, due to different irrigation regimes.

During the second year of study, eight sites and two additional habitats were included. Since spring 1999, arthropods were sampled in 24 sites (four replicates of each of the six habitats). During the study period, two sites had to be relocated due to development, bringing the total number of sites to 26:

5) Desert parks, included two new sites and two old sites. Though increasingly surrounded by suburbs, desert parks are still connected to the continuous Sonoran desert around the Phoenix metropolis. Accordingly, two new urban desert remnant sites were added to class 1.
6) Mesic yards, representing a different kind of a moist productive habitat than agriculture. Mesic yards are characterized by lawns (>50% of the total yard), high proportion of exotic plants, and irrigation systems.

The six habitats can be classified along a patch size/productivity gradient (Fig. 1). Understanding the processes that lead to the creation of each habitat along the axes of habitat fragmentation and productivity is important in order to understand how these processes may affect spider communities in an urban ecosystem.

Assessing the degree of fragmentation

We assessed the degree of landscape fragmentation by calculating the area of contiguous patch type (representing habitat) adjacent to, and including, each arthropod sampling location. A 1998 land cover classification of the Phoenix metropolitan area derived from Landsat Thematic Mapper (TM) data (Stefanov et al. 2001) was used as the base data set for the calculation of contiguous patch area. This land cover classification has the spatial resolution of TM data (28.5×28.5 m/pixel). The 12 original land cover classes of the 1998 classification were recoded into six patch types (desert, urban desert remnants, industrial, agricultural, xeric yard, and mesic yard) to ensure comparability with the existing patch classification scheme. The resulting raster data set was converted into a vector shapefile format in order to group similar patch type pixels together as single polygons using an edge-to-edge rule (i.e., diagonally adjacent pixels are not included in the total area calculation).

Arthropod sampling

We collected spiders and harvestmen (orders Araneae and Opilionida) with other arthropods using 10 dry unbaited pitfall traps at each of the 26 sites. Traps were spaced 5 m apart along a line transect (straight line, unless constrained by property boundaries at residential sites). Traps consisted of a 500-mL plastic cup stacked inside another cup, with the lip of the inner cup set flush to the ground surface. Traps were set for 72 consecutive hours each month. The inner cups of each trap and their contents were then removed without displacing the entire trap. When not in use, traps were tightly closed with plastic lids. Trap contents were collected and preserved in 70% ethanol, and spiders were identified to a family. Voucher specimens are housed in the Department of Biology at Arizona State University, Tempe.

Pitfall traps may have disadvantages when trying to assess species diversity, because they do not sample the entire community (Adis 1979, Topping and Sunderland 1992). The capture rate of spiders in pitfall traps is a function of both the activity level of the spider and its ability to escape from pitfall traps. In addition, *Linyphiidae* and *Lycosidae* species are often overrepresented in pitfall traps with respect to actual field density (Topping and Sunderland 1992). Nevertheless, pitfall traps allow a comparison of the relative abundance of different families between different sites and are therefore useful tools for indicating differences between communities at different sites. As pitfall traps are more easily standardized than some other sampling techniques they are especially useful in long-term studies, such as this one.

Climatic and remote sensing measures

Remotely sensed data were used to characterize vegetative biomass and ground temperature conditions at the pitfall trap sites during two different climatic regimes. To compare the climatic conditions between an El-Niño year (1997/1998) and an extremely dry year (1999/2000), we calculated the average monthly rainfall for the eight months before satellite multispectral data were acquired (October–May for each year). These data for the Phoenix area were available from the departments of Geography and Geological Sciences at Arizona State University. Landsat Thematic Mapper (TM) data were acquired on 24 May 1998 and Enhanced Thematic Mapper Plus (ETM+) data were acquired on 21 May 2000. The ETM+ is the successor instrument to the TM. It has similar band arrangements and wavelength coverage that allows direct comparison of data from both sensors at a ground resolution of 30 m/pixel (Parkinson and Greenstone 2000).

To quantify productivity in a given pixel, we calculated a normalized difference vegetation index (NDVI) for both the TM and ETM+ data sets. The NDVI calculates the relative percentage of actively photosynthesizing vegetation per pixel by rationing reflectance values in the visible red (low for plants) and near-infrared (high for plants) wavelengths (Botkin et al. 1984). The resulting values represent a spatial map of actively photosynthesizing vegetation and its abundance. Deserts typically have low NDVI values corresponding to low vegetation abundances and lack of leafy species, while forests have high values corresponding to high densities of leafy species (Turner et al. 2001). The NDVI values were calculated using atmospherically corrected reflectance data using the MODTRAN radioactive transfer code incorporated into the ATCOR software package (Richter 1999).

Differences in spider abundance between habitats may also be influenced by microclimatic conditions. Therefore, we calculated surface brightness temperatures (SBT) for the sample sites using the atmospherically corrected TM and ETM+ data. The SBT is the result of both radiant heating by the sun and stored energy released by emittance from surficial materials (Jensen 2000). As such, SBT provides a reasonable measure of the sensible heat a ground-dwelling organism would experience. The thermal band from each sensor (10.4–12.5 μm) was coregistered and resampled down to 30 m/pixel ground resolution to allow for comparison between the two data sets and vegetation index data. An average SBT was obtained from a 3×3 pixel region surrounding each sample site to minimize potential colocation error between the ground survey points and satellite pixels.

Statistical analysis

To describe gradients in spider distribution we applied canonical correspondence analysis (CCA) using CANOCO (ter Braak 1986). The data set included the total abundance of each spider family in each of the 26 sites. Environmental data included the six habitat types as dummy variables.

Due to logistical problems, several sites were not visited continuously on a monthly basis throughout the three years of study. Therefore, we pooled all time periods and calculated average spider abundance as the number of spiders per sample per site. We used ANOVA to assess the differences in diversity and abundance between sites and linear regression models to assess how site productivity and presence of nonspider predator affect spider abundance and diversity.

The differences in sampling effort among the 26 different sites could bias the number of spider families sampled (the random sample hypothesis sensu Connor and McCoy 1979). Rosenzweig (1995) suggested using diversity indices to control for sample size. Since different diversity indices have both advantages and disadvantages (Rosenzweig 1995) and may yield different results, we used three indices to control for sampling artifacts: (1) Simpson index (Simpson 1949) is particularly efficient in detecting dominance but may be affected by sample size, (2) Fisher's alpha (Fisher et al. 1943) does not detect dominance but is unbiased by sample size, and (3) rarefaction (Sanders 1968), a method that reduces all samples to the same level and yields comparable species accumulation curves. The analytic rarefaction program we used was developed by S.M. Holland and is available on the web.[1]

Results

Habitat patch size

The habitat differences in patch size are shown in Fig. 2. Examination of the area means for the six habitat types suggests a progression of fragmentation similar to that presented in Fig. 1 with one exception. Industrial sites were smaller patches than xeric yards. Total mean contiguous area decreases in the progression desert > agricultural > desert urban remnants > xeric yards > industrial > mesic yard. There is a large range of area values within these classes; this reflects the highly heterogeneous spatial patch structure of the Phoenix metropolitan region. Nevertheless, a general trend of fragmentation of spider habitat can be discerned using the mean patch area values.

Spider abundance and distribution

A total of 5574 spiders from 24 families were trapped between April 1998 and March 2001 in the 26 study sites (Appendix). In the two most mesic habitats 76.19% of the spiders were trapped: 3187 (57.09%) in agricultural sites, and 1066 (19.10%) in mesic yards. Spider abundance was also the highest in these habitats, with 21.62 individuals per site per month found in agricultural fields

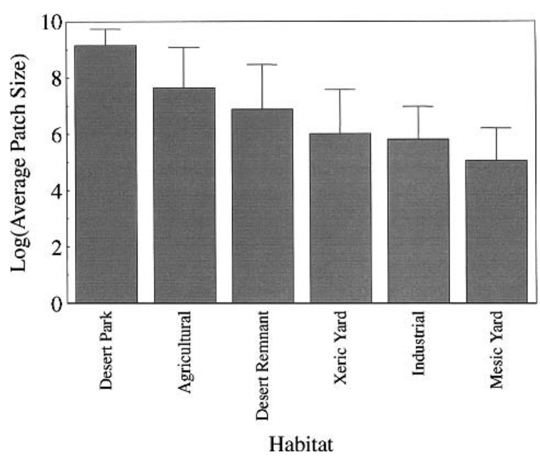

Fig. 2 Differences in patch size (in square millimeters) between six habitat types in the Phoenix area. Error bars represent +1 SD

[1] URL: <www.uga.edu/∼strata/software/AnRareReadme.html>

and 13.67 individuals found in mesic yards. The most abundant family was *Lycosidae* (wolf spiders) with 3280 individuals, representing 58.8% of the total, followed by Linyphiidae with 596 individuals (10.7% of the total). These two families were more abundant in agricultural fields and mesic yards than in the other four xeric habitats. Lycosids increased from 10–20% of the spider assemblages in the xeric habitats to 70–80% in agricultural fields and mesic yards. Linyphiids increased from 3–8% to 12–14% in the same habitats. The highest number of spider families (19) was found in desert remnants and the lowest (14) in xeric yards. Total spider abundance differed between habitats (ANOVA, $F = 4.387$, df $= 5{,}25$, $P = 0.007$). Abundance was the highest in agricultural fields, followed by mesic yards (Fig. 3).

The first axis of the CCA (Fig. 4) accounted for 26% of the spider family variance and separated desert parks and desert remnants from the other, human-managed habitats. This analysis suggests that spiders respond to land-use modification and changes in productivity. The second axis, accounting for 13% of the variance, separated agricultural, mesic yards, and desert sites from desert remnants, xeric yards, and industrial sites. Consequently, the spider community in xeric yards was similar to industrial sites, whereas the two most productive habitats, agricultural and mesic yards, had the most similar spider assemblages. The species–environment correlations were 0.88 on the first axis and 0.90 on the second axis. A Monte-Carlo simulation with 499 permutations indicated that species distribution along the axes was not random (first canonical axis, F ratio $= 4.546$, P value $= 0.002$; all canonical axes, F ratio $= 2.310$, P value $= 0.002$).

The eight most common families, making up 94.4% of all spiders collected, are marked bold in Fig. 1. The most common families are located around the origin (indicating that they were equally common in all habitats) or more associated with the mesic and agricultural sites, which were the most productive sites and where spider abundance was high. *Lycosidae*, which makes up >58% of the total abundance of the total trap, was one of the families most strongly associated with mesic and agricultural habitats, although they were also found in other habitats. The large tail of families that extends along the desert parks and remnants axis indicates that less common families were found at these sites. The tail also includes two commonly found families, Clubionidae and Pholcidae, indicating that these two families are strongly associated with desert parks and remnants.

Diversity

We used three indices to calculate diversity. All indices indicated that diversity was the lowest where spider abundance was the highest (i.e., in agricultural fields and mesic yards). These were Fisher's

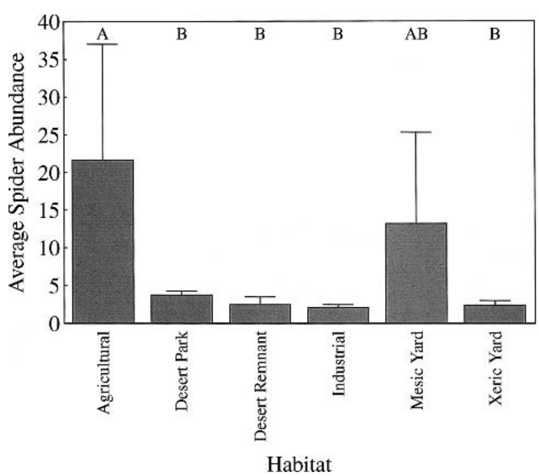

Fig. 3 Differences in spider abundance (spiders per sample) between six habitat types in the Phoenix area. Different letters above bars indicate significant differences ($P < 0.05$). Error bars represent +1 SD

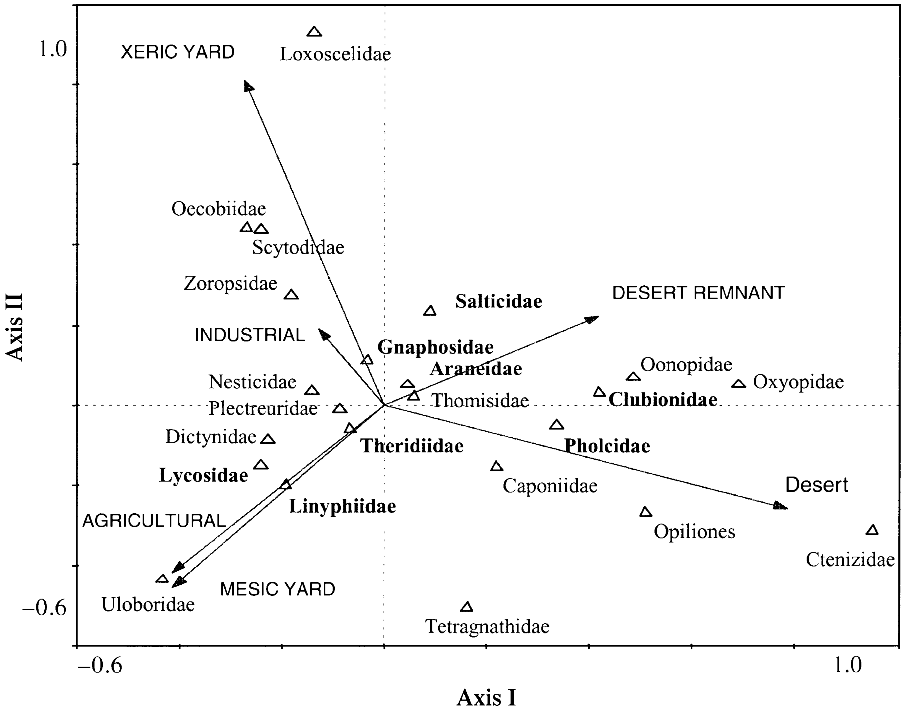

Fig. 4 Ordination diagram of the first two axes of canonical correspondence analysis for 23 spider families and six habitat types used as dummy variables. Arrows represent directions of greatest change in environmental variables. The eight most common families, making up 94.4% of all spiders collected, are boldface

alpha (ANOVA, $F = 3.64$, df $= 5{,}25$, $P = 0.017$), Simpson index (ANOVA, $F = 10.139$, df $= 5{,}25$, $P < 0.001$), and rarefaction curves (Fig. 5). Simpson index and Fisher's alpha were highly correlated across all sites (Pearson correlation, $r^2 = 0.71$, $P < 0.001$). Urban desert remnants showed the highest diversity according to both Fisher's alpha and rarefaction. Simpson index indicated a slightly lower diversity in this habitat than in desert parks, suggesting that spider assemblages in desert parks are more evenly distributed than in desert remnants (Fig. 5b). Diversity in xeric yards and industrial sites was moderate, but closer to desert remnants and desert parks than to agricultural sites and mesic yards.

Simpson index and rarefaction indicated that the lowest species diversity was in mesic yards. Though the rarefaction curve for this habitat appears similar to the agricultural habitat, it levels off earlier (Fig. 5a), whereas the agricultural habitat curve continues to increase moderately (Fig. 5b). Low Simpson index values in these two habitats (Fig. 5a) indicate highly dominant spider assemblages caused by the increase in the proportion of Lycosidae.

Spider diversity and habitat productivity

To test how habitat productivity influences spider diversity we used a multiple regression model in which habitat was a dummy variable. Of the six independent variables, (NDVI plus five dummy variables) only NDVI was significant ($r^2 = 0.63$, $P<0.001$ for NDVI, $F = 5.32$, $P = 0.002$ for the whole model). We then reran the analysis with the dummy variables only, extracted the residuals of spider diversity, and plotted them against NDVI as a sole independent variable. The results are

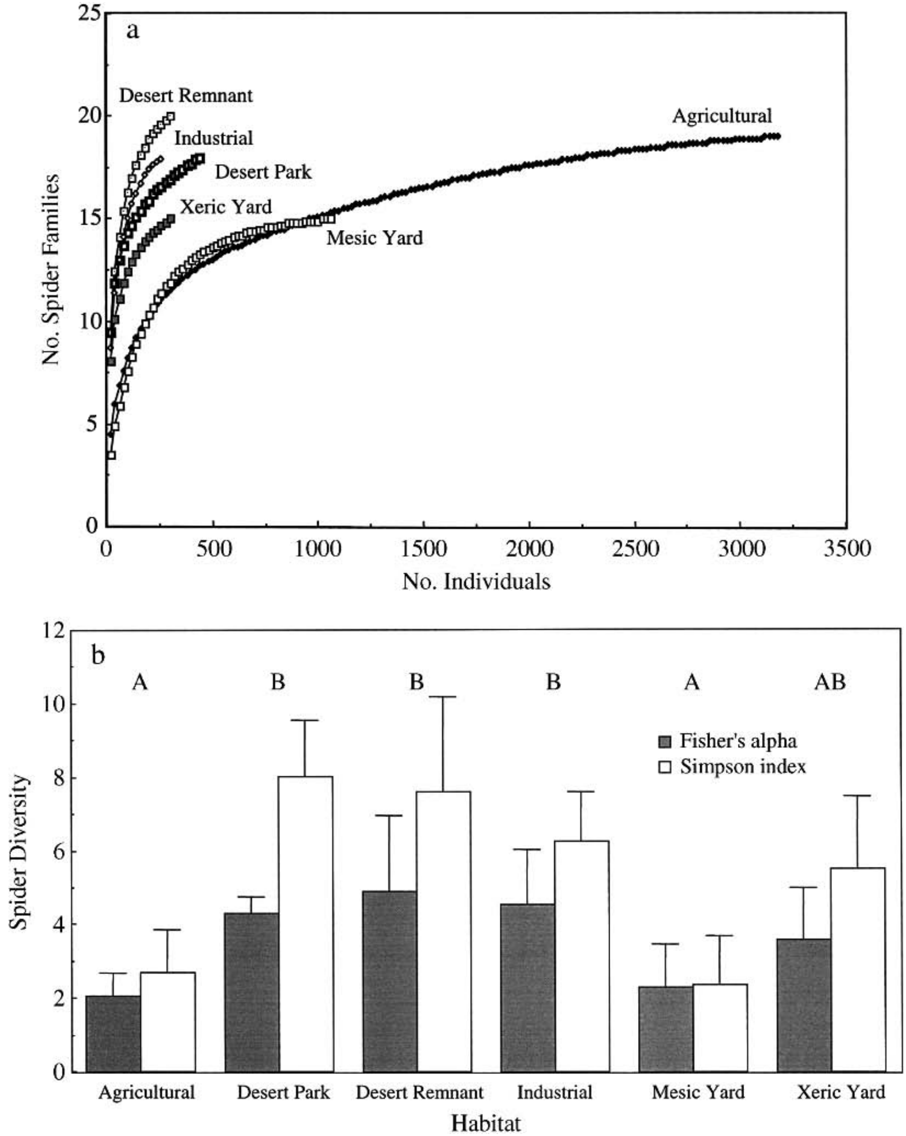

Fig. 5 Differences in spider diversity among six habitat types in the Phoenix area. (a) Rarefaction curves showing the expected number of spider families for any given number of individuals. (b) Average Simpson index and Fisher's alpha values. Different letters above bars indicate significant differences. Error bars represent +1 SD

shown in Fig. 6. To fit the nonlinear relationship between diversity and productivity we applied a locally weighted regression scatter plot smoothing (lowess; Neter et al. 1996).

The results indicate a decrease in spider diversity with habitat productivity. The NDVI data are taken from May 2000, an extremely dry year with a good contrast between desert and human-managed sites. The apparent decrease in spider diversity when NDVI was measured following an El-Niño year (May 1998) was not significant. Replacing family richness with diversity yielded similar results suggesting that the decrease in diversity was not due to a sampling artifact. The overall low fit is due to the high variance in richness/diversity between sites of low productivity. The few very productive sites (mostly agricultural and mesic yards) had lower diversity than most desert/xeric sites.

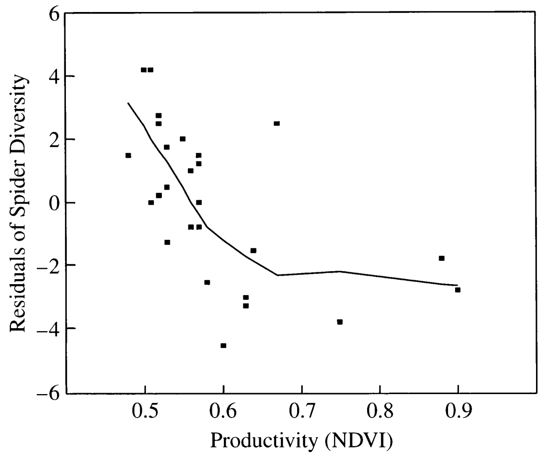

Fig. 6 The relationship between spider diversity and habitat productivity in the 26 study sites. NDVI refers to the normalized difference vegetation index

El-Niño effects on spider abundance

Compared with winter 1999/2000, the monthly average precipitation during winter 1997/1998 (October 1997–May 1998) was higher (Fig. 7a). Consequently, NDVI values measured in May 1998 were higher than those measured in May 2000 (Fig. 7b). The differences were not significant (paired t test, $t = 1.78$, df $= 25$, $P = 0.087$), probably because a few agricultural and mesic yard sites had higher values in 2000. Removing the mesic sites (where spiders were not sampled in 1998) from the analysis yielded significant differences (paired t test, $t = 3.440$, df $= 20$, $P = 0.003$).

To test whether the El-Niño event affected spider abundance, we compared spider abundance between May 1998 and May 2000. Average spider abundance (in all 14 sites where it was measured in both years) was five times higher in 1998 than that in 2000 (paired t test [log number of spiders], $t = 8.27$, df $= 13$, $P < 0.001$, Fig. 7c). We further tested whether this El-Niño effect occurred at the local (site) scale by regressing the “delta spider abundance” (the within-habitat differences in spider abundance) between years against the delta NDVI values, but found no correlation (linear regression, $r^2 = 0.023$, $P = 0.60$).

Spider abundance was lower in the drier spring (2000) in all five habitats (mesic yards were not sampled in 1998) than in the El-Niño spring of 1998 (Fig. 8). Abundance also decreased in the agricultural habitat (where it was the highest in both years), although irrigation is likely to compensate for the decrease in precipitation. However, whereas in May 2000 spider abundance in the agricultural habitat decreased to 71% of the abundance in May 1998, it decreased to 9% in desert parks, 10% in desert remnants, 35% in industrial, and 28% in xeric yards. In 1998, spider abundance in these habitats ranged between 13% and 62% of the abundance in the agricultural habitat. In 2000 it ranged between 5% and 8% of the abundance in agricultural fields.

In general, the trends in surface brightness temperature (SBT) variation were similar between 1998 and 2000, with the 2000 temperatures being consistently higher than 1998 (linear regression, $\text{SBT2000} = 12.57 + 0.85 \times \text{SBT1998}$, $r^2 = 0.65$, $P < 0.001$). This is in agreement with recorded precipitation and NDVI data; in both years SBT decreased with increasing NDVI, but this decrease was not significant in 1998, probably due to high vegetation density at most of the 26 sites. In contrast, the lower 2000 NDVI values at sites that are not actively managed and hence more susceptible to variations in climate were probably the cause of the sharp decrease in SBT with NDVI (linear regression, $\text{SBT} = 66.97 - 34.85 \times \text{NDVI}$, $r^2 = 0.83$, $P < 0.001$).

Total spider abundance was negatively correlated with SBT in both years (Fig. 9). We also found a negative correlation between SBT and spider abundance measured in the same months (May 1998

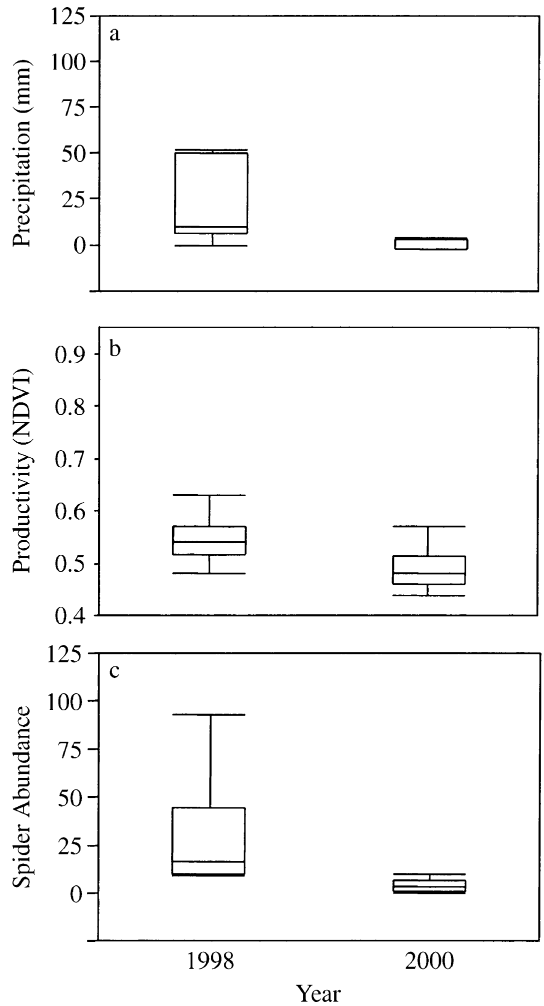

Fig. 7 Differences in (a) precipitation, (b) productivity (NDVI), and (c) spider abundance between May 1998 (following an El-Niño year) and May 2000 (following an extremely dry year). NDVI values and spider abundance are averaged across all but mesic habitats (where spiders were not sampled during the first year of study)

and May 2000), but this correlation was significant only in 2000 (spider abundance $= 95.30 - 1.82 \times$ SBT, $r^2 = 0.71$, $P < 0.001$).

Population level: bottom-up/top-down effects

We further tested for possible effects of prey abundance, predators, and habitat productivity on spider abundance. Here, we focused on the wolf spiders (*Lycosidae*). This family appears to be a key group, given its high abundance, especially in mesic yards and agricultural fields. We used a stepwise multiple regression model in which the dependent variable was wolf spider abundance, and the three independent variables were productivity (NDVI), prey abundance (Collembola, Diptera, and aphids, following Toft 1999), and competitor/predator abundance. This latter group included mantises (Order Mantodea), ant lions (Neuroptera), scorpions (Scorpiones), pseudo-scorpions (Pseudoscorpiones), and solpugids (Solifugae). These data were taken from the CAP LTER data set, collected simultaneously at the same sites. Individually, all three independent variables significantly correlated with wolf spider abundance. Wolf spider abundance increased with productivity and prey abundance

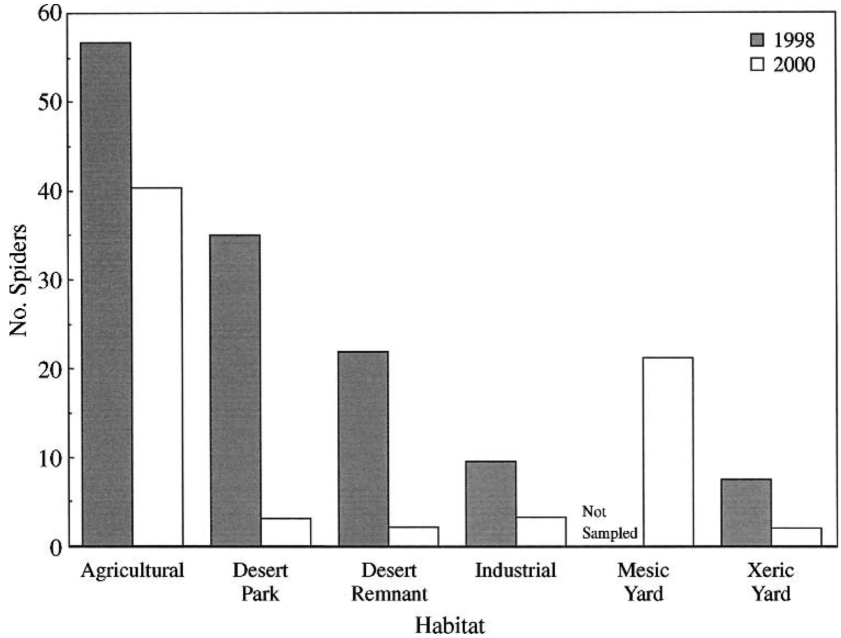

Fig. 8 Comparison of spider abundance between May 1998 and May 2000 within each habitat

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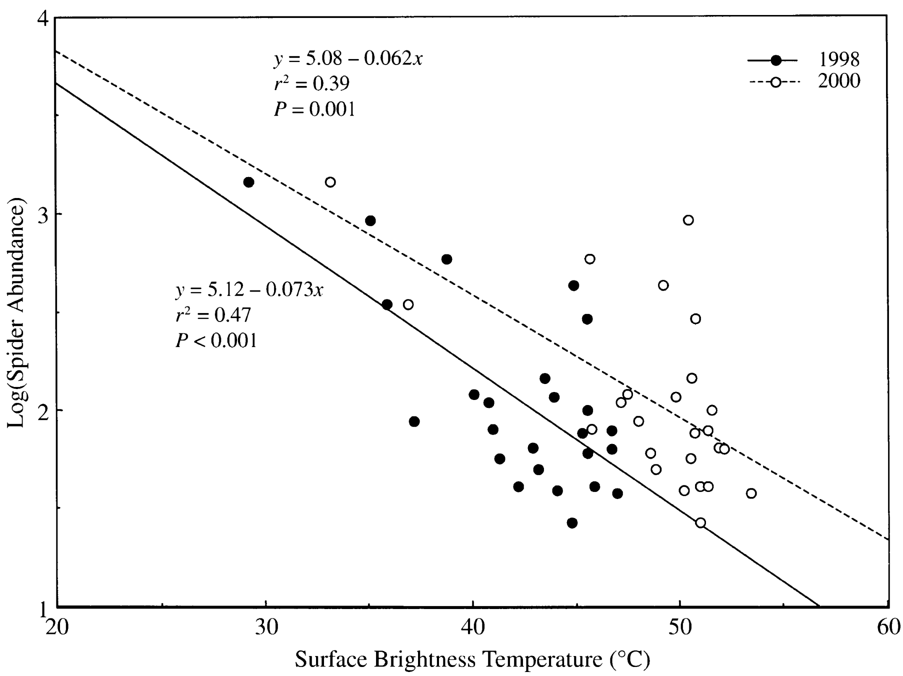

Fig. 9 The relationship between spider abundance and surface brightness temperature in the 26 study sites

Table 1 The effect of three environmental factors on the abundance of wolf spiders (*Lycosidae*).

A) Regression

Variable	Coefficient	F	P value	Accumulative R^2
Constant	0.014	0.019	0.985	
NDVI	3.089	2.485	0.021	0.237
Other predator arthropod	–0.478	–2.084	0.048	0.358
Prey (removed)	0.136	0.418	0.525	

B) Anova

Source	ss	df	Mean square	F ratio	P
Regression	6.654	2	3.323	6.422	0.006
Residual	11.900	23	0.517		

Notes: Stepwise multiple regression model included habitat productivity (normalized difference vegetation index [NDVI] values), prey abundance, and other predator arthropod abundance.

and decreased with other predator arthropod abundance. NDVI was the first variable to enter the regression equation followed by other predator arthropod abundance (Table 1). Prey abundance did not account for more variance and was removed from the model. Accounting for 35.8% of the total variance in wolf spider abundance (Table 1), the model's equation was:

$$\begin{aligned}&\log(\text{wolf spider})\\&\quad = 0.014 + 3.089(\text{NDVI})\\&\qquad - 0.478\ \log(\text{other predator arthropod}).\end{aligned}$$

Discussion

Anthropogenic effects, habitat management, and conservation of spiders have been thoroughly studied in agricultural ecosystems (reviewed by Bell et al. 2001). Much less is known about how urbanization influences spider communities. Land-use alteration by humans changes habitat structure and increases habitat fragmentation and site productivity (Gober et al. 1998, Lopez et al. 2002). The major response of spiders appears to be a general decrease in diversity (Fig. 5) and an increase in total abundance (Fig. 3). These processes are similar to the general pattern described for urban birds (Marzluff 2001). The increase in spider abundance is mostly due to a sharp increase in wolf spider (Lycosidae) abundance, and accordingly in their dominance. Wolf spiders thrive in mesic yards and agricultural fields. The dramatic increase in their proportion in these sites radically changes community composition and local diversity. Furthermore, the second most abundant family, Linyphiidae, is about twice as common in agricultural fields and mesic yards compared with the other four xeric habitats. Interestingly, these two families account for most species and individuals found in agricultural fields throughout central and northern Europe (Toft 1999). Furthermore, the similarities in community composition between the American southwest and Europe agree with the idea suggested by Blair (2001) that urbanization brings about the creation of homogeneous bird fauna. Our results suggest that this homogenization may also include other taxa and other human-managed habitats such as agricultural fields.

The high dominance of wolf spiders in productive habitats decreases spider diversity (Fig. 5). However, omitting this family from the analysis does not fully compensate for the decrease in diversity, and it remains the lowest in agricultural fields and mesic yards (data not shown). This suggests that increasing habitat productivity causes a general loss in diversity. Families that are adversely affected by desert land development are Clubionidae, Oonopidae, and Oxyopidae. Although uncommon in xeric habitats, these families are absent from highly productive habitats (Fig. 4; Appendix), suggesting that rarer families or species are most susceptible to losses when productivity increases.

The role of habitat productivity

Reviewing results of various studies on different taxa and provinces, Rosenzweig (1992) and Rosenzweig and Abramsky (1993), showed how biological diversity first increases, then decreases with environmental productivity. The diversity–productivity humped-shaped relationship is generally detected across very large spatial scales (e.g., species richness from the tropics to the temperate zone). However, urbanization can be viewed as a process that alters both habitat structure and productivity on local spatial scales.

Controlling for habitat type, our analysis indicates that the increase in productivity was the major cause for the reduction in spider diversity. The relationship between spider diversity and habitat productivity was not unimodal. Spider diversity decreased with productivity with a sharper rate in low productivity sites (Fig. 6). Possibly, the Sonoran desert already represents moderate productivity, and one should sample spiders in more arid deserts, such as the Mojave, for the increasing part of the curve. Our results differ from other studies on the response of arthropod diversity to productivity. Siemann (1998) found an increase in total arthropod diversity, as well as in the predator trophic level, in sites where productivity was experimentally enhanced. Kaspari et al. (2000) found an increase in ant diversity with productivity from deserts to rainforests. The mechanism responsible for the loss of spider diversity in highly productive sites in Phoenix is unclear. Several mechanisms may lead to the same pattern, but even under the same mechanism the results depend on which part of the productivity scale is being measured (Rosenzweig and Abramsky 1993). Siemann (1998) demonstrated how even by using an experimental approach it may be difficult to understand the complex relationship between diversity and productivity. For example, fertilization not only increases productivity, but also plant species diversity. Therefore, it can affect diversity both directly and indirectly. However, the decrease we found in spider diversity may suggest that some of the human-managed habitats are located extremely high on the productivity axis, and lead to low species diversity as predicted by Rosenzweig and Abramsky (1993).

The spider communities in industrial and xeric yards are located somewhere in the middle between desert and mesic sites (mesic yards and agricultural sites) (Fig. 4). As xeric yards contained mainly desert plants, we would have expected their communities to more closely resemble desert communities. However, owners of xeric yards in the Phoenix metropolitan area tend to water these habitats to produce rapid and sustained growth of plants (Martin and McDowell 1999). Thus, supplemental watering increases the productivity of the xeric habitat and spider and prey species may have responded accordingly.

Food abundance and competition

It is not clear how food abundance should affect spider abundance or distribution. Greenstone (1984) showed that spider diversity responded to vegetation structure more than to food abundance, though Bell et al. (2001) argued that changes in food availability affect prey abundance and that shortage of prey may influence growth rate and clutch size. In our study, productivity was the major factor accounting for wolf spider abundance (Table 1). Habitat productivity may confound other variables such as habitat structure or prey abundance—a variable that did enter the multivariate model of wolf spider abundance (Table 1). Competition or predation by other arthropods is not likely to affect wolf spider abundance. Although scorpions, solpugids, ant lions, and mantises were negatively associated with wolf spiders, their abundance accounted for only a minor portion of the variation in wolf spider abundance (Table 1). Vertebrate predators may have an even greater effect on wolf spiders. Foelix (1982) argued against the general overestimation of bird effect on spiders. However, the Phoenix area is very rich in lizards, which appear to be more common in natural, xeric habitats than in agricultural fields.

Since productivity increases in mesic yards and crop fields, the increase in wolf spider abundance may not necessarily involve out-competing other spiders. Indeed, in relatively productive sites local spider communities may be resilient to invasive spider species (Burger et al. 2001). In our study one widespread family, Araneidae, appears in agricultural fields at the same abundance as in other habitats. This family occupies a completely different niche to Lycosidae and therefore is unlikely to be in direct competition. Spider families that vanish or decrease in abundance in agricultural fields or mesic yards may be influenced by high disturbance rates. Mesic yards (mostly lawns) are often subjected to mowing, and agricultural fields are plowed, harvested, and subjected to a high turnover of crop types. Possibly, such disturbed habitats favor Lycosid and Linyphiid species. Lycosids are ground dwellers, and Linyphiidae are "wind dispersed" spiders that build small sheet webs close to the ground. A disturbed habitat where vegetation is close to the ground or removed (such as in a plowed field) may be highly suitable for them. This idea is strongly supported by Nentwig (1988) who found that Linyphiids predominated as pioneer species in intensively cut grass fields but become less abundant as dominance shifted from Linyphiids to Lycosids.

Climatic effects

The higher spider abundance in an El-Niño year (Fig. 7) suggests that extreme differences in annual rainfall between years may dramatically affect spider abundance. It is not clear how general this phenomenon is. McIntyre et al. (2001) found that both abundance and richness of the entire arthropod community correlated with air temperature, but not with rainfall. In contrast, Bolger et al. (2000) suggested that changes in rainfall in southern California might cause seasonal changes in arthropod diversity and abundance.

The lack of correlation between NDVI and spider abundance at the site scale suggest that spider population response to rainfall and changes in vegetation cover may not be obvious across small spatial or temporal scales. NDVI values represent snapshots of the result of a fairly long-term event (months) that affect an area much larger than the sampling points. The patterns described here suggest that spider abundance may show different responses to various biotic, abiotic, and anthropogenic factors, and across different scales of space and time. The decrease in spider abundance in sites where SBT is high is particularly interesting. De Keer et al. (1989) showed how changes in microclimatic conditions due to grassland management influence spider activity. Our results suggest that in addition to such behavioral changes, extreme alteration of microclimatic conditions can also affect spider abundance. The decrease in abundance with SBT may explain the high abundance in mesic yards and agricultural fields (Fig. 9). In addition to the correlation between higher SBT and lower productivity, it is possible that higher SBT directly affects spider diversity and abundance by decreasing the "comfort zone" for predators, leading to increased migration from areas of high SBT.

The response of spider abundance to the increase in productivity is also not straightforward. Obviously, El-Niño years increase local productivity throughout the environment, especially in desert sites. In May 1998 (El-Niño year), spider abundance was higher than in May 2000 (a particularly dry year) in all habitats (Fig. 7). Interestingly, it was higher by $\sim 50\%$ even in the agricultural habitat. This may reflect the much higher total spider abundance in the environment and the presence of many "floater" spiders in agricultural fields. It may also explain why there is no correlation between

NDVI values and spider abundance across all sites. After hatching in spring, and especially in high densities as in El-Niño years, spiders may move across different habitats searching for good quality patches to establish and build webs. During such periods the relationship between spider abundance and habitat structure or quality may not be at equilibrium. Yet, one important pattern that emerges (Fig. 8) suggests that in El-Niño years, the productivity of natural habitats, even deserts, increases to a level similar to that of human-managed habitats. Therefore, a possible effect of agricultural development and urbanization is the cancellation of seasonality.

Management and conservation

The results indicate that a moderate fragmentation of Sonoran desert into urban desert remnants where fairly large desert fragments may exist does not reduce spider diversity. Spider diversity in desert remnants may even exceed diversity in desert parks. However, further desert fragmentation into xeric urban yards decreases diversity. The reasons for this decrease are not totally clear, and correlative studies may not be sufficient for revealing mechanisms. Miyashita et al. (1998), Bolger et al. (2000), and Gibb and Hochuli (2002) all suggested complex mechanisms for the effect of habitat fragmentation on spider abundance. For example, as spiders represent a high trophic level, changes in food abundance due to fragmentation may cause local spider extinction in small habitat fragments (Miyashita et al. 1998).

Spider diversity may, therefore, peak at intermediate levels of urbanization. A similar pattern has been described for butterflies along an urban gradient (Blair and Launer 1997). However, any further development that changes the habitat structure of the once-natural habitat fragments decreases spider diversity. In the Phoenix area, xeric yards support a higher diversity than mesic yards, which represent the same level of fragmentation plus more radical habitat alteration. These results are similar to what McIntyre and Hostetler (2001) described for bees (Apoidea) from the same area. These results suggest that although negative effects of urbanization on arthropod diversity may be mediated through both habitat fragmentation and alteration, habitat structure alteration has a greater effect on arthropod diversity. A comparison of agricultural field and mesic yards further supports the idea that land use alteration has a greater effect on spider communities than the reduction of area per se. In terms of water availability and habitat productivity, agricultural sites appear as much larger versions of mesic yards. In this case, area does not compensate for the desert habitat alteration, as spider diversity in crop fields is still as low as in mesic yards.

Since other studies on urbanization effects on spiders focused on habitat fragmentation, it is hard to draw conclusions about the partial effects of fragmentation vs. habitat structure alteration. Gibb and Hochuli (2002) found that spider diversity was not affected by forest patch size fragmented by urbanization, though community composition did change. Bolger et al. (2000) described different results for different sampling methods and seasons. Spider diversity in scrub patches decreased with fragmentation, although abundance increased. As recommended by Gibb and Hochuli (2002), future studies should incorporate effects of fragmentation and habitat structure/land use alteration to better understand the complex effects of urbanization on arthropod communities.

In central Arizona, future landscape planning should (1) favor xeric over mesic yards, as recommended by McIntyre and Hostetler (2001) concerning pollinator communities, and (2) incorporate large remnants of natural habitats within the urban core, since such remnants appear to retain rich arthropod communities.

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Appendix

A table showing spider families and the number of individuals sampled in six habitat types in the greater Phoenix area between April 1998 and March 2000 is available in ESA's Electronic Data Archive: *Ecological Archives* A014-003-A1.

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Section V
The Anthroposphere: Human Dimensions

The nine selections that comprise *The Anthroposphere: Human Dimensions* bring into focus several of the key human components of Urban Ecology. Imbedded in all of these writings are four cornerstone concepts of urban ecology – drivers, patterns, processes and effects. Whereas the previous sections have concentrated more on the ecological aspects of this urban-ecological framework, these authors explore more of the social and political terrain of Urban Ecology. In the process human attitudes, values, and behaviors regarding the environment, often in cross-cultural contexts, are engaged and valuable insights are revealed. Both the excitement of discovery and arduous nature of pursuing transdisciplinary research are well illustrated and documented in these selections. Woven through most of these articles are clear examples of the social contexts, and hence social constructions, of many concepts used in Urban Ecology. Scale is a powerful theme that permeates all of these papers. On the human scale it ranges from consideration of the values, beliefs and behaviorally revealed preferences of an individual human to those of collective humanity. On the geographical scale, it ranges from individual urban land parcels and apartment dwellings to the entire globe. Most importantly a common thread links all of these works, with nodes of intense energy and material flows and transformations that we call cities.

Harrison & Burgess (2003) explore key social science concepts and frameworks used to understand how urban residents form their own values, attitudes and behaviors toward their surrounding environment. They provide a clear contrast between traditional reductionist versus contextualist/social theoretical approaches to sustainable development. They argue for the importance of environmental communication and how it is mediated by and contingent upon pervasive social, political and cultural factors. Interaction, development of mutual respect and trust thus become critical components for the creation of sustainable development strategies.

Hall (1993) continues this critical look at traditional approaches by contrasting the basic tenets of neoclassical economics applied to the modeling and explaining of human economic behavior regarding the environment with the concepts from environmental and natural resource economics. She clearly challenges the appropriateness of such dominate social constructions as "optimization," "value," "rationality," "marginalism," "impersonal market transactions," and the inevitability of "continuing and unlimited growth." She argues for the stronger, more explicit, imbedding of the concepts of "externalities," "public goods," "un-priced and under-priced goods," and, "scarcity" *per se.* While urging the incorporations of these latter concepts into ecological-economic analysis, Hall recognizes the difficulties of doing so while acknowledging explicitly the absolute necessity for integrated transdisciplinary research team efforts.

Waddell & Moore (2001) provide a framework for wide ranging thinking about the factors that generate demand for urban land. Especially important is their inclusion of the interactions among of the factors of urban land demand, land supply and various governmental policy interventions and their effects. They very logically and systematically elaborate increasingly complex and realistic models and techniques for analyzing markets and demand for urban land. At each stage of model development they are clear about the data requirements and limitations. Scale and units for analysis are treated very carefully. They present two analytical case studies and briefly introduce two simulation models – "MetroScope" and "UrbanSim". The authors end on a telling note about the difficulty of curbing "urban sprawl" in the U.S. and of articulating effective growth management policies that do not clash with principles of individual private property rights and local control. Again, the need for proactive environmental communication and stakeholder participation jumps out.

Ewing (1994) delves into the entire complexity of urban sprawl and its myriad patterns and the difficulty of defining sprawl operationally for the purposes of rule-making or policy-making. Having provided all of the nuances and caveats, Ewing goes on to identify the supposed costs of sprawl, such as, psychic costs, excess travel and congestion, energy inefficiency, air and water pollution, inflated infrastructure and public service costs, and loss of farmland. Most readers will find at least a few surprises here to challenge their thinking about urban sprawl.

In a way Rees & Wackernagel's (1996) classic article on urban ecological footprints can be interpreted as the global scale version of urban sprawl. Their simple yet powerful concept of ecological footprint has received wide international recognition. It remains a core part of the international discourse on sustainability from the local to the global. Simply put an urban ecological footprint is a measure of how large an area of productive land (and marine environment) is required to sustain a defined population indefinitely, regardless of where that land might be. Rees & Wackernagel provide a few calculated examples of their ecological footprint concept while acknowledging both the concept's strengths and weaknesses. They do not shy away from discussion of global equity concerns about unsustainable prodigious consumption by the "advanced" highly urbanized and rich industrial North contrasted by the under consumption of the poor South. Well worth reading is their section on "Ecological Pros and Cons of Cities." Here the reader will discover the true meaning and intent of their subtitle – "Why Cities Cannot Be Sustainable – and Why They Are a Key to Sustainability."

Stephen and Rachel Kaplan have long been at the forefront of research in environmental cognition and environmental perception. In this 2003 essay they invoke the concept of "the reasonable person" linking environmental factors with human behavior. They argue that people are more reasonable when the environment supports their needs for information of three sorts: the need for exploration and understanding, the need for meaningful action, and the need for restoration.

In rigorous analytical fashion Ewing et al. (2003) employ a cross-sectional hierarchical modeling approach to relate the characteristics of places and individuals to levels of physical activity, obesity, body mass index (BMI), hypertension, diabetes and coronary heart disease. This paper represents an excellent example of the integrated urban ecology research. Ewing and his colleagues ask bold and important transdisciplinary urban ecology questions. They work carefully and methodically with their individual characteristics, demographic, behavioral, health, and urban form data. Although small in magnitude, the data speak for themselves supporting ". . . the hypothesis that urban form affects health and health-related behaviors."

Expanding the theme of urban ecology and health, Kraas (2003) reviews how our largest cities ("megacities") act as victims and producers of risks. Concentrating people increases the risk of natural disaster to large cities, but human-made social, political, economic, and health risks also abound in cities. The increased risk of megacities and their typical siting within poor nations raises the question of environmental justice. Why? Because in effect the rich nations have been outsourcing not just their industrial production to these very nations, but along with it their air and water pollution. We have been externalizing a significant portion of our negative urban ecological footprint through international trade. In the final selection Bryant & Callewaert (2003) tackle the questions of urban ecosystems and environmental justice directly. Perhaps their strongest statement is that the cultural construct of environmental justice challenges the absolute authority of the market system by unequivocally identifying the interconnections amongst environmental quality, social justice, and civil rights. In the face of market failures to honestly reflect social-ecological-economic conditions, they offer three strategies to strengthen and make apparent these interconnections: 1) promote community-based research initiatives, 2) incorporate environmental justice concerns within a sustainable knowledge construct/database of urban ecosystems, and 3) support the formation of a new type of individual. Their last strategy here brings us full circle back to our recognition of the need for new forms of transdisciplinary graduate training and research in Urban Ecology.

Social Science Concepts and Frameworks for Understanding Urban Ecosystems

Carolyn Harrison and Jacquie Burgess

Keywords: sustainable development · Holland · environmental values · social sciences

Introduction

Cities are a potent demonstration of humanity's domination of nature; they are also the source of a wide range of environmental problems that enmesh city residents in a process of globalisation capable of touching even the most remote and rural of communities. In the context of the agreement reached at the International Summit at Rio in 1992 that all nations should move in the direction of sustainable development, cities also have a critical role to play in determining the rate and nature of that change. For example, were city residents to adopt more pro-environmental lifestyles, then considerable progress would be made towards achieving sustainable development. Against this background, education and communication strategies which seek to promote understanding of the linkages between how people live their lives and the quality of our environment have a potentially important role to play in moving society in the direction of sustainable development.

The purpose of this chapter is to explore some of the concepts and frameworks social scientists use to understand how city residents make sense of their own attitudes, values, and behaviors toward the environment. It does this first by drawing on recent research in the social sciences that support contextualist approaches to society, and second by using the findings of a cross-cultural study undertaken in two European cities: Nottingham in the United Kingdom and Eindhoven in the Netherlands. This study was designed to compare how local residents and decision makers in each city discuss their responsibilities and behaviors toward the environment. By offering a cross-cultural comparison, the study serves to highlight the role that social, political, and cultural factors play in influencing people's willingness or reluctance to adopt more pro-environmental behaviors. It also serves to demonstrate how education strategies designed to promote public understanding of urban ecosystems can be informed by arguments individuals employ to challenge exhortations by governments and other agencies for citizens to "go for green."

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Models of Sustainable Development

Many holistic models of sustainable development seek to emphasize links between society, economy, and environment in the manner of a natural system. Perhaps the most significant contribution these models have made is the extent to which they have encouraged decision makers and planners to identify and take account of the full costs to the environment of unsustainable development. Through the work of environmental economists in particular attempts have been made to assign costs to the losses and benefits of previously taken-for-granted environmental goods and services (see Costanza et al., 1997); however, the social, political, and cultural constraints that prevent environmentally sustainable development from taking place have not been elucidated so clearly (Benton and Redclift 1994). Sustainable development is often framed as an *environmental* problem that can be solved by a scientific approach, thereby excluding (whether deliberately or not) debate about the wider sustainable development issues such as the North–South divide, social inequalities, debt burden, and the endless pursuit of consumption (Wynne 1994).

It is important to understand cities as natural systems and to adopt lifestyles consistent with prudent use of resources, such as decreasing dependency on the car, insulating buildings, and recycling and reclaiming materials. There is no guarantee, however, that individuals or institutions will respond to this logic. Because natural systems have no moral authority and environmental science claims about urban ecosystems are formed and transformed through a range of cultural, social, and political processes, strategies for environmental education and communication need to be informed by a range of intellectual and practical approaches. For example, exhorting the public to adopt more pro-environmental lifestyles involves issues of rights and responsibilities, and raises questions about the role that structures and norms in society play in governing how people engage with these concerns. Recent social models of sustainable development point to a range of approaches that can inform public education strategies about urban ecosystems and promoting pro-environmental lifestyles (Burgess et al., 1999; O'Riordan and Voisey 1997).

Social Models of Sustainable Development

Throughout the last 30 years, new "contextualist" theories of society emerged in social sciences (e.g., psychology, anthropology, sociology, geography, and planning) (Giddens 1991; Giddens and Lash 1994). These contextualist theories have emerged in part to challenge the more traditional reductionist approaches in social science that posit society as an aggregation of individuals who behave rationally (i.e., in their own self-interest). In contrast, contemporary social theory sees individuals as social beings whose actions reflect their socially derived meanings, values, and knowledges. One of the leading theorists is Anthony Giddens, a British sociologist who has done much to explain how individual identity is an integral feature of the social structures that both shape, and are shaped by, individual actions (Giddens 1991). Contextualist theories suggest that how an individual behaves cannot be predicted as a logical outcome of cognitive processes alone. Instead behavior is seen as a more complex, reflexive process of active engagement that is contingent on many factors and circumstances. For example, what we might choose to do is contingent on people's experience with the past and with place, and also on the role structures and norms play in shaping behavior. "Structures" include institutions such as commerce, education, health care, and planning together with their rules and modes of organization, which literally structure social, economic, and political life. Formal and informal rules and regulations ensure that each society has "norms" and functions in a "proper" way. Viewed from a contextualist perspective, actions are interpreted as responses to feelings of emotional attachment and duty, questions of trust and authority, and to a sense of believing (or not) that individual actions can influence change. Given their emphasis on understanding behavior "in context," such approaches favor qualitative research methods where people are engaged in discussion, rather than experimental and questionnaire-based approaches characteristic of traditional social science.

Against the background of the present topic—understanding urban ecosystems—reductionist and contextualist approaches provide rather different perspectives on how scientific information about city environments is understood and acted on. For example, mass media campaigns designed to promote pro-environmental behavior tend to work with a reductionist cause-and-effect model in which the mind of each individual needs to be filled with new and “correct” information that will engender appropriate behavioral responses. On the other hand, contextualist models challenge this “stimulus-response” model by arguing that individuals engage critically with new information. In particular, information is always understood in the context of the social and cultural relations within which it is embedded. People already have well-developed ideas and opinions which are used reflexively to “interrogate” the authority, credibility, and legitimacy of new information. For example, several studies suggest that the social and cultural status of institutions has an important bearing on the extent to which the public trusts information (Wynne 1994; Irwin 1995). In the same way, questions of trust, authority, and legitimacy all influence public reception of communications seeking to promote an understanding of cities as ecosystems.

In this short chapter the reductionist and contextualist theories can only be treated schematically as “ideal types.” This is what Fig. 1 illustrates. Reductionist models anticipate that people will respond “rationally” to choices once successfully communicated; contextualist models suggest that any response is conteigent on whether these choices have authority and credibility in terms of social and cultural identification (or alienation) and not through any “assumed” or natural warrant. Scientific findings may not achieve authority with the public because the reception of information is shaped by a range of social, cultural, and political processes that change over time. Reductionist models construct individuals as “rational consumers” acting on their preferences, responding to market forces, and seeking to maximize their own self-interest; whereas contextualist theories construct them more as “ethical citizens” (see Fig. 1). In the case of the ethical citizen, normative judgments figure prominently in decisions, especially when these decisions impact on communal resources such as the environment and the public domain of streets, parks, and plazas.

These two frameworks can also help illustrate different conceptions of how individuals engage with the political processes that determine the rules and norms of society. Reductionist models favor a dominant role for individual preferences as expressed through the market, for “experts” and “pro-

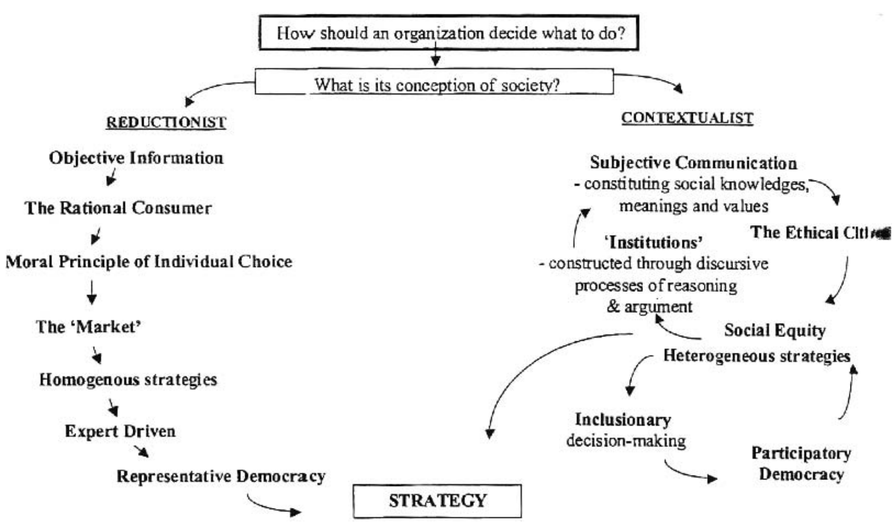

Fig. 1 Approaches to sustainable development

fessionals” in decision making processes and are consistent with forms of “representative” democracy that perpetuate existing rights and power relations (Forester 1993). Contextualist approaches on the other hand favor a stakeholder approach in which anyone who has an interest in the outcome of decisions has a right to be involved. Consistent with a shift toward more equity in the allocation of rights and responsibilities, contextualist perspectives also favor more participative forms of democracy in which a wider range of knowledge is respected and given credence (see Bryant and Callewaert, Chapter 3 in this volume; Forester, 1989; O’Hara 1996; Irwin 1995). Integral to this process of greater participation is the reconstitution of social relations through a process of mutual learning and understanding. In this reflexive process individuals, structures, and norms may be redefined and reconstituted (represented by the feedback loop in Fig. 1). The *transformative potential* of these deliberative and inclusionary processes of decision making contrasts with the reinforcement of existing power relations maintained by conventional, top-down and expert-driven processes of decision making (Fishkin 1991; Innes 1996; Healey 1997). In turn, communication strategies which seek to promote *new* understandings about the environment and society’s relationship with it must provide opportunities for open and fair debate that can question existing understandings and social norms.

These ideal type alternatives serve to illustrate how new social concepts provide frameworks for thinking about sustainable development and cities as ecosystems. In what follows, elements of both are used critically to examine how more pro-environmental behavior can be encouraged among city residents.

A Cross-Cultural Study of Urban Residents’ Commitments to Pro-Environmental Actions

The two-year study of residents’ and decision-makers’ attitudes to lifestyle changes required by global environmental change was undertaken in Nottingham (U.K.) and Eindhoven (the Netherlands) between 1993 and 1995. Both are medium-sized cities with a population of 274,000 and 195,000, respectively. Neither city had progressed environmental initiatives very far, although local authorities in both cities were sympathetic to developing integrated transport systems, recycling, and reclamation schemes. Nationally, the central government in the Netherlands had taken a more proactive approach to environmental planning than the U.K. government. Two National Environment Plans published in 1989 and 1993 set targets for all sectors of society to meet, and since 1990 the Dutch government has sponsored a mass media campaign to raise public awareness of how individual behavior could make a difference to global environmental problems. In the United Kingdom there were no such national plans, no sustained media campaign was undertaken, and the dominant approach gave priority to the operation of the market as the primary definer of both what were environmental problems and what their solutions might be. In the light of these national contexts, the overall purpose of the study was to determine whether citizens in Nottingham felt more or less empowered to assume responsibility and undertake pro-environmental behavior than citizens in Eindhoven, and if so to account for these differences.

Phase one of the study involved a questionnaire survey of 250 respondents in each city. The sample was generated randomly and the survey was conducted in comparable, suburban neighborhoods. Phase two involved conducting two in-depth discussion groups in each city—one with men and the other with women. The eight to ten participants in each group were recruited through the household survey and included a cross-section of the community as defined by age, income, and education. The groups met for 1.5 hours on each of five consecutive weeks. The household survey attempted to measure individual responses to questions about environmental awareness, attitudes, and behaviors, whereas the in-depth groups engaged discursively with a small number of people and gave participants time and opportunity to deliberate on the issues raised in the survey. The final stage of the reseach was to conduct a workshop with policy makers in each city to discuss the implications of the research for their environmental communication strategies (Burgess et al., 1998). The findings of the household questionnaire and the in-depth groups are drawn on here to provide an understanding of how people rationalized their own environmental responsibilities (see Harrison et al., 1996).

Anglo-Dutch Comparisons: Contrasts in Pro-Environmental Practices

First, we will briefly discuss the findings of the household survey as they relate to people's lifestyles and respondents' willingness to adopt more pro-environmental behavior. We will then move on to report on the findings of the in-depth discussion groups and foucs on the reasons participants use to resist calls on them to "go for green."

One of the most intractable environmental issues facing cities is the demonstrable need to reduce traffic and to increase independent mobility without relying on the motorcar. In both cities local authorites had attempted to promote a number of measures designed to reduce car dependency, including car-sharing, promoting public transport, designating high-occupancy vehicle lanes on commuter routes and providing cycle routes. Overall, people in Nottingham exhibited a much higher dependency on the car than in Eindhoven. Car ownership was slightly higher in Nottingham (77 percent) than in Eindhoven (74 percent) but 69 percent of car owners in Nottingham reported using their cars 5 days a week or more compared with 41 percent of car owners in Eindhoven. In addition, there was a greater reluctance to change transport behavior in Nottingham. When asked if they had changed their transport behaviour in the last 5 years, only 37 percent of Nottingham respondents said that they had, compared with 60 percent in Eindhoven. Of this latter group, 35 percent said they now used their car less often compared with only 17 percent of the former. Alternative transport modes used most frequently involved walking and cycling in Eindhoven and using public transport in Nottingham. Only in Eindhoven did people mention that they had changed their behavior "for the sake of the environment" (13 percent). On this evidence, although the majority of people in both cities depended on the car, more people in Eindhoven reported that they had changed their behavior in favour of less-polluting transport modes, and for some people these changes had been made for "environmental" reasons.

When it came to addressing the wider issues raised by sustainable development, such as the need to reduce consumption and use natural resources in more prudent ways, a similar picture emerges. The level of pro-environmental behavior was much higher in Eindhoven than in Nottingham. For example, people purchased more green products, recycled more materials and shared these tasks among members of the household. In this sense the overall commitment to recycling in Eindhoven was much higher than in Nottingham, but it was not clear whether this pro-environmental behavior had become a matter of routine, signifying a change in lifestyle, or whether commitment was more pragmatic and ephemeral.

Analysis showed that respondents who were most "environmentally active" (excluding car use) lived in households that are better educated than average, had higher incomes, and held managerial and professional jobs, although members of all social classes participated in pro-environmental behavior. This is consistent with the findings of other surveys that suggest a marked shift in environmental behavior since the early 1980s (Witherspoon 1994). Certainly residents of Eindhoven seemed more environmentally committed than residents of Nottingham. Whether this was the result of access to more information associated with the mass media campaign, access to more recycling facilities, or a greater predisposition to a "collective" approach to solving problems could not be determined from the household survey. Detailed statistical analysis revealed very little consistency between pro-environmental behavior and gender, education, class, voting intention, or how active in the local community people report themselves to be. In other words no simple and coherent "green" view about how to address environmental problems existed among these city residents, and pro-environmental behavior could not be predicted with any confidence from recorded variables.

One of the main purposes of the in-depth discussion groups was to explore the apparent ambiguities raised by the analysis of the questionnaires and to allow us to work with qualitative methods of inquiry that are more sensitive to contextualist accounts of society.

Resisting Calls to "Go for Green": Findings of the In-Depth Groups

The four groups (nine men and nine women in Nottingham and 10 men and 10 women in Eindhoven) met for 1.5 hours each week over 5 weeks. The groups followed a similar agenda. Topics included green consumerism, the impacts of technological and social changes on people's daily lives, their experiences of environmental changes, and ideas about sustainability. Through the discussions it became clear that assuming responsibility for addressing problems associated with global environmental change was a complex concept that involved a number of real and sometimes intangible constraints and benefits. Running through all the discussions was a powerful moral or normative dimension about what people *ought* to be doing, not only for the sake of the environment but also for the sake of society. For some people this sense of commitment came from deep personal conviction and was expressed with emotional force. Other people had a much weaker emotional commitment but wanted to engage altruistically in contributing to the collective good. Being able to exercise choice in what to buy and having the time to recycle was also important; however, people were concerned, too, about whether or not their actions were effective in achieving the goals they espoused and whether or not they could believe all the information they received about environmental problems and solutions.

The Role of Information

In both cities, the media played a particularly prominent role in discussions about these wider political and social concerns, in particular through their reporting of environmental issues, which severed to expose the "contingent" nature of environmental "truths." In both Eindhoven groups, members felt overburdened by information that was often contradictory. In Nottingham, too, "media food scares" for example provoked a real sense of confusion for both men and women. Wanting the best for their families but being dependent on expert advice, and coping with the conflicting claims of different interests as represented through media reports, left everyone feeling very angry and confused. John felt very strongly about this: "We talked last week about aerosols. Why didn't they just ban them straight away if they're dangerous? And if they're not dangerous why scare us? I've actually lost confidence in, um, supposed "experts" on environmental issues. Because . . . then you get politicians coming in and they don't tell you the truth. . .. Suppose I had asked your advice about food, what food I should eat, or whether an aerosol is dangerous. I'd want to know the credibility, that . . . where you're coming from? What experience have you got, er, to make an opinion?" The men struggled to come to terms with their belief that experts such as scientists, politicians, and people in the media couldn't be trusted and how this affected their ability to make justifiable decisions. They all agreed with John when he said: "I live in a period of confusion."

The Dutch men talked about their response to the Dutch government's media campaign. This campaign used an image of a burning globe held in a hand that was accompanied by a message exhorting people to "act locally, think globally." One of the men said: "One person cannot blow out the candle to save the world—it's much more complex than that!" The Dutch women were equally cynical about the media as a purveyor of trustworthy information. One of them said: "Fifty percent cannot be believed, but it's difficult because I don't know which half!" In these circumstances "following your own instinct didn't help either, because there comes a point where that's very difficult

if you are getting so much false or biased information. Often they say something in the morning and in the evening it's retracted."

Given this pervasive conviction that experts could not be trusted and a belief in the contingent nature of "truth" about environmental issues, it is not surprising to learn that people were unwilling to accept personal responsibility for the environment. This was not the passive response of in uncaring or ignorant public but rather an active resistance. It represented their own attempts to separate out environmental problems from the complex web of social, economic, political, and cultural practices people understood these problems to be embedded within. Much as Eden (1995) suggests, having a well-developed sense of "actionable responsibility" enables some committed environmentalists to adopt pro-environmental behavior even in the face of conflicting media reports of the efficacy of particular actions. Many more people when faced with the mixed messages promoted by the mass media feel impotent and do not know what to do for the best. In these circumstances they look to "others" to take the lead.

The Social Contract Between State and Its Citizens

Overall, what was impressive about the in-depth discussions held in the two cities was the extent to which the tenor of discussion in Eindhoven was much more optimistic and positive than in either of the two Nottingham groups. For example, with respect to recycling schemes, participants in Eindhoven linked recycling and reclamation of materials to possible improvements in the local economy. Such schemes were regarded as an industry requiring considerable investment but also as a source of potential new employment; however, although the Dutch groups looked to the national and local government to take these initiatives forward, they also believed that as individuals they had a social obligation to participate in the scheme. Introduction of a compulsory scheme locally had reinforced individual responsibility because as one man put it: "You can't hide from your responsibility at the local level." Organizing a compulsory local scheme seemed a good way of making abstract global problems "real." Although members of the Dutch men's group agreed when Jan said that governments "always promise more than they deliver," they also accepted his metaphor of environmental progress as the moves of a knight in a chess game. As Jan expressed it: "If the worst comes to the worst you go forward one and back two. But we must go on all out and try to keep going through it."

This positive attitude and willingness to accept some measure of self-ascribed responsibility for pro-environmental action especially when national and local governments had taken the lead, contrasted with the seemingly more defeatist attitude that pervaded discussions in the Nottingham groups. Some members of the Nottingham groups had attempted to organize a recycling program in their local school, only to see it fail through lack of effort and the vagaries of the wastepaper market. Others had also tried to make use of whatever local facilities were provided, even though they were poorly run and serviced. These frustrating individual experiences led to a complex and often furious debate about where the responsibility for changing attitudes and practices resided. Was it with individuals, with government, or with commerce? In these discussions there was more disagreement about the nature of individual responsibility than in the Dutch groups, but there was a clear consensus that in the United Kingdom neither national nor local governments were setting an example for people to follow. The imposition of Value Added Tax on domestic fuel in 1993 for example, was interpreted in both Nottingham groups as a means of raising revenue dressed up as an environmental measure; as one mand said, "That's how greedy this government is. It's not green. It points the way down the green road but doesn't go down it!"

Trying to separate issues of individual responsibility for the environment from broader changes in social values was difficult because these broader changes seemed to inhibit any real shift toward the kind of altruistic behavior that was required. For some of the Nottingham women it seemed that "People now are just so greedy and selfish. . .. It's like our country is selfish, we don't want to stop our people driving cars and stopping acid rain, because we want to drive our cars and we've got a right to do it. You know, we don't seem to have a moral conscience." Others felt that the free-market, individualistic ideology pursued by the national government was more to blame. Overall, however, they agreed that the absence of both personal and national commitment to the shared responsibility meant that there was no basis upon which a social contract between individuals and their neighbors and between the U.K. government and its citizens could be built. Under these circumstances the prospects for achieving sustainable development in the United Kingdom seemed more remote than in the Netherlands.

In Nottingham, people's frustration, alienation, anger, and in some cases despair, were all implicated in explanations for the contrast between the high level of environmental awareness reported in the questionnaire survey and the lower levels of reported pro-environmental behavior. In the Netherlands the discussion groups revealed a firmer basis to the social contract between the state and its citizens than was the case in the United Kingdom. Despite public scepticism about the effectiveness of the national government's mass media campaign designed to promote pro-environmental behavior, Dutch people were encouraged by the fact that state had taken the lead in acting responsibly towards the environment. By comparison, the ad hoc and laissez faire approach to promoting pro-environmental policy pursued by the U.K. government was often ridiculed by Nottingham residents. Taken together, such findings serve to highlight the multiple and pervasive influences that social, political, and cultural factors play in developing effective environmental communications—much as the contextualist conception of society suggests.

Conclusions

Summarizing and illustrating complex ideas in this brief way fails to do justice to the subtleties of both contextualist and reductionist conceptualizations of society. We suggest, however, that contextualist perspectives offer new insights about how individuals are engaged with society and how more effective strategies for environmental communication can be developed. Most obviously contextualist approaches ask natural scientists and policy makers to be more critical about their framing of who their publics are and what they will and will not do. In terms of developing a communication strategy for understanding urban ecosystems, educators and policy makers need to recognize the limitations of reductionist conceptions of society, which tend to assume a linear process of learning based on offering "the correct information." Numerous studies suggest that such an approach is not effective.

Working with contextualist conceptions of society means accepting that individuals are socially engaged actors whose environmental understanding and behavior is contingent on where they live, the history of events, their social networks, and social and moral norms. These approaches also recognize that the way society "works" depends upon a reflexive process of mutual trust through which individuals and structures (e.g., organizations, legal processes, rights and responsibilities) come to constitute each other. Gaining peoples' trust and support for education programs which seek to convert high public awareness of environmental problems into pro-environmental behavior, for example, is thus likely to require new ways of working. More participatory approaches to environmental communication and decision making that encourage face-to-face deliberation are capable of forging new social relations through a process that is based on mutual respect and trust. In this way, knowledge claims of experts such as educators, natural scientists, and politicians will add to, rather than displace, the legitimate knowledge claims of other groups in society.

In conclusion therefore, a contextualist approach to society suggests that effective education strategies which seek to promote a shared understanding of the inter relationships between lifestyles and environment will be:

- Heterogeneous in nature and content;
- Localized rather than universal in the scale of their delivery;
- Action-led rather than based on exhortation;
- Supportive of new public forums and arenas which encourage participatory democracy rather than reliant on existing structures and processes of representative democracy;
- Inclusive rather than exclusive in terms of the range of knowledges, experiences, and understandings they respect and accommodate.

Approached in this way, the task of understanding urban ecosystems is not simply one of information gathering and transfer, but one that also needs to acknowledge the influence of a range of other social, political, and cultural processes.

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The Iceberg and the Titanic: Human Economic Behavior in Ecological Models

Jane V. Hall

Keywords: models · economics · human behavior · externalities · natural resource value

Introduction

In the long run the planet has the upper hand, in the short run humans act as if they do and as if this will continue to be the case. This aspect of human behavior is manifested in economic institutions and policies, and in individual responses to economic signals. The implications for how human economic behavior can be incorporated in ecosystem research are founded in the nature of economic models and how they are used to formulate policy.

The title of this chapter reflects the author's perception that the body politic often founds decisions on the seemingly-precise prognostications of economic models, finding comfort in estimates carried out to the third (or greater) decimal. Reliance on guidance from these models in turn generates policies (and consequently affects human behavior and choice) based on the assumptions of the models that produce the estimates. The assumptions, therefore, become embedded in the behavior they are used to predict–in short, a classic self-fulfilling prophecy. This sort of collective social choice suggests a nation that regards itself as being as impregnable as was the iceberg, blithely assuming that any large clouds on the horizon are really only on the scale of the Titanic and that, in any event, ordinary economic decisions will correct the steering before any solid object is encountered.

Examples of this behavior abound. From the recent cover of *Time* asking whether we can do without nuclear power to the on-going debate over the National Acid Precipitation Assessment Program (NAPAP), we see a willing view of the growth economy as an iceberg, when it ultimately is not even in the scale of the Titanic, but more like a lifeboat, fragile and adrift among the complex biological and geochemical systems that typical economic assumptions and models treat so cavalierly.

Given this not altogether optimistic view of how economic understanding drives some basic policies which *in turn* introduce the human factor in ecosystems in new ways (for example, construction of a new highway opens up undeveloped land to housing in ways and on a scale that the choice of a rail system would not), how can an understanding of economics enable scientists to better incorporate the human factor in basic ecosystem studies?

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The answer rests on the most basic concepts and assumptions of general economics, observations from the fields of natural resource and environmental economics, and the degree to which these merely reflect dominant social values.

Basic Tenets of Economics

The majority of economists argue that humans react in specific and predictable ways to economic signals, in a mechanistic scheme often referred to as the price system. In our economy we do rely very heavily on prices to convey tremendously complex information in a very distilled form. The most fundamental tenets are the following:

Optimization. Each individual pursues his own self-interest in a way that maximizes his welfare. By this atomistic process, the system is also optimized, in the sense that no one can be made better off without making someone else worse off. When all producers and consumers follow this rule, scarce resources are allocated in the most efficient way. Two important assumptions that are embedded in most economic models fall out of this. One is that the pursuit of individual self-interest leads naturally and automatically to the best possible outcome for the system, given resource constraints. This then becomes a matter of definition; if individuals are free to pursue their self-interest and do so, the outcome is optimal. (This is also known as Reaganomics.) The other is that maximizing is equivalent to optimizing. Consumers maximize satisfaction, producers maximize profits which means minimizing costs. There is a corollary—the impact of an individual's choices (good or ill) falls on the individual—all costs and benefits of a choice are internal to the decision-maker.

Value. What something is worth is measured by how highly it is valued by humans, and generally only those values that can be monetized are really recognized. Hence the scramble to put a price tag on the ecosystems of Prince William Sound in the aftermath of the Exxon Valdez oil spill. This anthropocentric approach to determine value has been called "economic imperialism" by Herman Daly, but it is the basis for our price system of exchange and, therefore, for production and consumption. Beyond its anthropocentrism, this approach also places greatest weight on the values held by those who own or manage assets in the current time period. By comparing everything in present value terms, the future is discounted to obscurity.

Rationality. We assume all players in this game know their own self-interests, now and in the future, and the best way to successfully pursue them, i.e., to maximize given prices and their private resource constraints. A corollary is that information must be complete, accurate, and fully understood so that all implications of a choice are clear to the individuals.

Marginalism. All of the infinite number of decisions that individuals must make in this rational optimizing process takes place at the margin. The consequences of previous choices are history, and the next decision is forward looking. In short, what's past is past, not prologue. Lest this look like a completely absurd view of the world, economics also assumes that past decisions do enter into future ones in the sense that relative prices change in response to collective market forces. Past decisions to invest, produce, or consume will, therefore, influence current and future prices. The decision to fill your gas tank today is based only on the prospective utility to you of overcoming the inertia of your vehicle, but the collective impact of all of our past decisions about consuming and producing fuels is reflected in current prices and is, therefore, not completely out of the picture. If past choices lead to a change in relative prices, your current (but still marginal) choice will be affected, otherwise it will not. The aggregate impact of all of our marginal choices and the cumulative impact over time can be enormous, but the margin is what counts. This situation sends this message: don't worry about the aggregate in any physical sense. If it is important in terms of human welfare, as measured by value embedded in prices, prices will change and influence the marginal choice appropriately.

Impersonal transactions. All of this works through a complex scheme of production, consumption, specialization, and exchange. Most transactions are at a very distant arm's-length. When I buy an orange, I am nowhere near the grove where the orange was produced and hundreds of miles from the river that was diverted to irrigate the grove. I won't buy the orange directly from the grower or pick it directly from the tree. Our economic system is based on billions of such daily transactions. The implications of this, ecologically, are perhaps reflected best in the recent Goldman Awards, including one to the man who filmed dolphins drowning in tuna nets. That visual report very quickly reduced the arm's-length relationship between the fishery and the folks buying canned tuna.

Continuing and unlimited growth. Most economists, and our political system, at virtually all levels of government, assume that economic growth, and both the associated increased rate of resource mobilization and increased pressure on natural sinks and other species, is inevitable and unlimited (Costanza 1989). The process of substituting away from economically scarcer resources to less scarce ones is assumed to be unlimited. New deposits will be discovered, better technologies will be invented, an altogether different material will be found or developed to do the same thing. etc. This is politically seductive because it implies that even a larger population can (will) always be materially better off than its antecedents were as long as growth is sustained. It also appeals to economists who would otherwise have to face the messy issues of equity, distribution, and anthropocentric values. It is not a fluke that many winners of the Nobel Prize for economics were recognized for their work in championing the virtues and possibility of perpetual growth or demonstrating the mathematical mechanisms by which it can take place (Friedman, Samuelson, and Leontief, to name a few).

Concepts from Environmental and Natural Resource Economics

As human activity increased, along with population redistribution and growth, and industrialization, the mobilization of resources became truly immense and the residuals from this process began to overwhelm natural sinks. This eventually led economists (trailing along after the public health professions) to begin asking how it was that beneficial economic activity could have such unforseen adverse effects and what to do about it. Many of the ideas that emerged are couched in terms of "market failure," i.e., why did prices fail to signal these effects so that individuals would alter their behavior to reduce residuals to the appropriate (optimal) level? Questions about preservation of un-priced or under-priced non-market resources such as species diversity were approached later in adaptations of the same paradigm. The basic concepts follow.

Externalities

It turns out that, in violation of one of the basic tenets, the consequences of individual choice do not all fall on the individual. Say that I decide to cut down a tree and pay someone $100 to do it. The price to me is $100. The cost includes the $100, but also the loss of habitat for birds and other creatures, soil conservation, shade, reduced heat island effect (this is an urban tree), and the aesthetic virtues the tree held for neighbors and passers-by. When I decide I am willing to pay $100 to remove the tree, I am not considering in my decision these external costs that fall on others. It may be optimizing behavior for me, but it won't be optimal for society. This is a small example, but one that is close to home. This is the classic explanation for pollution. Bodies of water and air are often commonly held and will be over-exploited for waste reception because none of us pays a price for their use equal to the damage that our use does. This kind of case was eloquently explained by Hardin (1968). Absent proscriptive regulation, the price to dump industrial waste in a river is zero; the case is similar with clean but warm water from a power plant outfall into a body of water.

Ecosystem changes are external to the decision maker because they do not impact him directly in any material way and they are un-priced and, therefore, not part of his decision. A firm that is maximizing profit will select the least costly production process, as measured in prices that must be paid. There is, therefore, a tendency toward processes where costs are externalized. There is no self-correcting tendency in this case because there is no feedback mechanism, for example, a stream becomes so polluted that it can no longer be used for industrial processing, as was the case in the Ruhr Basin some years ago. The un-priced loss of soil microbes, urban airshed, habitat, CO_2 sinks, etc., continues because they are outside of the price system.

Public Goods

These are the flip side of externalities. They are *under* produced because the benefits of their production cannot be entirely captured by their producer(s). If available to anyone, they are available to many, without exclusion, and they are non-rival in consumption (your use does not affect mine). The traditional example is a lighthouse. Any individual who takes action to "produce" or protect such an asset knows that others will also benefit while they bear all of the costs. Other examples include clean air and species preservation. This is also referred to as the free-rider effect. The result is underproduction, stalwart altruists being the exception, not the rule.

Un-priced and Under-priced Goods

Goods or services that have value but that are un-priced will be overused and underproduced. This is really a generalization of the first two concepts. In essence, such goods and services are partially or completely outside of the price system that makes our economic world go around and will not be accounted for by the only feedback mechanism recognized by most economic models—market prices. Many, or most, of the biological and geochemical foundations of individual ecosystems are un-priced. On the grandest scale, the global ecosystem is un-priced. Narrowing it down, the complex mechanisms and sinks that "produce" stable climate (in terms of human time) are un-priced. Narrowing it again, the value of expanded wetlands is at best under-priced, except perhaps by duck clubs.

Scarcity

It is not traditional to include scarcity *per se* as a notion from the environmental and natural resource sub-disciplines of economics. Perversely, it is the recognition that all resources are scarce and that human wants can, therefore, not all be met, that drives the theory that scarcity can best be ameliorated by individual responses to price signals. Changes in relative prices then, at least in theory, signal increasing or decreasing scarcity. It is natural resource and environmental economists, however, who have focused debate on whether or not we in fact have any adequate methods to measure the degree of scarcity of natural resources and environmental services and what the implications may be of any inadequacies. Recent mathematical and conceptual constructs (Hall and Hall 1984;Hall et al.1992) have demonstrated that increasing scarcity is not always reflected in prices and that other measures are not always good indicators either. The manifestation of scarcity depends on the way in which a resource becomes scarce and on the physical nature of the resource itself. This is in part because un-priced or under-priced goods and services are critical inputs to production and processing of the natural resources we use, and the value of such inputs is, therefore, not reflected in prices of the resources whose production and use they support. Consequently, the price of the resource could actually fall, indicating *decreased* scarcity, even as the most essential inputs to its production and processing became more scarce. Consider the case of a ton of coal mined in Wyoming. Looking at the price trend for that coal will tell you nothing, or in fact mislead you, about the economic scarcity of Wyoming coal. The value of grassland habitat lost to strip mining will be under-priced since reclamation requirements will not restore it fully, the water used is under-priced because of federal subsidies to water projects, the air and water quality impacts of the mining operation will be un-priced, and so on. So, as clean air and water, and grassland become scarcer in Wyoming, the price of coal will not reflect this and coal will be assumed to be no more scarce, or even less scarce while the associated support structure necessary to mine it is rapidly depleted. This logic can be extrapolated to consider what happened to California agriculture during the 1987–1992 drought, etc.

Incorporating These Concepts in Economic Analysis

Economists, in concert with federal agencies such as the Environmental Protection Agency and the Department of the Interior, have put tremendous effort into trying to determine how to incorporate un-priced and under-priced environmental and natural resource assets into traditional economic models such as cost-benefit analyses. They do this, for example, by trying to elicit information about what people are willing to pay to preserve, protect, or enhance some environmental characteristic or factor. The logic of this is that market prices actually paid should reflect what society is willing to pay for a product because that is what is actually paid. So, if you can get people to reveal what a non-priced "product" is worth, you have a surrogate for price, and the economic models will work just fine.

Most of what is now traditional environmental economic analysis is aimed at forcing ecological reality into fairly standard economic models with the inevitable conclusion that once correct price signals flow through the system, optimal use of environmental services and natural resources will result. This comes about either by changing individual behavior directly, or changing it indirectly through legislation and regulation that explicitly correct for these problems by mandating or limiting particular choices. Examples include the average vehicle fuel economy standards and the prohibition on dumping used motor oil in a storm drain.

There are economists who propose, and are developing, measures of value that explicitly incorporate the effects of changing natural assets. (See, for example, Pearce et al. 1989). Price can be a complex concept involving both monetary and physical measures of value. This approach falls under the general rubric of environmental accounting and is a promising concept within the broader field of sustainable development. Ultimately, the influence of resource availability and quality (including environmental amenities) on an economic measure of welfare—sustainable income—would be measured, producing a clear link between ecological and economic models.

Human Behavior in Ecological Models

The similarities between economic and ecological models have been elucidated elsewhere (see Rapport and Turner 1977) and will not be dwelt on here. Suffice it to say that there are parallels in consumption, production, specialization, and exchange. What is more crucial to the issue of how the human factor can be accounted for in basic ecosystem research is how the models are different. The models reflect reality as it is understood by the respective practitioners and, therefore, basic differences must be taken into account. Economic models can be taken as reflecting the significant dominant human values of our society, especially as expressed in the public policies that manage our common non-market resources. Such policies in turn influence human behavior, in essence creating reality. Humans are, in fact, inseparable from nature, but dominant western culture and its social and economic paradigms miss this point (White 1967).

This is where the metaphor of the iceberg and the Titanic comes in. The iceberg, largely submerged and out of sight, is a threat to the existence of the Titanic which looms large. Our industrialized economy and the human behavior that drives it are perceived to be the iceberg. The impacts of economic activity and growth on the essential natural systems that the economy depends on are submerged.

Economic models work on the tenets set forth earlier and, to the extent to which these models either accurately reflect values or determine behavior because of their use in policy formation, these tenets become reality. Understanding the human factor is, therefore, based on assumptions that individual maximization optimizes the system, that the past is past, that very simple signals, i.e., prices, encapsulating extraordinarily complex information are sufficient to maintain this optimization process, that human values are captured in such signals which are adequate measures of value, that each economic actor knows what he is doing, and that growth is inevitable, unlimited, and desirable. Many of these assumptions run directly counter to observations from ecological models. Individual maximization will not likely optimize a natural system (Norgaard 1987). Optimizing and maximizing are not the same thing (the mockingbirds who have converted my shrub to a nursery are not always building bigger nests). Growth within a system is not unlimited. Feedback mechanisms rely on multiple kinds of signals. Interactions are often quite direct. Everything adds up. And so on.

Perhaps most fundamentally, economists assume that the basic concepts can be generalized over the human universe, with no consideration of context (see Vayda, Chapter 6, this volume). The outcomes will differ slightly with culture and resource availability, but the assumptions, mechanisms, and objectives are the same. This is in diametric opposition to ecologists' discovery that context is crucial and even basic concepts may not travel well (diversity begets stability, for example) (Norgaard 1989).

As our political-economic structure has developed, these assumptions embedded in our behavioral models have almost become articles of faith to the body politic. Econometric modeling is more and more relied upon to provide policy guidance on everything from the level of optimal taxation to the value of the Valdez oil spill. Consequently, since large public expenditures, along with an array of taxes, subsidies, and direct regulations, result from such advice and since these resulting policies themselves are determinants of economic behavior, the behavior predicted by the models becomes a self-fulfilling, self-perpetuating prophecy. At this point, whether the assumptions are logical or empirically verifiable becomes irrelevant because behavior is altered by the predictions, even if the assumptions are invalid.

This is why the human factor, at least as represented by economic behavior predicted or measured by economic models, should for now be treated as an independent variable in ecological models. This might not remain true and will not be true in all cases. One can easily find examples of local communities making political choices that diverge from what the models recommend, but this is not easy to find at the national level. We have set ourselves up in the role of the iceberg, the values embedded in the assumptions of the model drive policy and, therefore, behavior, and so we drift along. This is undoubtedly entirely unrealistic as a sustainable role. Any reasonable view of the First and Second Laws of Thermodynamics makes it transparently clear that this cannot go on indefinitely. Yet, the assumption that rising prices signal scarcity, thus creating an incentive to invent alternatives and to substitute something less scarce, underlies the notion that economic growth is a concrete example of the realness of perpetual motion.

As long as policy-makers rely on the illusory precision of economic models to guide them, and society continues to view maximization of material goods as desirable, human economic behavior should realistically be treated as an independent variable in ecological models. This will be true until values change, non-price signals are conveyed in a meaningful way, or the iceberg and the Titanic swap roles–that is when some catastrophic ecological threshold is reached.

Implications for Ecological Models

First of all, the picture is not entirely as bleak as it might appear. There are many excellent economists working hard to point out the fallacies of the dominant economic thinking and to construct useful new paradigms (Norgaard 1987; Boulding 1966; Daly 1984 among others). Others build models that reach policy conclusions at odds with the results of the dominant models, leading to gradual acceptance of alternative views and assumptions, ultimately changing policies and thence behavior (Chapman and Drennam 1990; Pearce et al. 1989; Fisher and Hanneman 1986; Hall 1990; Hall et al. 1992 among many). It is noteworthy that virtually all of the economists in this cadre have embraced direct involvement with their colleagues in the natural, physical, and other social sciences and that they rely explicitly on integrated understanding of the social, biological, and physical systems of the problems they study.

Effective change often takes place first at the local level where much ecosystem research is carried out. Consequently, the most useful way to incorporate the human economic factor into ecosystem studies is to ask what human behavior is predicted to be (housing development, road construction, water diversion, whatever) and then analyze how that predicted behavior will alter the path of an ecosystem's development either directly or indirectly.

The importance to the system of the differences made by the human element can then be evaluated. This in turn ties back to the evolving field of environmental accounting. For example, I tell you that the human population in a region will increase by 35%, leading to a 40% increase in demand for park services. You then tell me what this implies for the integrity of an urban fringe wilderness park. Between us, we know a lot about what the changing human factor means about the quality of the area and the future usefulness of the park to humans as wilderness. Then we know something that can alter human behavior as this information becomes feedback to these humans, probably via the political system. Ehrlich (1981) identifies this as a dual task for ecologists; to predict the consequences of various courses of action and to communicate the results to the public. Without the human factor, the results won't mean much.

So, intellectually and in concrete cases, change in economic thinking is taking place, however slowly. This is a propitious time to begin developing ways to explicitly incorporate the economic manifestations of human behavior into ecological models.

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Forecasting Demand for Urban Land

Paul Waddell and Terry Moore

Keywords: public policy · development · supply and demand · demand for land · models · urbanism · real estate · Utah · market · economics

To this point, the chapters in this volume have focused on land supply. They have addressed issues such as (1) what constitutes buildable land; (2) how environmental constraints, infrastructure and other public policies can affect its buildability; and (3) how to identify land that, though effectively built, can still accommodate new growth through redevelopment. Their assumption has been that there would be some need for a supply of buildable land: in other words, that a demand for buildable land exists. Supply of and demand for buildable land are essentially two sides of the same coin. Assessment of land supply would be incomplete without an assessment of the demand for buildable land. This chapter provides a framework for thinking about the demand for urban land and provides some examples of techniques for estimating that demand.

That land use planning should explicitly address the supply of and demand for urban land is not surprising. Land use planning in the twentieth century United States has always been about forecasting and tinkering with market forces. Land development in the U.S. results from market transactions that occur in the context of public regulations.[1] Municipal and regional governments try to anticipate the demands of growth and to supply the serviced, buildable land to accommodate it. In the U.S., planning occurs in the context of markets. Planning for growth and land use means, inevitably, intervening in the markets for land and development.

If planners think such intervention will make an urban area better (more efficient, aesthetic, satisfying or fair) for the people who live or work there, they must make some assessment of where the market is likely to go with and without a proposed intervention (e.g., a land use regulation). Any evaluation of public policies to manage growth and land development must consider markets, and the accepted paradigm is basic microeconomics: supply and demand.

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[1] There is an ongoing, extensive, unresolved and probably unresolvable debate in the professional literature about the right amount of government regulation of land use and related activities. Most combatants agree that private market transactions in land increase efficiency. The debate is, in our opinion, primarily about the extent of the costs external to those transactions. Free market proponents believe those external costs are small in most cases, and that planning should follow the market rather than attempt to channel it into smart growth (Mills 1999; Staley and Scarlet 1998). Those favoring stronger environmental and land use regulation and planning point to the problems of air and water quality, traffic congestion, sprawl and so on, and believe the opposite. Here, we simply observe that land development has and will continue to be the result of market transactions in the context of land and environmental regulations.

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Consider *sprawl*, as urban areas across the nation are doing increasingly. Sprawl has been cited by voters in recent political polls as the most serious problem facing regions such as Denver and San Francisco (Pew Center for Civic Journalism 2000). Yet there remains substantial debate about the causes and effects of sprawl, and about the efficacy of alternative strategies to contain or mitigate its adverse consequences. Planning for smart growth requires an understanding of the determinants of demand for urban land, the interaction of demand and supply within urban land markets, and how government intervention affects the operation of these markets.

Achievement of public policy objectives for metropolitan areas requires balancing individual and community property rights, local control and regional responsibilities. Consumer choices about location and development within metropolitan areas are responding to available alternatives, prices, constraints and incentives that are heavily influenced by government action within local political and economic structures that have never approximated free markets in the U.S. Moreover, social and environmental externalities within urban markets for land and transportation, natural monopoly in the provision of some public infrastructure and services, and concerns regarding the equity of various market outcomes have long provided a basis for market intervention through planning, regulation and infrastructure provision. In short, policy makers, planners and the public must better understand the nature of market demand and supply of urban real estate if they hope to improve the public policies that intervene in those markets.

This chapter has two aims that shape its organization. The first is to present an understandable framework for exploring the nature of demand for urban land, including its interaction with land supply and the effects of governmental interventions in urban land markets. The second is to assess techniques based on the framework for use in planning to achieve growth management goals. We do not extensively review the literature on the factors influencing demand for specific types of real estate. Rather, we focus on the requirements for a useful analytical approach, and provide an example to illustrate the application of the framework.

A Framework for Analyzing Urban Land Markets

Overview of the Demand and Market for Urban Land

An analysis of demand for urban land logically begins with identification of the consumers for urban land. We focus our analysis of demand on households and firms as the final consumers of urban real estate, in the form of housing and commercial and industrial floorspace. We treat land as a component of the demand for housing and nonresidential floorspace to be occupied by households and firms—one of many attributes of the demand for buildings and their associated lots and location characteristics.

There are many ways to categorize urban real estate. Here, we distinguish among residential, commercial and industrial property as broadly representing demand from three types of consumers: households, firms occupying retail and office space and firms occupying industrial and warehouse space. These broad categories do not illustrate the vast complexities of the urban real estate markets, but do highlight some broad differentiation within the market, and can be extended to support more detailed analysis.

Demand for urban real estate operates within a broad framework of urban markets for land, labor and public infrastructure and services. Figure 1, though simplified, shows how complicated the relationships can be, using the housing market as an example.

Households and firms occupy housing and nonresidential floorspace, and in turn, consume urban land. Developers respond to these demands by using land in creating new housing and nonresidential floorspace, or modifying existing real estate to meet changing demands. Households and

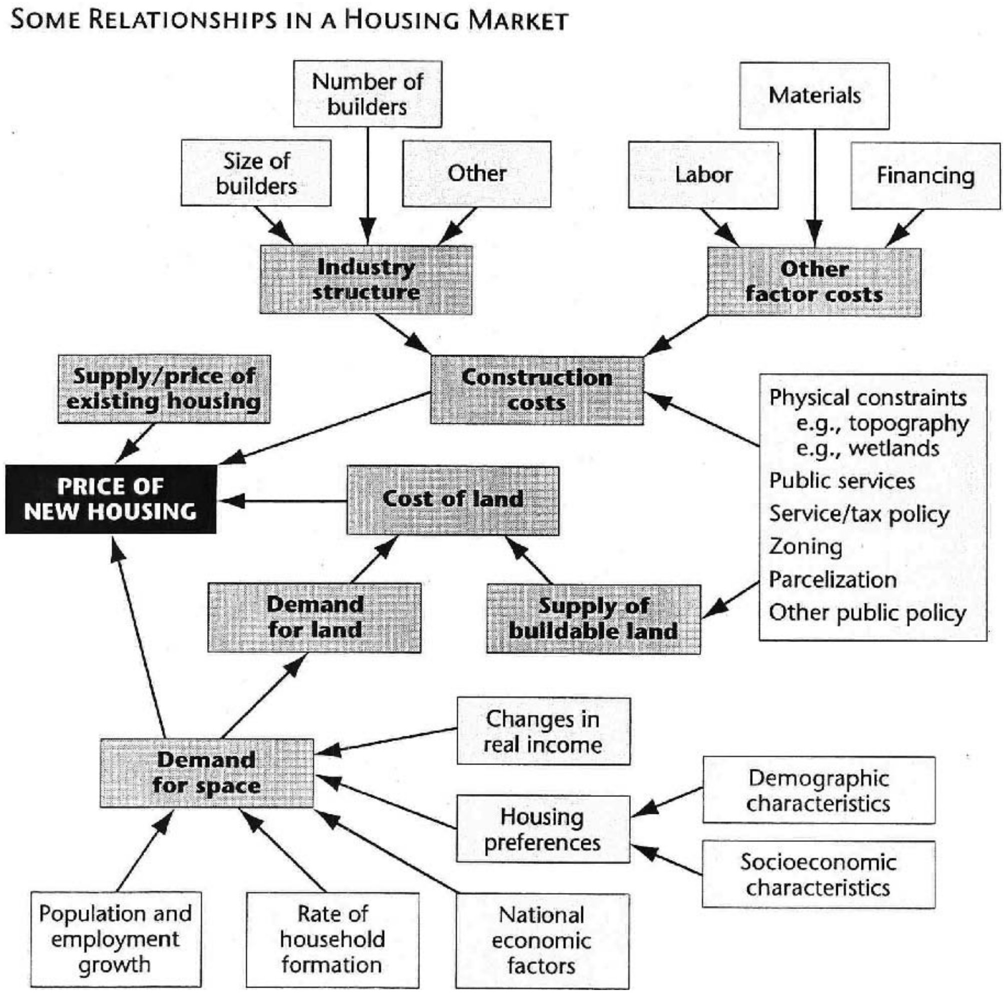

Fig. 1 Some Relationships in a Housing Market

businesses interact through consumer purchases and through the labor market, using transportation infrastructure to access these activities. Governments play an integral role in urban land markets, of course. They build the transportation, water and sewer infrastructures that create opportunities for urban development. They regulate land through zoning, UGBs, environmental regulations and many other mechanisms. They provide public services and facilities such as schools, parks, libraries and police and fire protection. They also levy taxes on property, sales and income, collect fees for various services and offer subsidies for various activities.

Overview of the Analytical Framework That Follows

A fundamental point about the regional demand for land is that it is an aggregation of individual demands for land. Every consumer (as an individual, household, or business) has some preferences for the bundles of services that real estate (both land and structures) can provide, and some ability to pay for those services. In concept, one could think of the demand for land in a metropolitan region as the summation of all these individual demands.

There are two broad approaches to analyzing market demand: one is from the perspective of the individual consumer, which we will refer to as the *disaggregate* approach, and the other is from the perspective of groups of consumers, which we refer to as the *aggregate* approach. There is a continuum between a completely aggregate and a completely disaggregate approach, and moving along this continuum presents trade-offs in terms of analytical requirements and limitations. Aggregate models are simpler to work with and understand, but may obscure important factors that influence demand. Disaggregate approaches require more data and computation, and are inherently more complex. For the purpose of developing a framework for understanding the concepts involved in demand analysis, we begin with an aggregate perspective, and sequentially add details to the analysis to capture important aspects of demand, moving toward the disaggregate perspective in the process.

We begin with a discussion of issues related to real estate demand for a metropolitan region by looking at the aggregate analysis of groups. It is typical for demand analysis to start at a high level of aggregation to get some estimate of the key drivers of real estate demand: expected population and employment growth. That aggregate forecast provides a good starting point, but is not sufficient alone to forecast the demand for real estate products. There are many types of products, consumers and locations; all must be considered to make an informed forecast of demand.

The most simple and aggregate model of demand can be stated as: more people in a metropolitan area (as residents and workers) create more demand for built space, which creates a derived demand for land (a place to put the buildings). That aggregate picture becomes more complicated and disaggregated as one makes it more realistic by considering the following factors:

- *Product differentiation*. There is no single product called “urban land.” Rather, there is a demand for many types of real estate products (e.g., residential and commercial; within residential, single-family or multifamily units; within single-family units, different lot and housing sizes; for a given size, different quality and price).
- *Market segmentation*. Consumers of residential, commercial and industrial land have different characteristics, such as income, that cause them to have different preferences for real estate products and, by extension, land. This fact leads to attempts to break large groups into smaller groups to refine the analysis.
- *Location*. The characteristics of a lot may allow the construction of a building, but the characteristics of the neighborhood and larger subarea contribute to value and demand. Location matters.

Following this aggregate treatment of demand, we then look at individual consumer choices and how those choices can be aggregated to get an estimate of regional demand. That discussion has two subsections:

- *Individual preferences and constraints*. Market segmentation deals with differences among large groups. But in urban areas, those groups comprise tens of thousands of individual decision makers who are not homogeneous and who have varied propensities to accept different real estate products.
- *Submarkets*. The variability of products, consumers and locations creates submarkets that are more or less substitutable.

Finally, we consider some factors that complicate any type of analysis of the demand for land in metropolitan areas:

- *Durability of real estate products*. In the residential market particularly, the long life of buildings means that mobility and filtering must be taken into account when forecasting demand.
- *Public policy*. Policy can affect all aspects of market demand and supply relationships. Future absorption cannot be predicted without assumptions—if not explicit, then implicit—about future public policy: will it be about the same and if not, how it will differ from current policy?

- *Interactions of demand and supply*. What people often refer to as "historical demand" is, technically, the intersection of demand and supply factors at some price. In other words, a forecast of housing absorption must result not only from a consideration of demand-side factors (e.g., demographics, income), but also supply-side factors. If geography or public policy strongly limits buildable land, land prices rise and the amount of land (and, by correlation, housing) changes, as demand and supply factors get into equilibrium with the new prices.

Demand From the Perspective of Groups of Consumers

The Fundamental Drivers of Aggregate Demand

An initial assessment of metropolitan demand for residential, commercial and industrial real estate almost always begins at the level of metropolitan area as a whole. At this level, one would not consider demand for locations within the metropolitan area, but would concentrate instead on the overall quantity of demand. Analysis at the metropolitan level might be used to address such policy questions as how much developable land should be included within an urban growth boundary that is intended to accommodate a 20-year supply of land. For reasons we will explain shortly, however, the answers one might generate with this level of analysis will likely be incomplete and perhaps significantly misleading. Nevertheless, an aggregate assessment is a critical starting point for an analysis of the demand for urban real estate. There are two fundamental drivers of demand for urban real estate at this scale of analysis:

1. The macroeconomic growth of the metropolitan economy, as measured by growth in households (or population) and employment.
2. The space used by each household or job.

Clearly, growth in the aggregate number of households should be expected to precipitate growth in the demand for housing units, just as growth in employment would generate demand for nonresidential floorspace. Such macroeconomic changes may result from interactions between local economic structure, national and global economic shifts, changing production technology, migration flows, interest rates and many other factors. Methods for predicting aggregate changes in population and employment have been presented elsewhere (Isserman 1984).

Given a particular prediction of aggregate economic activity and future growth in households and employment, an assessment of demand for real estate requires estimation of the pattern of real estate utilization of households and firms. Most households will occupy a single housing unit; business establishments require a given quantity of square footage of space per employee. If one can reliably estimate these space utilization ratios (one unit per household and N square feet per employee for firms), then one could make a straightforward translation of growth in households and jobs into anticipated new demand for housing units and nonresidential floorspace for any particular period of analysis. To simplify notation, we will refer to households and jobs collectively as *consumers*, and all types of housing and nonresidential space collectively as *real estate*.

$$D = AR$$

Where D = demand for real estate
A = aggregate number of consumers
R = space utilization ratio (quantity of real estate per consumer)

This simple equation is just that: simple. Even if one knew the space utilization ratios for households and employment in the aggregate, the knowledge would be of little help for understanding the demand for different types of real estate, such as apartments or offices.

More detailed analysis (disaggregation) is required, which we add incrementally in the sections that follow. Our objective is to specify demand by subgroups of consumers for specific types of properties at different locations within a metropolitan area. Thus, we add to the basic equation in the next sections to account for the market segmentation of consumers and differentiation of real estate products by type and by location characteristics.

Product Differentiation

The differentiation of real estate products complicates our assessment because it adds a dimension of choice of real estate type to the assessment of demand.

Consider just the residential market, partitioned into different housing types (e.g., single-family, duplex and multifamily). There is no longer a simple translation of the number of net new households predicted for a metropolitan area and numbers of housing units of each type that will be demanded. It will depend on the probability that new households will choose each of these three types of housing. More generally, we could rewrite the initial equation as:

$$D = \sum_i D_i = \sum_i A P_i R_i$$

Where
D_i = the demand for real estate of type i
A = the aggregate number of consumers
P_i = the probability that a consumer will choose to occupy real estate type i
R_i = the space utilization ratio for real estate type i

The classification of real estate should be meaningful to consumers and suppliers and to consideration of policy. An analyst does not want the burden of unnecessary detail to complicate or confuse the analysis; nor, however, does she want to generalize across categories of real estate that fundamentally differ in how consumers, suppliers or policy view them. One can examine these products as a set of separate but interdependent real estate markets, sometimes referred to as real estate *submarkets*.[2]

Various classification schemes for urban land and real estate are available and potentially useful for this analysis. Such classifications are developed and used by municipalities, appraisal districts and other local governments for planning. Zoning and land use plan designations provide alternative means to differentiate urban land according to development restrictions. The real estate industry uses different typologies for monitoring market activity. Within the planning and urban design communities there is substantial interest in classifying urban land and real estate in ways that assist in evaluation of and planning for transit and pedestrian access, neotraditional design and more dense and mixed-use neighborhoods (Calthorpe 1994).

It is not our purpose to systematically review these alternative classifications and make conclusive recommendations on a preferred classification. Instead, we offer a simple recommendation for a basic scheme for use in assessment of demand for urban real estate, based on the analysis requirements discussed here, and on general expectations of availability of such data within local administrative records (e.g., tax assessor parcel files).

[2] Note that submarkets may be defined on the basis of the real estate product, on the basis of geographic location or on some combination of these two factors.

Table 1 Real Estate Product Differentiation

Residential	Commercial	Industrial
Single-family detached	Retail	Light manufacturing
Rural density	Strip center	Heavy manufacturing
Low-density	Neighborhood center	Industrial incubator
Mid-density	Power center	Warehouse
High-density	Community center	
Duplex/townhouse	Power center	High-tech/flex space
	Regional mall	Research and development
Condominium	Office	
Low-density	Low-rise	
High-density	Mid-rise	
Apartments	High-rise	
Low-density		
Mid-density	Campus	
High-density		

Table 1 differentiates residential real estate in a way that would support assessment of household demand for housing and land by income and household size. It illustrates the potential problem of distinguishing between structure type and tenure in residential markets (e.g., “condominium” and “apartment” are tenure types, not structure types). Commercial and industrial real estate are differentiated to support assessment of business real estate demand by industry and occupational mix.

This classification scheme would be appropriate for assessing demand for real estate by different types of consumers, but does not address such important aspects as price or location. It could be augmented by cross-classification with price range and location attributes, such as area types defined by density and mix of land uses, but at the cost of additional complexity and data requirements. Many assessor files are not likely to contain this detail of land use classification of commercial and industrial real estate, necessitating some generalization of these categories.

Beyond these characteristics, for purposes of classifying real estate products for analysis, we recognize that many structural attributes of housing and commercial structures are relevant to the demand for real estate. Such housing characteristics as house size, number of bedrooms, style of construction, age, quality of construction and the combination of amenities are all potentially important in influencing household demand. Similarly, the structural characteristics and quality of commercial and industrial buildings will influence demand. These attributes could be examined within disaggregate analysis of individual properties, or through a cross-classification of real estate types and locations, using averages or distributions of the attributes.

Market Segmentation and Market Conditions

The recognition of real estate product differentiation is directly related to the observations that consumers are not identical, and that consumers make substitutions across alternative real estate products based on their attributes and prices. First, examine the composition of consumers for residential real estate. Households differ by income, stage of lifecycle, number of workers, number of children and many other salient features that influence their preferences for housing types. Housing utilization ratios may also be influenced by market conditions. Under tight market conditions with high prevailing rents, households may choose to either double up within a single unit or delay the formation of a new household. In other words, though household characteristics may be correlated with a propensity to choose a certain housing type, those choices are affected by market forces—the same household may choose a different housing product in response to changing market conditions (especially price).

Just as households represent different types of demands for housing, so do businesses vary by industry and occupational composition of the jobs at each establishment, with concomitant variation in their demands for differing types of nonresidential space. At first glance, use of industry as a classification for jobs would seem to provide a valuable tool for predicting the distribution of demand for different types of nonresidential floorspace. Manufacturing sector jobs tend to occupy industrial space, retail sector jobs tend to occupy retail space, and finance and service jobs tend to occupy office space.

It is clear, though, that these generalizations are not robust, and they become problematic when one considers the variation in space utilization geographically. Downtown office buildings usually contain space occupied by headquarters or other administrative functions of firms, classified as manufacturing and other "industrial" sectors. Retail establishments occupy office space and service firms occupy retail space. Using both industry and occupation may provide a more robust, though still limited, means to identify reasonable patterns of space utilization by business establishments.

The space utilization ratio of employees, moreover, is not likely to be fixed, even within a particular type of space, such as an office. The number of square feet per employee may vary among industrial, warehouse, retail and office uses, but there will also be substantial variation in the utilization ratio within each type of real estate, in response to market conditions. Tight markets with expensive lease rates will prompt a decrease in the quantity of space used per employee, as firms economize on an expensive input. Moreover, rates will vary with business cycles. For example, after a long trough it is possible that existing office space has a lot of vacant space, and that many years of expected employment growth could be accommodated with relatively low rates of new construction. In these overbuilt periods, vacancy rates in excess of a normal rate needed to facilitate normal market turnover trigger rent decreases and stimulate businesses to lease more space per employee than they would under tighter market conditions with higher rents.

Technological changes—such as those in production technology or increased use of information technology—may prompt changes in the intensity and nature of space utilization of firms. As manufacturing processes have been transformed over the last several decades, they have altered real estate demand among manufacturing firms toward more horizontal configurations that are more land intensive. In addition, changes in transportation technology favoring truck transport have led to changes in locational requirements toward suburban highway facilities (DiPasquale and Wheaton 1996).

By distinguishing between different types of real estate, an assessment of demand may begin to address the complexities of the market for residential, commercial and industrial development. It raises the possibility of identifying distinctions between different types of consumers that have varying preferences or requirements for these types of real estate. Small households at an early lifecycle stage with relatively low income are likely to demand smaller, less-expensive housing units, as compared to affluent households with large families. Wealthy, empty-nest households may exhibit less demand for large-lot housing than they did at earlier stages in their household's lifecycle, and may opt more frequently for high-quality, in-town condominium housing. These kinds of preference maps are what common sense and observation might suggest, but are not of much value without a systematic means for quantification and analysis.

Our simple model of space demand could account for market segmentation and price effects by adding a component to distinguish types of consumers, and making the probability of choice of a real estate type and the space utilization rates both functions of prices, as follows:

$$D = \sum_i \sum_j D_{ij} = \sum_i \sum_j A_j P_{ij}(\pi) R_{ij}(\pi)$$

Where

D_{ij} = the demand for real estate of type i by consumer type j
A_j = the aggregate number of consumers of type j

$P_{ij}(\pi) =$ the probability that a consumer of type j will choose real estate type i, as a function of prices (π)

$R_{ij}(\pi) =$ the space utilization ratio by consumers of type j for real estate type i, as a function of prices

Like the classification of real estate products, we need to classify the characteristics of real estate consumers salient to the analysis of their demand for real estate. Tables 2 and 3 present market segmentation strategies for households and employment used by the authors in a project to develop models of real estate demand and supply.

Table 2 Market Segmentation of Households

Income	Age of Head	Household Size	Children
Under \$10,000	Under 20	1	0
\$10,000–24,999	20–49	2	1 or more
\$25,000–49,999	50–64	3	
\$50,000 or more	65 or over	4, 5, 6 or more	

Table 3 Market Segmentation of Employment by Industry

Two-Digit Standard Industrial Classification	Description
01–14	Resource extraction
15–17	Construction
20–39	Manufacturing
40–51	Transport, communications, utilities, wholesale trade
52–53, 56–57, 59	General retail
54, 58	Restaurants and food stores
55, 75	Auto sales and services
60–62, 67	Finance
63–66	Insurance and real estate
73, 81, 87	Business and professional services
80	Health services
70–72, 76–79, 83–86, 88–89	General services
82, 91–99	Government and education

Location

The first truism of real estate is that location matters. Real estate is unique as a commodity because a building is (generally) inseparable from the land upon which it sits. For example, a housing unit inherits, by virtue of its location, the characteristics of a particular neighborhood and its social composition, and a pattern of accessibility to employment, shopping and other amenities. We address location by adding a location subscript to differentiate demand by each type of consumer for each type of real estate at each location:

$$D = \sum_i \sum_j \sum_k D_{ijk} = \sum_i \sum_j \sum_k A_j P_{ijk}(\pi_k) R_{ijk}(\pi_k)$$

Where

$D_{ijk} =$ the demand for real estate of type i by consumer type j at location k

A_j = the aggregate number of households or employment of type j

$P_{ijk}(\pi_k)$ = the probability that a household of type j will choose housing type i, as a function of prices at location k

$R_{ijk}(\pi_k)$ = the space utilization ratio by consumers of type j for housing type i, as a function of prices at location k

The simple model now contains the elements of a framework with which we can assess spatially distributed demands for different types of real estate by different types of consumers.

Since a building is generally inseparable from its location, demand for real estate is intimately related to demand for location. The assessment of real estate demand in the aggregate may, in fact, be quite biased by unmeasured differences in demand for location. Stagnant overall population levels in a metropolitan area do not necessarily indicate stagnant demand for new construction.

Consider the unfortunate reality many metropolitan areas currently face. The inner core of residential and employment activity may suffer substantial decline while the suburban areas grow rapidly, fueling sprawl in a stagnant overall economy. Causes for this internal redistribution and dispersal of population and employment have been attributed variously to middle-class flight from poor schools and high crime in the inner city, racial prejudice, subsidized suburban infrastructure and fiscal competition (Orfield 1997). In these cases, there is substantial mismatch between aggregate economic growth and aggregate demand for new urban land consumption. Failure to recognize this pattern could lead to a serious underestimation of the actual rate of urban land conversion. More significantly, failure to understand and address the causes of this pattern could undermine an otherwise solid growth management strategy.

How should one measure location? What aspects of location are important, and how do they inform a program of monitoring or planning for smart growth? To begin, one could consider several alternative units of analysis for location effects on demand: an individual parcel; zones, e.g., traffic analysis zones; a neighborhood or employment center; a city or county; an arbitrary grid. Each geographical unit has merit as a potential basis for analysis. City or county boundaries form natural political subdivisions within a metropolitan area that would serve to reflect different combinations of public services, infrastructure and taxes. They also form the basis for land policy through zoning and the jurisdiction's comprehensive plan.

But there are many location factors that may be meaningful to an analysis of demand that are more detailed in scale than city or county boundaries. At the extreme one could examine demand at the parcel level, allowing the maximum ability to identify location characteristics, especially using the modern capacities of geographic information systems (GIS). But, while parcel-level data is attractive from the perspective of representing a functional unit of analysis of real estate demand, it will be subject to substantial data error that complicates its use (Waddell, Moore and Edwards 1998). Moreover, complications occur because a legally transferable and developable *parcel* is not always identical to an assessor's *tax lot*.

One of the other intermediate geographic units will probably be more suitable for general use in analysis of real estate demand. Traffic analysis zones are particularly useful because metropolitan planning organizations will maintain travel model systems that can produce important accessibility estimates at this level, and may monitor development at this level as well. Neighborhoods and employment centers are more natural units of analysis, since they represent functional geographies for real estate and may approximate meaningful submarkets.

All of these geographic units (including parcels) are subject to one criticism related to their use in quantitative analysis: they are arbitrary in shape and vary significantly in size. The implication is that the results of the analysis will be heavily influenced by the geographic unit of analysis, a problem that geographers have labeled the *modifiable aerial unit problem*. If one uses an arbitrary rectangular grid as a unit of analysis, some of these problems are reduced, and there are gains

Table 4 Location factors influencing demand for real estate

Residential	Commercial	Industrial
Access to employment	Highway or arterial frontage	Access to highway
Access to shopping	Access to customers	Access to airport
Socioeconomic composition	Access to labor	Access to labor
Open space	Access to related businesses	Access to related businesses
Land use mix	Land use mix	Land use mix
Taxes	Taxes	Taxes
Public services	Public services	Public services
Transit access		

in processing efficiency, but additional processing is needed to report findings for more common boundaries that the public understands.

What location factors influence real estate demand? Table 4 summarizes some principal location factors identified in the literature as influencing household or firm location.

Accessibility has long been recognized as a key determinant of demand for real estate by location (Muth 1969). Much of the character of modern urban form has been attributed to adaptation to the now-ubiquitous automobile. For households, access to employment and shopping are significant, though perhaps less so today with the degree of automobile ownership and transportation capacity (Giuliano 1989). There has been a recent surge in interest in more localized, pedestrian-scale accessibility and its potential for reducing vehicular travel (Handy 1993). There is a substantial body of research documenting the importance of socioeconomic, lifecycle and ethnic composition of neighborhoods in determining residential location choices (Downs 1981). Similarly, neighborhood crime and school quality are often cited influences on housing choice.

For business location, access to customers and labor force figure prominently. For firms producing or distributing transport-costly goods, access to a shipping node by rail, highway or air may also be significant. The agglomeration economies, or efficiency gains of locating close to other related businesses with which face-to-face contact is important, is generally credited as being the origin of the patterns of clustering that form central business districts and subcenters (DiPasquale and Wheaton 1996). With the most rapid growth of employment occurring outside recognizable centers, it is possible that these effects are dwindling over time in response to other factors (Waddell and Shukla 1993).

Demand From the Perspective of Individual Consumers

Individual Preferences and Constraints

The previous sections explain the demand for residential, commercial and industrial real estate products and land based on the behavior of consumers as *groups*. Now we shift perspectives to look at the behavior of the *individual consumers* within these groups, which allows more direct examination of product differentiation, market segmentation and locational effects.

We focus on the probability term in the preceding version of the model. The idea is to formulate a model that predicts the probability P that a representative consumer of type j will choose a particular real estate site z from the available vacant inventory:

$$P_j(z) = f(Y, \pi, H, N)$$

Where

$Y =$ household income or firm profit
$\pi =$ the price of the building

H = a set of structural characteristics of the building
N = a set of location characteristics at the site

The most common statistical technique for estimating this model is the logit model, which predicts the probability that each of a set of discrete alternatives will be chosen, as a function of the characteristics of these alternatives and the characteristics of the consumer (McFadden 1973):

$$P_j(z) = \frac{e^{\alpha(Y-\pi_z)+\beta H_z+\delta N_z}}{\sum_z e^{\alpha(Y-\pi_z)+\beta H_z+\delta N_z}}$$

Where

$P_j(z)$ = the probability of a consumer of type j choosing site z
Y = household income or firm profit
π = the price of the building
H = a set of structural characteristics at the site
N = a set of location characteristics at the site
α, β, δ = sets of estimated demand parameters

A principal challenge in operationalizing this model is choosing the unit of analysis. As noted earlier, the unit of choice could range from the parcel to the city, with clear tradeoffs along the way. Let us suppose, for now, that we are using the parcel or a relatively fine grid as the unit of analysis, thereby reducing one aspect of complexity by allowing us to make the simplifying assumption that each alternative represents a homogeneous choice. At this level of detail we could not possibly estimate the choice problem with all the alternative parcels actually enumerated as choices. Fortunately, it has been shown that we can obtain consistent estimates of the parameters using only a random sample of alternatives (Ben-Akiva and Lerman 1985).

A significant concern regarding the assessment of demand for alternative types of real estate and location emerges from consideration of new forms or configurations of real estate. The most common approach to analyzing the demand for a variety of products is to observe the purchases made by various consumers, the characteristics of the consumer and the product they purchased, and to then undertake a quantitative analysis of the consumers' "revealed preferences." Assuming there are three brands of a product available and that we observe a substantial sample of consumers choosing from these products, we could estimate a choice model such as that in the preceding section, and measure the effect of consumer income, age and other observed personal characteristics, as well as the effect of price, quality and other product characteristics, on their choice. The resulting model would encapsulate the relationships between these characteristics of the consumer and the alternative products, and provide a basis for predicting market share based on changes in the mix of consumers or in one or more of the characteristics of the products.

Now consider the introduction of a new product not currently on the market. Assume it is substantially the same as the other products, in the sense that each of its attributes falls within the range of the observed attributes of existing products, and the mix of attributes was comparable to existing products. Under these conditions, we might make reasonable predictions about shifts in market shares precipitated by the introduction of the new product, using our estimated model of revealed preferences. However, what if the new product's characteristics substantially differed from the existing products, either because one or another of its attributes were beyond the range of the existing products' attributes or its mix of attributes was substantially different or it contained a significant new attribute? It would then be inadvisable to consider applying the model based on revealed preferences. Our experience with these products would not provide much insight into how consumers would compare the new product to existing ones.

In this example, we face a problem common in urban land use and transportation planning: introduction of a new transportation mode, such as light rail; or consideration of real estate alternatives that are relatively new and untested in the marketplace, such as certain configurations of mixed-use, high-density development or neotraditional urban design in suburban settings. The problem is that we have inadequate information—from an examination of revealed preferences—to make plausible predictions about the market shares that will be commanded by these new products. This problem has surfaced in other fields, particularly in market research. The general solution is the use of stated-preference surveys that ask respondents to compare hypothetical choices constructed to efficiently measure the relative preferences of consumers to the characteristics of new alternatives as compared to existing ones. Since responses to hypothetical situations differs from actual behavior, stated preference surveys alone are perhaps somewhat suspect. But when combined with revealed preference analysis, such responses can provide a valuable means to assess new alternatives that adjust for such biases (Small and Winston 1999).

Submarkets

To a large extent, we have already discussed submarkets by discussing differences in consumers, real estate products and location. In this section, we want to combine those ideas into the single idea that such variability creates submarkets.

Consumers will consider the available real estate alternatives and may determine that some available products are relatively similar, and so are close substitutes, whereas others are so different that they are not substitutes at all. Similar substitutable alternatives—such as adjacent houses with the same floorplan in the same subdivision—form collections of real estate alternatives toward which particular consumers may be relatively indifferent. As a result, they will price-shop among available alternatives within the preferred cluster. In other words, they will be price-elastic in their demand for alternative units that are relatively substitutable. As we compare this cluster to more distinct (therefore less substitutable) alternatives, we could group these latter alternatives into relatively homogeneous clusters that represent other submarkets, the demand for which will reveal varying degrees of substitutability.

The submarket concept is useful in that it may provide a valuable means to reduce the complexity of assessing the full detail of the real estate inventory, parcel by parcel, while capturing the essential elements that affect differing demand and supply reactions and resulting effects on vacancy and prices.

If we use the differentiation of real estate products to represent different submarkets that are subject to different consumer preferences, supply functions or policy, we can treat them as linked submarkets. It is likely that submarkets, like individual parcels, will be regarded by consumers as substitutes with varying degrees of similarity. Similarly, from the supply perspective, these submarkets represent different targets for the supply of real estate, with differing rates of return. To incorporate the choice of submarket into the demand function presented above, we can introduce it as a marginal choice that conditions the choice of an individual property within a submarket. To do this, we estimate a second equation for the marginal choice of submarket, and make the choice of property conditional on this choice:[3]

[3] This formulation, known as a nested logit model, allows changes in the availability or characteristics in one submarket to influence the choices between submarkets (Ben-Akiva and Lerman 1985).

$$P_j(z)|P_j(s) = \frac{e^{\alpha(Y-\pi_z)+\beta H_z+\delta N_z}}{\sum_z e^{\alpha(Y-\pi_z)+\beta H_z+\delta N_z}}$$

and

$$P_j(s) = \frac{e^{\alpha+\beta\Phi_s}}{\sum_s e^{\alpha+\beta\Phi_s}}$$

Where $P_j(s) =$ marginal probability of choosing submarket s
$\Phi =$ inclusive value (sometimes referred to as the Logsum) from the conditional choice of property, or expected utility from choosing a property within this submarket

The full probability of choosing a particular submarket and a property within it is then the product of these marginal and conditional probabilities. This formulation allows changes in the availability or characteristics in one submarket to influence the choices between submarkets. In the marketplace, one would expect that if a particular submarket became relatively less expensive it would attract more consumers, all else being equal.

Submarkets offer a mechanism to analyze groupings of real estate products organized by structural and locational characteristics into groupings that represent relatively similar and substitutable alternatives to consumers. Within these submarkets, consumers and suppliers interact to demand and produce real estate, and government policy intervenes through various mechanisms such as infrastructure, services, regulation, taxes, fees or subsidies. They exhibit varying degrees of substitutability, based on the perception of similarity by consumers. They are also connected, so that a change in one will affect other submarkets to the degree that they are similar and substitutable (Rothenberg et al. 1991).

One risk in the proposed method of analysis is that the logit technique assumes *independence of irrelevant alternatives*, meaning that if we add a seemingly irrelevant alternative, it will draw proportionally from all other alternatives. This assumption is fine unless some alternatives are more similar substitutes for the one being added, and therefore more of the shift in probability should draw from these alternatives. Provided that the essential characteristics that describe the similarity are included in the model estimation, this does not present a problem. Nesting the model as a set of marginal and conditional choices is a general solution to observed violations of this assumption (Ben-Akiva and Lerman 1985).

The discussion to this point has led to the development of a relatively simple model of real estate demand that incorporates some of the key elements: product differentiation, market segmentation, individual preference, constraints and submarket substitutability. We now seek to frame this discussion within the broader dynamic of real estate markets, and explore the effects of interaction of demand and supply in the short and long run, and of governmental intervention.

Real estate submarkets are linked, not only within a particular type of real estate, but also between real estate types that are not close substitutes, i.e., residential, commercial and industrial, for at least three reasons. First, all real estate is competing for location within a finite inventory of land. Restrictive single-purpose zoning practices may make this competition secondary to policy constraints, but it does not obviate the basic competition for land that exists in the market. Second, properties of one type may generate externalities (like pollution) that either positively or adversely affect properties of other types in the vicinity. This, after all, is the basis for zoning. Third, there is a continual interaction between these real estate submarkets as housing follows jobs and vice versa, in a slow dance of decentralization.

Acknowledging these interactions, the analysis of real estate demand for one type cannot be thoroughly understood without consideration of the others. Nevertheless, it is quite uncommon to see these interactions systematically explored in an analysis of real estate demand.

Other Considerations in Assessing Demand

Whether one attempts to estimate land demand based on an analysis of groups or individuals, those estimates should include consideration of the durability of real estate products, the influence of public policy on the markets for real estate, and how demand and supply interact.

Durability of Real Estate Products

The built stock of real estate is highly durable, lasting over 100 years if well constructed and maintained. Given this longevity, real estate supply is predominantly offered from the existing inventory of building stock, and new construction reflects only 2–3 percent of inventory in any given year (Rothenberg et al. 1991). New housing construction is principally targeted at middle-class and affluent consumers with preferences for larger houses and lots in suburban locations. A combination of building codes, minimum-lot-size zoning and competition from the downgrading of the existing housing stock make new construction at the low- and moderate-income scale relatively less profitable (Rothenberg et al. 1991).

Viewed from the perspective of the consumer (the demand perspective) the durability of the housing stock and the tendency of new construction to be high-end housing lead to a pattern of upward mobility through the housing stock. This process, described by some as *housing filtering*, also means that much of the housing stock will move gradually toward lower-income households, with accompanying reductions in housing maintenance. Upper-income households are relatively more likely to move into new housing, creating vacancies in existing housing. Lower-income households are typically making upgrade moves within the existing housing stock, following this chain of vacancies. Ultimately the chain of vacancies accumulates in the neighborhoods viewed by households as least desirable (that is, those with the least demand) and may lead to abandonment of units that become unprofitable to maintain.

Housing mobility is also likely to vary substantially over the lifecycle of a household, with newly formed young households experiencing relatively high rates of mobility compared to elderly households. This tendency is compounded by consideration of the effect of housing tenure. Renter tenancy requires relatively low transaction costs to move, whereas the relocation costs for a homeowner involve substantial transaction and search costs, both in selling and buying a new home.

The implications of these general tendencies, for an analyst estimating the demand for residential land, are (1) that changes in the balance of demand and supply can influence the rate of housing filtering, leading to more or less demand for new housing construction on vacant land; and (2) that housing demand is influenced by patterns of mobility tied to the changing demographic composition and income distribution of the population.

Public Policy

Direct and indirect effects of government intervention cannot be ignored in assessing demand for real estate. Table 5 shows some of the potential direct and indirect effects of public policy on demand for real estate.

Table 5 Direct and Indirect Effects of Local Government Policies

Direct Effects on Demand	Indirect Effects via Supply
Zoning, affecting land use mix	Zoning, affecting density
Provision of parks and other amenities	Property taxes
Property taxes	Permitting requirements and delays
Infrastructure, especially transportation	Development impact fees
Quality of public services, especially schools and public safety	Concurrency requirements
	Urban growth boundaries

The direct effects of local government policy on demand are those that alter the attractiveness of a particular location for different types of households and businesses. Clearly, the provision of better infrastructure and public services, accompanied by lower taxes, should increase demand, while the reverse combination will depress it (all else being equal). Property taxes affect demand both directly and indirectly; they affect owner-occupants directly, and tenants of housing or nonresidential floorspace indirectly, as property taxes are passed on, at least in part through rents.

Government policies also affect demand indirectly, through their effects on the supply of real estate. Zoning, for example, may restrict the supply of land for certain types of development that are in high demand, thus driving up prices for this submarket. Effects of UGBs on real estate markets—particularly on housing prices—have been widely debated, but with little conclusive evidence as to the magnitude of the price effect (Knaap and Nelson 1993). Construction delays created by permitting and review processes make market supply less elastic with respect to fluctuations in market demand, and create greater volatility in prices in the short-run. In turn, price fluctuations affect market demand and may significantly alter location patterns for different types of consumers. For example, a delayed supply response to a surge in demand for apartments may create a tight housing market (low vacancy rates) in the short-run, which could mean rent increases that precipitate displacement of low-income renters to less expensive (less desirable) locations. The way that consumers are sorted out across the available real estate inventory, then, is a function of the relative availability, characteristics and pricing of the real estate, all of which may be directly or indirectly affected by government policy.

Analysis of demand for urban real estate may be motivated by an interest in setting public policies (e.g., UGBs or zoning), but demand is not independent of these policies. The question of where to draw a UGB to accommodate a 20-year supply of land becomes more complex once one acknowledges that the answer depends largely on the effects of government policies on demand and supply of real estate, in both the short- and long-term. In other words, attempts to assess real estate demand without considering the interactions of demand and supply and the effects of government policy, will provide biased information.

Interactions of Demand and Supply

The preceding discussion focuses on choices consumers make from the available supply of real estate. A condition of variable demand and fixed supply effectively describes the short-run situation that active consumers in the real estate market face. If demand surges within a metropolitan market over a brief period (say, one month to a few months) because of significant job growth or in-migration of households, housing choices of such new arrivals are constrained by the existing supply of real estate. New construction may be initiated in response to the surge in demand, the drop in vacancy rates and the resulting surge in prices. But it takes time for developers to acquire sites, secure permits, obtain financing and actually build the new supply. These delays cause much of the short-term fluctuation in vacancy rates and prices, which in turn influence demand.

In analyzing demand, then, one must consider the effects of short-term supply constraints and the volatility in vacancy rates and prices that follow the changes in demand. In the longer-term—during which developers can respond to vacancy and price signals in the market and bring new construction to the market—one can consider supply elastic, or responsive to the demand-induced changes. In this longer-run scenario, new demand will translate into adjustments in the supply of real estate, both from new construction and from modification of the existing stock. If one considers the real estate market as a set of related submarkets, changes in demand that affect one submarket will end up triggering an array of responses, both within this submarket and other submarkets close to it in substitutability. New construction and conversion of real estate from other submarkets will flow into a submarket that has experienced a surge in demand and, therefore, in real estate prices and rates of return. These modified prices will induce some consumers to switch to relatively less-expensive options in the short-run, with partially offsetting moves after supply adjustments partially offset the initial changes.

Implications

These models of demand, though simple, provide some insight into what a more-detailed analysis of demand should consider:

- Differentiation of real estate product types along dimensions meaningful to real estate consumers;
- Market segmentation that recognizes important distinctions among different types of consumers;
- Location factors influencing demand;
- The effects of durability of the building stock on demand for and modifiability of existing real estate;
- Public policy. Demand for real estate is not independent of governmental action that may affect demand directly through property taxes, or indirectly through their effects on supply, such as potential housing price effects of zoning constraints, UGBs or development impact fees;
- Demand and supply interactions. Demand cannot be completely understood without considering interactions with supply; for example, the supply and demand decisions of owner-occupant households are flip sides of the same coin.

An Example of a Method for Forecasting Land Demand

We have built a case for evaluating demand within a broader context of market demand and supply interactions and government policy interventions, and pointed out the limitations of extrapolations of demand as a method for informing policies related to managing growth. But a framework needs specific methods if it is to have a practical application. In this section we discuss methods that could allow planners and policy makers to make more informed, useful estimates of (1) long-term demand for residential, commercial and industrial land and real estate by location within a metropolitan area; and (2) the potential consequences of smart growth policies.

First, a prerequisite to any intelligent analysis of the interactions of market factors and public policy is the development of a suitable system of monitoring different types of development (see Bramley, this volume). Second, suitable analytical methods should be brought to bear on these data. The complicated interactions among demand variables (and among demand variables with supply variables) lead to modeling.

Good models become complicated. The next chapter explores the Metro-Scope model used by Portland Metro, which links a land monitoring system to real estate and transportation activity. An even more detailed model of real estate demand and supply, UrbanSim, has been recently developed and applied in several metropolitan areas (Waddell 2000; 2001). UrbanSim is a simulation system that interacts with metropolitan transportation models and integrates model components that simulate real estate development subject to planning constraints, real estate demand by households and jobs and real estate prices, and the interaction of these using locations defined by 150 meter grid cells. The model system internalizes many of the considerations identified in this chapter, but is complex and requires substantial data such as parcel and employment databases to calibrate and apply. It is available as open source software at *www.urbansim.org*. An overview and case study of the application of UrbanSim in Eugene-Springfield, Oregon is available elsewhere (Waddell, forthcoming). Simpler models, however, can still be consistent with the main principles outlined earlier in this chapter.

In this section, we provide a simpler application from a regional study in the Salt Lake City metropolitan area, entitled the Greater Wasatch Area Housing Analysis. This analysis does not require specialized software or an integrated model; it works with standard data sources and spreadsheets to make conclusions about future housing demand.[4] Our purposes in describing this model are to illustrate how some of the concepts of demand described above become operationalized in a real-world setting, and to provide a starting template for regional governments that might want to do this type of forecasting. The principal differences between this methodology and those presented in the analytical models is that the latter deal with interactions between demand and supply and add considerable detail in the treatment of location.

The Greater Wasatch Area Housing Analysis was a forecast of housing demand (it did not consider nonresidential demand) completed for Envision Utah, a public-private community partnership dedicated to studying the effects of long-term growth in the Greater Wasatch area of northern Utah (ECONorthwest 1999). Envision Utah was charged with developing land use alternatives to accommodate 20 years of growth in a 10-county area around Salt Lake City. Debate about the feasibility of some of those alternatives led to a recognition that the process needed some analysis of the long-run future of regional housing markets. The purpose of this analysis was to describe, at a regional level, what kind of housing existed in 1999, and what kind of new housing would be likely to be demanded in the next 20 years, given likely changes in demographics, market forces and public policy.

The analysis took a long-run perspective on housing—it looked at long-run trends and downplayed short-run cycles. That approach was shared by the Governor's Office of Planning and Budget, which prepares the official state and county population forecasts. By using official population forecasts as the basis for the analysis of housing demand, the analysis implicitly considered many of the key demographic and economic variables that influence those forecasts.

The housing analysis was based on a conceptual framework that recognizes the relationship of demand and supply, and the exogenous effects of demographic and socioeconomic changes, physical constraints and government policy. The framework is consistent with the description of the interaction of real estate markets described earlier in this chapter. In the context of the demand framework described above, the housing analysis covered, at some level, the following categories of variables:

- *Aggregate demand*. The state's long-run demographic forecasts, by county, was the main driver of housing demand: more people in a metropolitan area (as residents) create more demand for housing, which creates a derived demand for residential land (a place to put the dwelling units).

[4] This model looks only at housing demand. McClure (this volume) looks at nonresidential demand, which is driven by forecasts of employment growth.

- *Product differentiation*. The analysis considered four categories each of single-family and multifamily housing. The differentiation was done by density (by lot size of single-family units; by persons per acre for multifamily units) to facilitate conversion to demand for land.
- *Market segmentation; individual preferences and constraints*. The analysis addressed key characteristics considered in professional literature and practice to be highly correlated with housing choice: in particular, household size, income and age-of-household-head. The analysis did not include a specific logit specification of individual preference, but did use a simple regression analysis to correlate consumer characteristics with a probability of having a certain housing type.
- *Location*. The locational disaggregation was only to counties: 10 for the Greater Wasatch area. Locational factors typically considered important in explaining housing choice (e.g., school district, travel times to activity centers) were not explicitly considered. Thus, the analysis is useful for evaluating regional implications, but not for neighborhood-level or even municipal-level analysis.
- *Individual preferences and constraints*. The analysis did not model individual choice.
- *Submarkets*. The analysis had no specific modeling of submarkets or the substitutability of one submarket for another. The implicit assumption was that the long-run population that allocated to counties was, again implicitly, considering those tradeoffs. The 10 counties were aggregated into four subareas, but these were very large and were not the type of submarket referred to earlier.
- *Durability of real estate products*. The analysis did not explicitly consider the filtering process. It implicitly allowed for households to shift to other housing types as their characteristics changed.
- *Public policy*. The analysis did not explicitly model the constraints imposed or benefits conferred by public policy. Key public policies, however, were identified and described in the report summary and their implications were considered qualitatively in the final demand allocations.
- *Demand and supply interaction*. The analysis does not model the interactions, but addresses them qualitatively by commenting on, for example, how a public policy, like restricting the supply of land for multifamily housing, could lead to a lower absorption of that type of housing than a baseline demand forecast might suggest.

The analysis simultaneously acknowledged the complexity of the housing market and the need for some type of forecast of future housing demand, as well as for some assessment of the implications of that forecast for regional households and urban form. Such forecasts are inherently uncertain. Their usefulness for public policy often derives more from the explication of their underlying assumptions about the dynamics of markets (demand and supply conditions) and policies than from the specific estimates of future demand.

The analysis then focused on long-run (to 2020) demographic change and new housing. The long-run focus allowed the analysis to ignore short-run events, such as business cycles, changes in interest rates, vacancy rates, lease rates, projects in the pipeline and so on. The assumption was that the region's official long-run population forecast was at least approximately correct. The main steps in the analysis were:

- Define the study area (10 counties centered around Salt Lake City).
- Describe current and forecasted demographic and socioeconomic characteristics that affect the amount and type of housing consumers will demand and the market will build.
- Analyze the current housing market (type of housing existing and being constructed).
- Describe how changing economic and demographic trends are expected to impact the future housing market.
- Identify public policy barriers that prevent the market from meeting current housing demand, and barriers that may prevent the market from meeting future demand.
- Simulate demand by housing types and lot size, by county, to 2020.

The analysis showed that the state's long-run forecast assumed—based on past trends and assumptions about future growth—that almost 70 percent of the population growth in the Greater Wasatch area would be from natural increase (births less deaths), with 30 percent from net migration (more in-migration than out-migration). The relatively high percentage of growth from natural increase (roughly twice the rate of other metropolitan areas on the West Coast) reduced some uncertainty about future housing types (because of relatively strong relationships between household life cycles and housing choices). The analysis also showed that, compared to the entire U.S., the Greater Wasatch area would have a smaller share of households in the retirement phase of their lifetime, and a larger share of young singles, young couples and families.

Given the propensities of these classes of households, one should expect (all else being equal) the regional market to build a larger share of multifamily rental housing, affordable housing for first-time homebuyers and single-family housing for couples with children, than national averages for similar sized regions. Other things being equal, the trend of decreasing household size should increase aggregate demand for housing units and increase demand for smaller single-family housing and for units in multifamily structures (because of lower space needs and less income per household). That is, for a given population increase, more new units will be needed when household size is decreasing, because there are more households.

The analysis examined these and other national and local data (including some rough estimates of buildable, serviceable residential land by county) to determine which key forces would be affecting future housing markets. It concluded:

- *Number of housing units*. The economic forecasts predicted substantial population growth—the Greater Wasatch area would have to provide housing for an additional 363,000 households over the next 20 years (the aggregate demand forecast).
- *Type of housing structures*. The expected growth in income did not necessarily mean households would purchase more large-lot dwellings. The expectation nationally was that the money would go into larger single-family and multifamily units with more amenities but on smaller lots.

The main demographic changes—migration of mobile young adult and elderly households to the West, smaller household size and the increasing average age of the population—all argued for a shift toward smaller units and more multifamily units.

While the large amount of potentially buildable land in the region suggested that land prices would stay relatively low and average lot size could stay relatively high, the analysis noted the possibility that public policy in this area could change for a number of reasons. These include public concerns about sprawl, congestion and natural resources, and increasing fiscal pressures of trying to serve expansive development while also providing infrastructure and maintaining environmental quality.

In response to these forces, the study predicted more planned-unit developments in the future, which could include mixed uses, a mix of housing types, smaller lot sizes for single-family units and overall increases in housing and site amenities.

Housing affordability would continue to be a problem in this region as it is elsewhere. As in the past, the public sector would be unable to supply resources to have much effect on the problem. The expectation was that consumers would be more willing to give up lot size than built space, and would make various choices regarding tradeoffs between built space and amenity. The implication is a shift toward smaller lots, multifamily units and manufactured housing.

- *Housing tenure*. Evidence clearly shows that increasing income and increasing age-of-household-head correlate with increasing home ownership, and that single-family detached homes have been the preferred form of home ownership. The big question for the Wasatch area was whether the economic forecast of increasing average real income would hold, and how that income would be distributed. For example, if real income increases are driven largely by increases in the upper 10 percent of all households, there might be little effect on tenure, as those households already own homes.

These and other conclusions led to a framework for assumptions that were incorporated into a spreadsheet model that simulated future housing demand.[5]

The simulation used the state's household forecasts as a base to calculate the number of new households by county. It then applied an overall residential vacancy assumption to derive the number of new dwelling units (new DU = households/(1 − vacancy rate)). It allocated new housing units by type and tenure. To estimate the percentage of dwelling units that would be expected to be single-family, the researchers conducted a regression analysis using the Public Use Micro-Sample (PUMS) data (U.S. Census 1990). That regression analysis estimated the percentage of housing that would be single-family, based on factors that the study argued are theoretically linked to the choice of housing type, including household size, age-of-household-head, income and number of workers in the household. The regression specification for 1990 explained about 80 percent of the variation in structure type.

The regression model required the distribution of households by size, age-of-household-head and income to predict the ratio of single-family units in 2020. Because income projections were unavailable and household size was only available as an average, the researchers analyzed the relationship between the three variables to develop an expected distribution for 2020 of household sizes and incomes, based on age-of-household-head and average household size.

The simulation illustrated how changes in demographic variables known to influence housing preference might influence housing demand. Implicitly, the simulation assumed the housing market to 2020 will be no more constrained by public policy (in terms of the type, mix and density of housing units) than it was up to 1990.

Ultimately, the study focused on two simulations. The base simulation assumed a continuation of trends observed between 1990 and 1998 in the Greater Wasatch area. The alternative simulation modifies the baseline scenario to account for projected demographic shifts in the population and assumptions about the effect of reasonable changes in public policy.

To make the base simulation, ECONorthwest built a model to forecast the percentage of housing units that would be single-family in 2020, by county, for Utah. The model predicted the share of single-family housing units as a function of the number of households in the county, the shares of households in each of four household size categories, the shares of households in each of four household income categories and the shares of households in each of four age-of-household-head categories.[6] It also differentiated between the 10 most developed counties in Utah and the remaining counties.

The model explained the shares of single-family residences quite well, with the exception of two counties that were not part of the study. The included variables explained 68 percent of the variation between counties. Additional variables probably would improve the model, but the included variables were those for which historical and consistently forecasted 2020 values were available. The model (using the same independent variables) also was estimated to predict the shares of mobile and single-family attached homes.

[5] The study emphasized the word *simulate* rather than forecast, because possible combinations of changes in variables that will affect housing were large. The study explicitly made different assumptions about key variables to illustrate a reasonable range of future demand by housing type. The key issues were whether the future housing market would produce housing in the next 20 years of types and in quantities that looked like today's housing products, or whether it would shift.

[6] Two of the variables in the model were transformed to better match the theoretical specifications of the general linear model. The share of single-family residences was converted to the log-of-the-odds (the natural log of the share, divided by one minus the share) and the number of households in each county was logged.

Developing the model meant, essentially, estimating the coefficients for the independent variables (e.g., household size, income and age-of-household-head), based on historical data (1990 PUMS data). The next step was to insert into the model forecasts of the independent variables (long-run demographic forecasts by state agencies) to obtain a prediction of the independent variable (number of households likely to be residing in single-family dwellings in the forecast year). An implicit assumption of the analysis is that other factors (the health of the national economy or the structure of the local economy) will not vary in ways that would affect housing choices.

In essence, the model just described was the study's attempt to deal with the equation in the "Market Segmentation" section of this chapter: specifically, with the probability variable P. The model is using household type, defined by several measurable variables, to estimate P, which can then be applied to estimates of aggregate households (A) to get demand (D) for product by type as a function of consumer type.

The simulations reported new housing units by housing type and tenure. Single-family housing was further broken down by lot size because land consumption by residential development is a key consideration for defining and evaluating the vision for a change in development patterns being proposed by Envision Utah. Multifamily housing was broken down into four subcategories: duplex, row house, garden apartments (also called walk-up apartments, usually with at-grade parking but having no more than three stories and no elevator), and urban apartments (with more than three stories and including high-rise apartments). Duplex refers to two units in a single structure; row house refers to three or more units in a single structure (also known as townhomes). Duplexes and row houses are distinguished from apartments because units are side-to-side rather than stacked.

Table 6 shows a sample output (for the base simulation), based on trends exhibited in the Greater Wasatch area between 1990 and 1999, primarily as evidenced in building permits for that period.[7] This simulation provided a baseline for the area assuming continuation of past trends.

A continuation of past trends was useful for providing a baseline for analysis, but many factors pointed to a shift in the type of new housing that would be built in the Greater Wasatch area between 2000 and 2020. Key factors included decreasing household size, increasing ages and increasing average real incomes. An alternative simulation used the regression analysis to predict the percentage of new units that would be single-family or mobile-manufactured housing based on expected demo-

Table 6 Base Simulation: New Housing Units by Type and Tenure, 2000–2020, Greater Wasatch Area

Housing Structure Type	Owner-Occupied	Renter-Occupied	Manufactured	Total	Percent of Total
Single-family by lot size (sq. ft.)					
< 5000	15,659		7,263	22,922	6
5,000–9,999	141,487		405	141,892	37
10,000–19,999	94,708		0	94,708	25
> 20,000 (1/2 acre +)	30,739		0	30,739	8
Subtotal	*282,593*		*7,668*	*290,261*	*76*
Multifamily by type					
Duplex	1,590	7,719		9,309	2
Row house	12,871	10,463		23,334	6
Garden apartment	4,432	42,452		46,884	12
Urban	818	11,036		11,854	3
Subtotal	*19,712*	*71,670*		*91,381*	*24*
Total	*302,305*	*71,670*	*7,668*	*381,642*	*100*

Source: ECONorthwest (1999)

[7] The forecast of growth for housing units is slightly greater than the forecast of growth for households because of vacancy rates (considered explicitly in the model) and demolition of existing units (not considered explicitly).

Table 7 Distribution of Housing by Type, Actual and Predicted, Greater Wasatch Area

	Actual			Predicted			
				Base Simulation		Alternative Simulation	
Housing Type	1999 Total (%)	Change 1990–1999 (%)	1999 Total (%)	Change 2000–2020 (%)	2020 Total (%)	Change 2000–2020 (%)	2020 Total (%)
Single family	66	73	67	74	70	59	64
Multifamily	29	25	29	24	27	40	33
Mobile/Manufactured	5	2	4	2	3	1	3

Source: U.S. Census (1990); BEBR (1990–1999); ECONorthwest (2000–2020; 2020)

graphic shifts. The model predicted declining shares of single-family dwellings in most counties (in other words, the percentage of single-family housing built in the next 20 years will be lower than the current (1999) percentage of total housing that single-family housing composes), primarily because of declining average household size and increasing average age-of-household-head. Multifamily dwellings account for nearly 40 percent of new housing built between 2000 and 2020 in the alternative simulation, a significant shift from trends experienced between 1990 and 1999.

Note that the shift in share from single-family to multifamily housing demand was driven entirely by demographics. It was not influenced by assumptions about other factors such as decreased land supply, or increased price of land or public services.

The study was explicit that the simulations were two of many possible futures for housing demand. The intent was to put some reasonable bounds on future demand.

Table 7 shows how the differences were summarized (detailed tables showed eight categories of housing type by county). The main difference between the two simulations is a shift in the composition of new housing development of about 15 percent from single-family units to multifamily units during the 2000–2020 period.

The alternative simulation was driven largely by expected demographic shifts. Other factors, however, could affect the distribution of housing by type and density:

- Public policy can play a key role in housing types and densities through land use designations, capital improvement plans and other policy tools. For example, restrictive zoning policies can reduce the number of multifamily dwellings that are built, or can designate larger overall lot sizes for single-family dwellings. Public policy can also lead to more compact growth and a different housing mix through decisions as simple as eliminating or reducing minimum lot-size constraints where market demand is encouraging developers to build more densely. Other policies are more complicated (politically, if not technically), like minimum density zoning, UGBs or incentives for public housing (e.g., extending the period during which a redevelopment agency can capture tax increments if it provides a certain percentage of housing as part of its redevelopment projects).
- Total land supply did not appear to be a constraining factor in the region for the next 20 years, since there is plenty of land that is not constrained physically (steep slopes, wetlands, floodplains). But other factors (local water supply, public service policies or pubic service costs) could lead to some reductions in the relative availability of buildable, serviceable land, which would in turn increase land prices and housing costs.
- Long-term income trends suggested an increase in real income region-wide. All other things being equal, increases in income mean more single-family dwellings, larger dwellings and higher home ownership rates. But other factors could offset the impact of increased incomes. For example, a recession or real increases in housing cost could eliminate or counter real increases in income. Moreover, expected income increases will not affect all households equally: the region will still have low-income households looking for affordable housing.

- The ability to sustain the expected rate of development over the next 20 years may be affected by air quality, congestion or other environmental constraints. These factors have negatively affected other communities in the U.S., though they did not yet appear to be critical constraints for the Greater Wasatch areas.

These factors were not accounted for in the alternative simulation. The schedule, budget and available data did not call for the development of a model that would incorporate such considerations as independent variables. Thus, the study used a more practical method to address these issues: it used the simulations as a point of departure for discussions among local housing professionals. Of the 23 developers, builders, lenders and realtors who reviewed this report, 83 percent thought the estimates in Table 7 were reasonable. Staff at the Governor's Office of Planning and Budget added some useful but minor changes. Comments from the technical advisory group for this project were included in this report—none changed the estimates in Table 7. A presentation to the steering committee for the Envision Utah project led to discussion of the implications of the findings, but suggested no direction to change the findings.

The housing market analysis played an important role in the regional planning for the Greater Wasatch area. It provided strong evidence that socioeconomic and demographic factors largely beyond the control of typical public policies would be changing the mix of housing from that being demanded and built in current markets. When that information was coupled with other information about land supply, services and likely directions for public policy, most participants in local housing concluded that a shift (not huge, but a shift nonetheless) toward multifamily housing products was likely. Thus, public policy aimed at accommodating such a shift (e.g., the preferred growth alternative being developed by Envision Utah) was reasonable.

Implications for Urban Land Monitoring

This chapter provides a framework for considering the demand for urban land and illustrates one approach that planners in metropolitan areas might apply when attempting to forecast future land consumption. We see several implications for land monitoring.

- Monitoring land supply is only a subset—albeit an important one—of monitoring land markets. Given advances in databases, land use information and GIS technology, it becomes cheaper and easier to obtain a "snap-shot" of current land supply. For example, Metro in Portland has been refining its analysis for 20 years. It now has a relatively accurate picture of where and how much vacant, buildable land exists in the urban area, and the ability to cross-reference that land to a variety of physical, market and policy attributes.

But, monitoring demand has not reached the same level of detail in public agencies, and the prediction of future demand (absorption) is more complicated yet. Monitoring building permits is a step toward demand-side monitoring, and vacancy rates and building absorption are particularly good indicators of shifts in demand. But, as emphasized, price is clearly a key variable. More work must be done to standardize measures of price change for quality-controlled real estate products. One can imagine a public development policy that would bring land into a UGB or trigger infrastructure investment in response to land price increases that exceeded some adopted threshold.

- Given the importance of both demand and supply, their interaction in urban land markets and the complexity of these market interactions, some form of analytical modeling of demand and supply may be useful to inform a monitoring program. To the extent smart growth planning involves asking questions—such as when to extend a UGB or modify growth constraints to avoid unnecessary housing price inflation and housing affordability concerns—there may be a valuable role for modeling. Models such as MetroScope and UrbanSim can potentially be useful in providing more rigorous and unbiased answers to questions about the likely effects of a range of smart growth policies that will cause differing outcomes under different conditions. Failure to account for many aspects of the analytical framework presented here in the assessment of potential effects of many land use and transportation policies may produce inaccurate or biased results.

Ultimately, the rising public concern about urban sprawl provides a tremendous opportunity to overcome some obvious pitfalls of a politically divided debate over the role of planning and markets in enhancing quality of life in America's metropolitan areas. Those who seek to curb the consequences of urban sprawl will have to be sensitive to the difficult challenges of implementing policies that interfere with the deep-rooted principles of local control and individual property rights. Generating a consensus about how metropolitan real estate markets work, and the influence of public choices on these markets, is a step in the direction of finding more fundamental common ground for visions to collectively improve our metropolitan regions. Perhaps the bipartisan surge of concern about sprawl is indeed an opportunity for smart planning, and for smart growth.

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Characteristics, Causes, and Effects of Sprawl: A Literature Review

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Editor's Note: This literature review was prepared for the Florida Department of Community Affairs (DCA) as background to the amendment of Florida's local comprehensive planning rule, Rule 9J-5. On October 2, 1992, DCA published a proposed amendment that, among other things, included criteria for the review of plans to ensure that they discourage urban sprawl. The proposed rule was challenged by the Florida Association of Realtors, Florida Home Builders Association, Florida Farm Bureau Federation, and other parties. DCA settled with most parties, but not all, and the matter went to administrative hearing on September 13, 1993. The hearing officer's order is to be issued momentarily, and if favorable to DCA, will allow DCA to adopt a final rule by early 1994.

The physical characteristics, causes, and effects of sprawl must be understood before sprawl can be effectively regulated. Relying on the literature in the field, this paper provides a conceptual framework against which DCA's proposed sprawl rule can be judged and upon which the final rule can rest.

Classic Sprawl Patterns

Sprawl has been equated to the natural expansion of metropolitan areas as population grows (Sinclair, 1967; Brueckner and Fansler, 1983; Lowry, 1988) and to "haphazard" or unplanned growth, whatever form it may take (Peiser, 1984; Koenig, 1989). More often, though, sprawl is defined in terms of "undesirable" land-use patterns—whether scattered development, leapfrog development (a type of scattered development that assumes a monocentric city), strip or ribbon development, or continuous low-density development. Table 1 indicates which patterns have been labeled sprawl in the technical literature of the past three decades. Scattered development is probably the most common archetype, but any "non-compact" development pattern qualifies.

Using archetypes to define sprawl still leaves us short of a working definition. Like obscenity, the experts may know sprawl when they see it, but that is not good enough for rulemaking. There are two problems with the archetypes.

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Table 1 Sprawl Archetypes

	Low-Density Development	Strip Development	Scattered Development	Leapfrog Development
Whyte (1957)			x	x
Clawson (1962)			x	
Lessinger (1962)	x		x	
Boyce (1963)	x		x	
Harvey and Clark (1965)	x	x		x
Bahl (1968)				x
McKee and Smith (1972)	x	x	x	x
Archer (1973)			x	x
Real Estate Research Corporation (1974)			x	x
Ottensmann (1977)			x	
Popenoe (1979)	x	x	x	
Mills (1981)			x	x
Gordon and Wong (1985)			x	
Fischel (1991)	x			
Heikkila and Peiser (1992)			x	

First, sprawl is a matter of degree. The line between scattered development and so-called polycentric or multinucleated development is a fine one. "At what number of centers polycentrism ceases and sprawl begins is not clear" (Gordon and Wong, 1985, p. 662). Scattered development is classic sprawl; it is inefficient from the standpoints of infrastructure and public service provision, personal travel requirements, and the like. Polycentric development, on the other hand, is more efficient than even compact, centralized development when metropolitan areas grow beyond a certain size threshold (Haines, 1986). A polycentric development pattern permits clustering of land uses to reduce trip lengths without producing the degree of congestion extant in a compact, centralized pattern (Gordon et al., 1989).

Likewise, the line between leapfrog development and economically efficient "discontinuous development" is not always clear. Leapfrogging occurs naturally due to variations in terrain (Harvey and Clark, 1965). New communities nearly always start up just beyond the urban fringe, where large tracts of land are available at moderate cost (Ewing, 1991). Some sites are necessarily bypassed in the course of development, awaiting commercial or higher-density residential uses that will become viable after the surrounding area matures (Ohls and Pines, 1975). "The sprawl of the 1950s is frequently the greatly admired compact urban area of the early 1960s. An important question on sprawl may be, 'How long is required for compaction?' as opposed to whether or not compaction occurs at all" (Harvey and Clark, 1965, p. 6). Whether leapfrog development is inefficient will depend on how much land is bypassed, how long it is withheld, how it is ultimately used, and the nature of leapfrog development (Breslaw, 1990).

Nuances arise with other sprawl archetypes as well. The difference between strip development and other linear patterns (e.g., Mainstreet USA or transit corridor development) is a matter of degree. So, too, the difference between low-density urban development, exurban development and rural residential development. Wherever one draws the line between sprawl and related forms of development will be subject to challenge unless based on an analysis of impacts. It is the impacts of development that render development patterns undesirable, not the patterns themselves.

The second problem with the archetypes is that sprawl has multiple dimensions, which are glossed over in the simple constructs. It is sometimes said that growth management has three dimensions—density, land use, and time. The same is true of sprawl. Leapfrog development is a problem only in the time dimension; in terms of ultimate density and land use, leapfrog development may be relatively efficient. It is known, for example, that infill parcels tend to be developed at higher densities than the land surrounding them (Peiser, 1989; Peiser, 1990). Keep the time frame short and leapfrog development ceases to be inefficient.

Similarly, low-density development is problematic in the density dimension, and strip development in the land-use dimension (since it consists only of commercial uses). If development is clustered at low gross densities, or land uses are mixed in a transportation corridor, these patterns become relatively efficient. Again, it is the impacts of development that render development patterns desirable or undesirable, not the patterns themselves.

Dimensions of Sprawl

Studies analyzing the costs of alternative development patterns have, by necessity, defined alternatives in multiple dimensions, rather than limiting themselves to simple sprawl archetypes. One cannot analyze the costs of "leapfrog development," for example, without specifying densities and land uses. Even the classic anti-sprawl study, *The Costs of Sprawl*, depicts multidimensional community prototypes such as "Sprawl Mix Community," "Low Density Planned Community," and "Low Density Sprawl Community."

Table 2 lists alternative development patterns analyzed in several recent studies. While tailored to their respective regions, all alternatives may be characteriz in two dimensions, density and land use (a third dimension, time, is never considered in these static analyse. Relative to the base case (suburban sprawl), most alternatives involve greater concentration of development and/or greater mixing of uses. Some alternatives stri for a "balanced" mix of jobs and housing by subare others simply for a degree of mixing that does not occur regularly in suburbia. Some concentrate development the urban core(s) of the region, others in a few major suburban centers, and still others in a multitude of small centers.

Poor Accessibility

Ultimately, what distinguishes sprawl from alternative development patterns is poor accessibility of related land uses to one another. The concept of "accessibility is central to urban economics (in simple models of urban form) and travel demand modeling (in gravity-type models of trip distribution). But for some reason, accessibility is hardly ever mentioned in the literature on spraw even though poor accessibility is the "common denominator" among sprawl archetypes.

In scattered or leapfrog development, travelers and service providers must pass vacant land on their way from one developed use to another. In classic strip development, the consumer must pass other commercial uses (usually on crowded arterials) on the way to the desired destination. Of course, in low-density development, everything is far apart due to large private land holdings.

This suggests that sprawl might be characterized generically as any development pattern with poor accessibility among related land uses. Poor accessibility may result from a failure to concentrate development and/or to mix land uses.

The beauty of equating sprawl to poor accessibility is twofold. First, unlike the simple archetypes, the definition recognizes that real-world development patterns are a matter of degree and are multidimensional (a discussed above). No real-world pattern will exactly match an archetype. By defining sprawl generically, we need not debate whether a given pattern is sufficiently similar to an archetype to constitute sprawl.

Second, this generic definition is readily operationalized. Many different accessibility measures are found in the literature (Hansen, 1959; Ingram, 1971 Vickerman, 1974; Burns and Golob, 1976; Dalvi and Martin, 1976; Weibull, 1976; Morris et al., 1979; Pirie 1979; Wachs and Koenig, 1979; Koenig, 1980; Leake and Huzayyin, 1980; Richardson and Young, 1982; Hanson

Table 2 Travel Impacts of Alternative Development Patterns

	Concentrated Jobs	Clustered Housing	Mixed Land-Use	VMT (% change)	Speed (% change)
Washington, D.C.[1]					
– Balanced		○	●	−9	+1
– Concentrated	●	○	●	−9	+2
Seattle, WA[2]					
– Major Centers	●	●		−4	−7
– Multiple Centers	○	○	●	−1	0
– Dispersed Growth			○	+3	0
– Preferred Alternative	●	●	●	−3	+7
Baltimore, MD[3]					
– Centralized Residential	●	○		−1	0
– Decentralized Residential	●			+2	−2
– Transit-Accessible Residential	●	○		−1	−5
Middlesex, NJ[4]					
– Scenario 1	●	●	●	−12	+21
– Scenario 2	○	○	○	−9	+11
Dallas-Fort Worth, TX[5]					
– Rail Corridors	●	●	○	−1	−2
– Activity Centers	●			−1	−1
– Uncongested Areas			○	−4	+2
San Diego, CA[6]					
– Move Jobs			●	−6	NA
– Move Housing			●	−8	NA
– Move Jobs (transit access)			●	−5	NA
– Move Housing (transit access)		○	●	−9	NA
– Job Concentration	●			+11	NA

Key

Jobs	Housing	Land Use
● Large Concentrations	● Highly Clustered	● Balanced Jobs-Housing
○ Small Concentrations	○ Slightly Clustered	○ Mixed Jobs-Housing

[1] Metropolitan Washington Council of Governments (1991)
[2] Puget Sound Council of Governments (1990)
[3] Baltimore Regional Council of Governments (1992)
[4] Middlesex Somerset Mercer Regional Council (1991)
[5] North Central Texas Council of Governments (1990)
[6] San Diego Association of Governments (1991b)

and Schwab, 1987). Simple measures of accessibility, such as average trip length or average travel time can be obtained from travel surveys. More sophisticated measures, such as vehicle miles or vehicle hours of travel (VMT or VHT) per capita, can be estimated with any standard four-step travel modeling system such as FSUTMS (Florida Standard Urban Transportation Model Structure). FSUTMS will even calculate and report “accessibility indexes” for individual traffic zones, using travel time, cost, or distance between zones as the measure of accessibility.

Lack of Functional Open Space

Another characteristic common to all sprawl archetypes is a paucity of functional open space. Strip development presents a solid wall of commercial uses. Low-density suburban development subdivides land until every developable parcel is spoken for; while there is abundant open space, if you count people's yards, it is all in private hands or in holdings too small for community uses. Even leapfrog development, which leaves large areas undeveloped, fails to provide functional open space. The leftover lands are no longer farmed and yet, being in private hands, are unavailable for community uses. Land in suburbia, not actually used for urban purposes, typically is not used at all. It was once estimated that there is about as much idled land in and around cities as there is land used (in any meaningful sense) for urban purposes (Clawson, 1962, p. 107).

As Schneider (1970, p. 69) notes: "It is physically impossible to preserve large open spaces in reasonable proximity to people when millions of people are spread out in uniform low densities. The barrack-like development of land leaves people with the monotony of urban space and form at the scale of the street and private yard." He argues instead for well-designed nodal developments, a complete range of urban spaces (from the most compact to the most open and spacious), and high net densities coupled with low gross densities. The resulting urban form is the antithesis of urban sprawl. Functional open space can be used, then, to define sprawl in much the same way as accessibility. It can be readily operationalized and treated as a matter of degree.

Causes of Sprawl

Low-density suburban development is a "natural" consequence of rising incomes, technological changes, and low travel costs and high travel speeds (for a thorough discussion of the economic forces at work, see Boyce, 1963; Giuliano, 1989). Rising personal income has allowed households to spend more money on travel and on residential space. Industry has shifted from vertical to horizontal production processes, and agglomeration economies have become less important. Increased auto ownership and the construction of high-speed highways have improved the accessibility of outlying sites, causing the urban boundary to shift outwards and flattening land rent and density gradients. Growth and decentralization of population have led to the decentralization of other activities, as market thresholds have been reached at outlying locations.

Leapfrog and scattered development are also a product of market forces (see Clawson, 1962; Bahl, 1968). Expectations of land appreciation at the urban fringe cause some landowners to withhold land from the market. Expectations vary, however, from landowner to landowner, as does the suitability of land for development. The result is a discontinuous pattern of development. The higher the rate of growth in a metropolitan area, the greater the expectations of land appreciation, and the more land will be withheld for future development (Lessinger, 1962; Ottensmann, 1977).

Even strip development results from market forces, albeit forces powerfully shaped by public policy favoring the automobile (Garrison et al., 1959; Boal and Johnson, 1968; Achimore, 1993). "The packaging of 50,000 daily vehicles (and therefore, a total daily population of 60,000 to 70,000 drivers plus passengers) into a single arterial street leads to the irresistible urge to sell things to this population, and creates a sellscape along the street. . .. Once in place, almost no power on earth will stop its march toward strip commercial" (Kulash, 1990).

While a product of market forces, sprawl development patterns are not economically efficient. "[W]e may accept urban sprawl and speculation in raw suburban land as the natural consequences of the economic and social processes we have described, and at the same time we may seek to change one or more stages or bases of those processes because we dislike their final outcome" (Clawson, 1962). In economic theory, a perfectly functioning market requires many buyers and sellers, good information about prices and quality, and no external costs or benefits. The land market meets none of these requirements. The number of buyers and sellers of raw land is limited at any point in time. The rate of land appreciation is speculative. Suburban development is subsidized directly and through the tax code. The land market is rife with externalities. And government regulation of development introduces additional market distortions. See Lee (1979) for an overview of land market imperfections.

In effect, market imperfections define sprawl and provide the justification for public intervention to discourage sprawl.

Subsidies

Raup (1975) lists all manner of subsidies for suburban development. Owner-occupied housing is heavily subsidized through the income tax code. "Although owner-occupied housing is available in central city locations, it seems likely that the net effect of these (tax) provisions has encouraged suburban housing and so contributed to an urban-rural border farther from the center than that which would have obtained in the absence of such a subsidy" (Fischel, 1982).

Infrastructure is also subsidized. While less true today, federal funding of waste treatment systems (and related regulations that led to excess capacity) contributed to the sprawl of the 1960s, 1970s, and early 1980s. "Several important aspects of the current EPA design, review, and funding process raise serious questions concerning the role of EPA-sponsored interceptors in furthering suburban sprawl. In the initial physical design phase of interceptor projects, it appears that current regulations and procedures encourage—or at least permit—the design of unnecessarily large and extensive sewerage systems" (Binkley et al., 1975, p. 2).

Transportation remains heavily subsidized. The magnitude of subsidies for the private automobile is estimated by Renner (1988), Cameron (1991), Voorhees (1991), Hanson (1992a and 1992b), and MacKenzie (1994). Hanson concludes: "The result (of subsidies) is the over-provision of transportation infrastructure relative to what it would be if user fees existed to capture more or all of the direct costs—not to mention externalities—of transportation infrastructure use. . .. Sprawl and discontiguous urban growth are logical outcomes. The glue holding the compact city together has been lost" (Hanson, 1992a, p. 62).

To illustrate, Archer (1973) analyzed a case of leapfrog development in Lexington, Kentucky. By bypassing five tracts of land well-suited for residential development, developers drove up private and public costs by $272,534 per year. Some of these costs were incurred by residents of the outlying development in the form of higher travel costs; they presumably paid less for land and housing than they would have at a more accessible site, in keeping with efficient resource allocation. The remaining costs, however, were defrayed by other consumers and taxpayers in the area, who ended up subsidizing outlying development (see Table 3). Furthermore, the social costs of auto use were

Table 3 Additional Costs of a 200-acre 'Leapfrog' Residential Development Near Lexington, Kentucky

Item	Total Additional Costs per Annum ($)	Who Paid These Additional Costs
Water	8,766	Consumers, Lexington area
Gas	1,013	Consumers, Lexington area
Telephone	13,931	Consumers, statewide
Electricity	937	Consumers, statewide
Sanitary Sewage	9,016	City taxpayers
Refuse Collection	638	City taxpayers
Fire Protection	208	City taxpayers
Police Protection	7,425	City taxpayers
Mail Service	374	Federal taxpayers
School Bus Service	737	County taxpayers
Commercial Delivery Services	54,677	Consumers, Lexington area
Automobile Commuting	172,207	Development's residents
Bus Commuting	2,483	60% by consumers, Lexington area 40% by development's residents
Road and Street Maintenance	122	County taxpayers
Total	272,534	

Source: Archer (1973)

Table 4 Social Costs of Urban Travel (Cents per VMT)

Accidents	4.1–12.5
Air Pollution	1.5–2.5
Water Pollution	0.2
Noise	0.1–0.4
Public Services	0.7–1.1
Congestion	12.5–20.0

Source: DeCorla-Souza (1991)

not even factored into the calculation, though such costs are comparable in magnitude to the direct costs of auto use (see Table 4).

As Archer notes, "[T]he land market will only ensure the efficient use of urban-fringe land if the landowner, developer and homebuyer participants are confronted with the costs and benefits of their respective decisions." They are not so confronted generally in the suburban land market. The result is faster decentralization, lower densities, and more bypassed land than optimal from an economic standpoint.

Externalities

"The use and value of a tract or parcel of land within a metropolis (city center and suburbs) is affected more by the use and value of other tracts or parcels of land than it is by what takes place on the tract itself" (Clawson, 1971, p. 166). Clawson reviews the various externalities and interdependencies that cause suburban land markets to fail.

Particularly relevant to the discussion of sprawl are the external benefits associated with open space. A large body of empirical work shows that buyers are willing to pay more for land if it is close to public open space, whether waterbodies, parks, orpublicly owned greenbelts (Hendon, 1971; Darling, 1973; Weicher and Zerbst, 1973; Hammer et al., 1974; Correll et al., 1978; Pollard, 1982; McLeod, 1984; Didato, 1990; Larsen, 1992). Yet, as a public or quasi-public good, open space tends to be undersupplied by the private sector (Gardner, 1977).

Government Regulation

Local governments have elaborate regulatory systems composed of zoning ordinances, subdivision regulations, and building codes. The common rationale for such regulation is that land markets fail because developers do not take into account the costs (negative externalities) they impose on others in their decisions concerning land use. However, the effect of regulation may be to compound market failure (Richardson and Gordon, 1993). Suburban land-use regulation encourages low-density development and the strict separation of land uses. The result is sprawl (see Moss, 1977).

Costs of Sprawl

A recent assessment of two development plans for the State of New Jersey, one a continuation of current sprawl and the other a more compact pattern, estimated that sprawl would cost an additional \$740 million for roads and \$440 million for water and sewer, would destroy an additional 90,000 acres of prime farmland and almost 30,000 acres of environmental lands; would produce an additional 4,560 tons of water pollutants; and so forth (Center for Urban Policy Research, 1992). The savings with compact development were typically 30 percent or more of the base amounts.

Table 5 Costs of Sprawl

	Psychic Costs	Excess Travel/ Congestion	Energy Costs	Environmental Costs	Inflated Costs of Infrastructure/ Services	Loss of Agriculture Land/Open Space	Downtown Decay
Whyte (1957)					x	x	
Clawson (1962)	x				x	x	
Boyce (1963)							x
Harvey and Clark (1965)		x			x	x	x
Sinclair (1967)						x	
McKee and Smith (1972)		x			x		
Archer (1973)		x			x		
Real Estate Research Corporation (1974)	x	x	x	x	x		
Ottensmann (1977)	x	x	x	x	x	x	
Popenoe (1979)	x						x
Hainca (1986)			x				
Lowry (1988)	x	x					
Neuman (1991)	x				x		

One can quibble with these estimates, but there is no doubt that sprawl carries a large price tag, and not just in monetary terms. Table 5 itemizes the costs of sprawl, as posited in the literature. Several are highlighted below.

Psychic Costs

In the article, "Urban Sprawl: Some Neglected Sociological Considerations," Popenoe (1979) identifies sprawl with two types of psychic costs: environmental deprivation and deprivation of access. The former he defines as a deficiency of elements that provide activity and stimulation. The visual uniformity, the lack of identifiable community, and other unaesthetic qualities of sprawl may all contribute to environmental deprivation.

While it may appear self-evident that sprawl is unaesthetic, the reaction of the American public is less clear. Surveys reveal that people, when shown images of both sprawl and traditional communities, favor the latter by wide margins (Neuman, 1991). Yet, some of the hallmarks of sprawl are apparently to the public's liking. By a margin of 66 to 34 percent, people favor "homogeneous neighborhoods" over "mixed neighborhoods where different types and sizes of houses are in the same general area and where small stores and other commercial activities are nearby" (Bookout, 1992). Survey research in Florida demonstrates a distinct preference for low-density suburban or exurban living (Audirac et al., 1989).

The other psychic cost of sprawl, deprivation of access, is much clearer. In a sprawl development pattern, nearly half of the American public, those who cannot drive, are deprived of good access to community facilities and services. "[I]t is hard to escape the conclusion that urban sprawl is an urban development pattern designed by and for men, especially middle class men. . .. Fewer women than men are able, or wish, to drive a car, and if a family has just one car, that car traditionally is used by the man. No teenager can legally drive a car until a certain minimum age, usually sixteen or seventeen. And automobile driving is often impossible or economically difficult for the elderly, the poor, and the handicapped" (Popenoe, 1979, pp. 262–263). The impacts of poor accessibility on these groups have been well-documented (Schaeffer and Sclar, 1975; Popenoe, 1977; Berg and Medrich, 1980; Carp, 1980; Millas, 1980; Carp, 1988).

Excess Travel and Congestion

Sprawl has an impact on travel demand and traffic congestion. These two performance criteria are not equivalent. All else being equal, more travel will translate into more congestion. But all else is not equal with alternative development patterns; the level of roadway congestion depends on where travel occurs as well as how much occurs.

The extensive literature relating travel and traffic to urban form is reviewed by Gilbert and Dajani (1974), U.S. Department of Transportation (1976), Neels et al. (1977), Giuliano (1989), Holtzclaw (1991), Cervero (1991a), Downs (1992), and Steiner (1994). High densities generate fewer vehicle miles of travel (VMT) per capita than do low densities (as in Figure 1). Trips become shorter as densities rise, and a growing percentage of trips are made by walking ortransit (Levinson and Wynn, 1963; Bellomo et al., 1970; Guest and Cluett, 1976; Pushkarev and Zupan, 1977; Smith, 1986; Newman and Kenworthy, 1989; Spillar and Rutherford, 1990; Prevedouros, 1991; Dunphy and Fisher, 1994). At the same time, high densities are associated with high levels of traffic congestion (Neels et al., 1977; Newman and Kenworthy, 1988). The net effect of shorter trips and heightened congestion on travel times and travel costs is unclear *a priori*, but recent empirical evidence suggests that the former overwhelms the latter (Prevedouros, 1991; Ewing, 1994).

The picture is further complicated by the multiple dimensions of development. Urban and suburban centers may be large or small and may have single or mixed uses. These variables may have more impact on travel and traffic than do densities per se.

Mixing complementary land uses reduces trip lengths and encourages alternatives to the automobile. Indeed, since dense urban environments tend to be mixed-use environments, the positive impacts attributed to density in the literature may result as much from mixed uses. Cities with the highest rates of walking/bicycling are those with a balance of jobs and residents in their central cities (like Boston); high employment densities alone do not foster alternatives to the automobile (Newman and Kenworthy, 1989, p. 47). Jobs/housing balance is the best predictor of trip length in the San Diego region; trip lengths average 8.8 miles for commuters living in balanced areas,

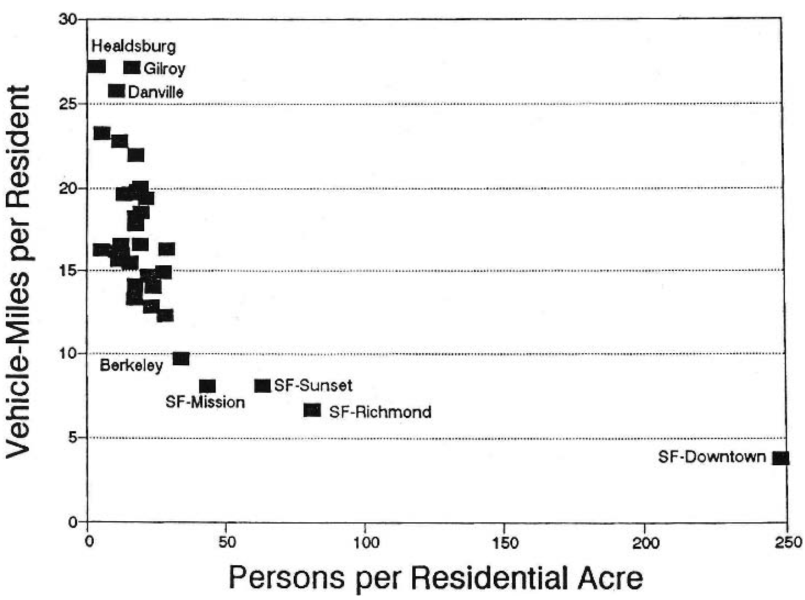

Fig. 1 Residential Density Versus VMT Per Resident
Source: Harvey and Deakin (1991)

two miles less than the regional average and four miles less than areas with an excess of housing over jobs (San Diego Association of Governments, 1991a and 1991b). Transit and walking account for bigger shares of Seattle's work and shopping trips in census tracts with mixed uses; this is true even controlling for densities and demographic factors (Frank, 1994). A balance of jobs and housing increases the amount of walking and biking to suburban centers, and reduces the level of congestion on nearby freeways (Cervero, 1989). The addition of retail uses to office-oriented centers also increases the amount of walking, biking, ridesharing, and transit uses (Cervero, 1988; Cervero, 1991b).

How to break out the effects of different development variables on travel and traffic? Alternative development patterns have been analyzed for their travel and traffic impacts in many computer simulation studies (for a review of studies, see Gilbert and Dajani, 1974; Giuliano, 1989; DeCorla-Souza, 1992). We focus here on recent simulation studies because they have achieved greater realism in the depiction of alternative development patterns.

For each of six recent studies, Table 2 compares alternative development patterns to a "base case." The base case is a continuation of current sprawl patterns. The alternatives are typically more concentrated in their development patterns and/or more mixed in their land uses than is the base case. For each alternative, percentage changes in VMT (reflecting travel demand) and average peak-period travel speed (reflecting traffic congestion) are shown relative to the base case. The alternatives that perform well in both dimensions are those with mixed land uses. Simply concentrating development, without mixing uses, reduces VMT in some cases at the expense of travel speed. (Note, though, that none of the simulations allowed for significant mode shifts to transit as densities rose.)

Energy Inefficiency and Air Pollution

Higher densities mean shorter trips and more travel by energy-conserving modes. They also mean more congestion. In a paper entitled, "The Transport Energy Trade-Off: Fuel-Efficient Traffic versus Fuel-Efficient Cities," Newman and Kenworthy (1988) find that the former effect overwhelms the latter. Even though vehicles are not as fuel-efficient in dense areas due to traffic congestion, fuel consumption per capita is still substantially less because people drive so much less. On balance, fuel consumption per capita declines by one-half to two-thirds as city densities rise from four to 12 persons per acre (or roughly, 10 to 30 persons per hectare) (see Figure 2).

As with travel, the relationship of fuel consumption to urban form is complicated by the multiple dimensions of development. Studies comparing centralized development to low-density sprawl consistently find the former to be more energy-efficient. But when polycentric development is included in the comparison, that pattern emerges as the preferred alternative from an energy-efficiency standpoint (see Table 6).

Air pollution from mobile sources is less affected by urban form than is energy consumption. As with energy consumption, vehicle emissions are directly related to VMT and inversely related to vehicle speed (up to about 40 mph). But the emissions associated with "cold starts" and "hot soaks" add another element. In an early simulation study, extreme changes in development patterns within the Denver region were found to have "little or no effect on ambient air quality" (Scheuemstuhl and May, 1979). A recent study found that while balancing jobs and housing in the San Diego region would reduce VMT by 5 to 9 percent, and reduce congested miles of freeway by 31 to 41 percent, it would cut vehicle emissions by less than 2 percent. "[B]alancing jobs and housing does not cause a significant change in vehicle trips. A major part of auto pollution comes from cold starts which are directly related to vehicle trips (as opposed to VMT or congestion)" (San Diego Association of Governments, 1991b).

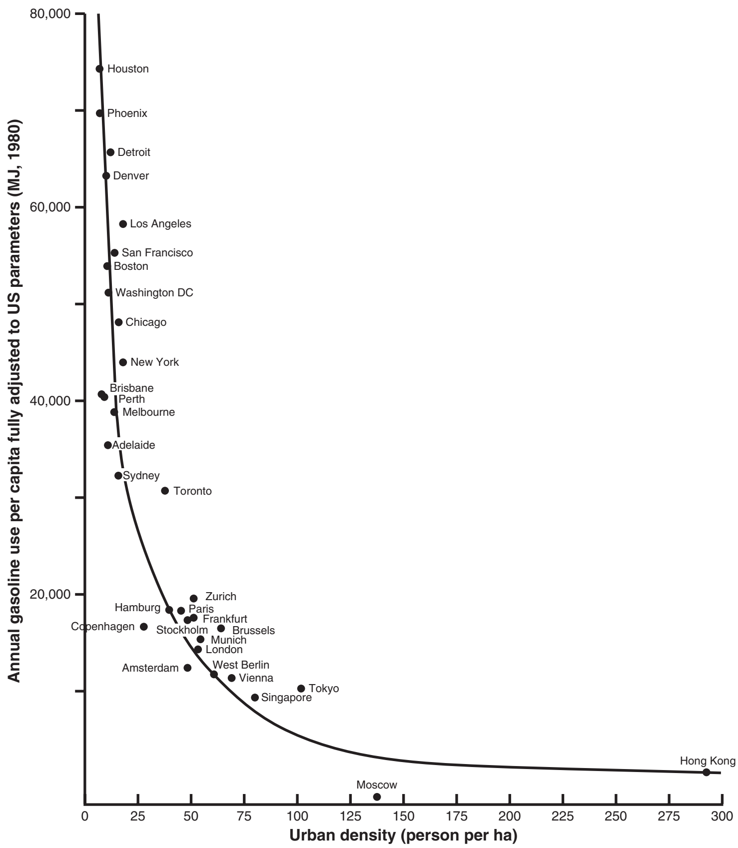

Fig. 2 Urban Density Versus Gasoline Use per Capita Adjusted for Vehicle Efficiency
Source: Newman and Kenworthy (1989)

Inflated Infrastructure and Public Service Costs

The Costs of Sprawl and other studies have shown that neighborhood infrastructure becomes less costly on a per-unit basis as density rises (for a review of literature, see Priest et al., 1977; Frank, 1989). Just how much costs decline with density has been debated, with some arguing that cost differences largely evaporate when one adjusts for dwelling unit size. Also debated has been the shape of the cost curve, with some contending that costs rise at very high densities (due to the special needs of high-rise structures) and fall at very low densities (due to the feasibility of septic systems, natural drainage, and rural street cross sections). But the basic inverse relationship between density and neighborhood infrastructure costs is well-established.

Having said this, it turns out that neighborhood infrastructure costs are not particularly relevant to the issue of sprawl. As long as developers are responsible for the full costs of neighborhood infrastructure, and pass such costs on to homebuyers and other end-users of land, lower-density development patterns will meet the test of economic efficiency (at least with respect to infrastructure costs).

Table 6 Conclusions of Land Use Studies Concerning the Most Energy-Efficient Urban Form

Bacon (1973)	centralization	
Franklin (1974)	centralization	
Real Estate Research Corporation (1974)	centralization	
Council on Envirn. Quality (1975)	centralization	
Hoben (1975)	centralization	
Nadler (1975)	centralization	
Kydes et al. (1976)		multinucleation
Slayton (1976)		multinucleation
Cornehls (1977)	centralization	
Burby and Bell (1978)		multinucleation
Carroll (1978)	centralization	
Edwards (1978)		multinucleation
Roberts (1978)	centralization	
Romanos (1978)		multinucleation
Soot and Sen (1979)		multinucleation
Windsor (1979)		multinucleation
Small (1980)		multinucleation
Van Til (1980, 1982)		multinucleation
Keyes (1981)		multinucleation

Source: Haines (1986); full citations for these studies can be found in Haines.

Where inefficiency is more likely to arise is in the provision of community-level infrastructure. Inefficiency may also arise in the operation and maintenance of infrastructure, and in the provision of police and other public services. These tend to be financed with local taxes or user fees that are independent of location, causing remote development to be subsidized.

"[C]osts beyond the neighborhood level are not fully passed onto the consumer as part of buying a house. . .. The costs associated with distance from central facilities, although potentially larger than many of the other costs at distances of 20 miles, are almost completely ignored in pricing schemes like impact fees. The inevitable results of both of these factors are to stimulate overconsumption of housing in costly-to-serve circumstances and to subsidize the more costly locations with the less costly ones" (Frank, 1989, p. 42).

From the standpoint of community-level infrastructure, costs vary not so much with residential density but with the degree of clustering and/or proximity to existing development (Howard County Planning Commission, 1967; Stone, 1973; Real Estate Research Corporation, 1974; Barton-Aschman Associates, 1975; Downing and Gustely, 1977; Peiser, 1984). So, too, the costs of public services (Archer, 1973; Real Estate Research Corporation, 1974; Downing and Gustely, 1977; Peiser, 1984). See, for example, Table 7.

Table 7 Annual Cost of Providing Public Services per Mile Distance from Public Facility Site: 1973

	Capital* or Operating** Costs per Mile ($)
Police	438*
Fire	216*
Sanitation	3,360***
Schools	19,845***
Water supply	21,560**
Storm drainage	6,187**
Sanitary sewers	12,179**
Total Cost	68,498

* Includes only operating costs
** Includes only capital costs
*** Includes both capital and operating costs
Source: Downing and Gustely (1977)

Loss of Farmland

Urbanization generally, and sprawl in particular, contribute to loss of farmlands and open spaces. Because lands most suitable for growing crops also tend to be most suitable for "growing houses" (being flat and historically near human settlements), a disproportionate amount of prime farmland is lost to urbanization. For estimates of losses, see Berry and Plaut, 1978; Fischel, 1982; Nelson, 1990.

Having acknowledged the relationship between urbanization and loss of farmland, we must still ask whether (1) the loss is in some sense peculiar to sprawl-type development, and (2) the loss is a result of some market failure demanding public intervention.

Insofar as sprawl consumes more land in toto than does compact development, sprawl may cause the loss of more agricultural lands and open spaces. In *The Costs of Sprawl*, a community prototype representing low-density sprawl leaves no land in its natural (unimproved) state (Real Estate Research Corporation, 1974, Table 43). In contrast, planned prototypes leave anywhere from 18 to 57 percent unimproved land by means of clustered, contiguous and/or higher-density development.

The literature offers differing views on the extent to which rural/urban land conversion results from market imperfections, and on the concomitant need for public intervention to preserve farmlands. Arguing that the problem is illusory or at least overstated are Gardner (1977) and Fischel (1982). On the other side of the issue are Raup (1975) and Nelson (1990).

There is evidence of market failure in three areas:

(1) Urban "spillover effects" (externalities) that make nearby farming operations less profitable and cause farmers to disinvest; such effects may extend up to three miles from urban development (Nelson, 1986).
(2) The "impermanence syndrome" that causes farmers to abandon operations prematurely in anticipation of urban development; perhaps as much as one acre is idled for every acre converted to urban uses in the Northeastern United States (Plaut, 1976; also see Sinclair, 1967).
(3) Misjudgment of the value of farmland to future generations (Peterson and Yampolsky, 1975).

Other market imperfections that accelerate farmland conversion are discussed in Nelson (1992).

Conclusion

With a few refinements, the list of "sprawl indicators" in DCA's proposed rule can be brought into harmony with the technical literature on sprawl of the past three decades. DCA's basic approach is sound. It has defined a variety of sprawl indicators, some relating to *characteristics* of sprawl, others to *effects* of sprawl, and still others to *causes* of sprawl. In essence, the rule calls for policies to avoid the characteristics of sprawl, remove the causes, and mitigate the effects.

It might seem like overkill to attack sprawl on three fronts; certainly, it makes the rule lengthier and more cumbersome to apply than some might wish. Yet, market forces and public-policy biases favoring sprawl are strong and pervasive. The three-pronged approach in the proposed rule may be more successful than, for example, simply asking new development to pay 100 percent of the costs of public facilities and services it requires or, alternatively, drawing an urban service boundary on the future land use map in the hope that development within the boundary will be at reasonable density and entail a mix of uses.

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Urban Ecological Footprints: Why Cities Cannot be Sustainable—and Why They are a Key to Sustainability

William Rees and Mathis Wackernagel

Keywords: ecological footprint · sustainability · economics · resource use · energy · global trade · environmental justice

Introduction: Transforming Human Ecology

It is sometimes said that the industrial revolution stimulated the greatest human migration in history. This migration swept first through Australia, Europe, and North America and is still in the process of transforming Asia and the rest of the world. We refer, of course, to the mass movement of people from farms and rural villages to cities everywhere. The seeming abandonment of the countryside is creating an urban world—75% or more of the people in so-called industrialized countries now live in towns and cities, and half of humanity will be city dwellers by the end of the century.

Although usually seen as an economic or demographic phenomenon, urbanization also represents a human ecological transformation. Understanding the dramatic shift in human spatial and material relationships with the rest of nature is a key to sustainability. Our primary purpose, therefore, is to describe a novel approach to assessing the ecological role of cities and to estimate the scale of the impact they are having on the ecosphere. The analysis shows, that as nodes of energy and material consumption, cities are causally linked to accelerating global ecological decline and are not by themselves sustainable. At the same time, cities and their inhabitants can play a major role in helping to achieve global sustainability.

Starting Premise

Our analysis starts from the premise that the late 20th century marks a nontrivial turning point in the ecological history of human civilization. For the first time, since the dawn of agriculture and the possibility of geographically fixed settlements 12,000 years ago, the aggregate scale of human economic activity is capable of altering global biophysical systems and processes in ways that jeopardize both global ecological stability and geopolitical security.

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Examples abound—more artificial nitrate is now applied to the world's croplands than is fixed from the atmosphere by microbial activity and other natural processes combined (Vitousek 1994); the rate of human-induced species extinctions is approaching the extinction rates driven by "the great natural catastrophes at the end of the Paleozoic and Mesozoic era—in other words, [they are] the most extreme in the past 65 million years" (Wilson 1988); "residuals" discharged by industrial economies are depleting stratospheric ozone and altering the preindustrial composition of the atmosphere, and both these trends contribute to (among other things) the threat of climate change, itself the most potent popular symbol of widespread ecological dysfunction. Perhaps most significant from an ecosystems perspective is the evidence that human beings, one species among millions, now consume, divert, or otherwise appropriate for their own purposes 40% of the product of net terrestrial photosynthesis (Vitousek et al. 1986) and up to 35% of primary production from coastal shelves and upwellings, the most productive marine habitats (Pauly and Christensen 1995). Were it not for the fact that fish catches are in decline from stock depletion, both these proportions would be steadily increasing.

The empirical evidence suggests that the human economy is overwhelming the ecosphere from within. This unprecedented situation has prompted some development analysts to argue that the world has reached an historic turning point, a point at which the world must shift from the assumptions of "empty-world" to those of "full-world" economics (Daly 1991).

Carrying Capacity as Maximum Human "Load"

An environment's carrying capacity is its maximum persistently supportable load (Catton 1986).

The notion that humanity may be up against a new kind of limit has rekindled the Malthusian debate about human carrying capacity (see, for example, *Ecological Economics*. November 1995. 15(2)). Carrying capacity is usually defined as the maximum population of a given species that can be supported indefinitely in a defined habitat without permanently impairing the productivity of that habitat. However, because we humans seem to be capable of continuously increasing the human carrying capacity of Earth by eliminating competing species, by importing locally scarce resources, and through technology, conventional economists and planners generally reject the concept as applied to people. As Herman Daly critically observes, the prevailing vision assumes a world in which the economy floats free of any environmental constraints. This is a world "in which carrying capacity is infinitely expandable"—and therefore irrelevant (Daly 1986).

By contrast, we argue that the economy is an inextricably embedded subsystem of the ecosphere. Despite our technological and economic achievements, humankind remains in a state of "obligate dependence" on the productivity and life support services of the ecosphere (Rees 1990). The trappings of technology and culture aside, human beings remain biophysical entities. From a trophic-dynamic perspective, the relationship of humankind to the rest of the ecosphere is similar to those of thousands of other consumer species with which we share the planet. We depend for both basic needs and the production of cultural artifacts on energy and material resources extracted from nature, and *all* this energy/matter is eventually returned in degraded form to the ecosphere as waste. The major material difference between humans and other species is that, in addition to our biological metabolism, the human enterprise is characterized by an industrial metabolism. In thermodynamic terms, all our toys and tools (the human-made "capital" of economists) are "the exosomatic equivalent of organs" and, like bodily organs, require continuous flows of energy and material to and from "the environment" for their production and operation (Sterrer 1993). Carrying capacity therefore remains central to sustainabilty.

Because of this continuing functional dependence on ecological processes, some analysts have stopped thinking of natural resources as mere "free goods of nature." Ecological economists now regard the species, ecosystems, and other biophysical entities that produce required resource flows as forms of "natural capital" and the flows themselves as types of essential "natural income" (Pearce et al. 1989; Victor 1991; Costanza and Daly 1992). This capital theory approach provides a valuable insight into the meaning of sustainability—no development path is sustainable if it depends on the continuous depletion of productive capital. From this perspective, society can be said to be economically sustainable only if it passes on an undiminished per capita stock of essential capital from one generation to the next (Pearce 1994; Solow 1986; Victor 1991).[1]

In the present context, the most relevant interpretation of this "constant capital stocks" criterion is as follows:

> Each generation should inherit an adequate per capita stock of natural capital assets no less than the stock of such assets inherited by the previous generation.[2]

Because of its emphasis on maintaining natural (biophysical) capital intact, the foregoing is a "strong sustainability" criterion (Daly 1990). The prevailing alternative interpretation would maintain a constant *aggregate* stock of humanmade and natural assets. This latter version reflects the neoclassical premise that manufactured capital can substitute for natural capital and is referred to as "weak sustainability" (Daly 1990; Pearce and Atkinson 1993; Victor et al. 1995).

We prefer strong sustainability because it best reflects known ecological principles and the *multifunctionality* of biological resources "including their role as life support systems" (Pearce et al. 1989). Most importantly, strong sustainability recognizes that manufactured and natural capital "are really not substitutes but complements in most production functions" (Daly 1990). In other words, many forms of biophysical capital perform critical functions that cannot be replaced by technology. For sustainability, a critical minimal amount of such capital must be conserved intact and in place. This will ensure that the ecosystems upon which humans depend remain capable of continuous self-organization and production.[3]

In this light, the fundamental question for ecological sustainability is whether remaining *natural* capital stocks (including other species populations and ecosystems) are adequate to provide the resources consumed and assimilate the wastes produced by the anticipated human population into the next century, while simultaneously maintaining the general life support functions of the ecosphere (Rees 1996). In short, is there adequate human carrying capacity? At present, of course, both the human population and average consumption are increasing, whereas the total area of productive land and stocks of natural capital are fixed or in decline. In this light, we argue that shrinking carrying capacity may soon become the single most important issue confronting humanity.

The issue becomes clearer if we define human carrying capacity not as a maximum population but rather as the maximum (entropic) "load" that can safely be imposed on the environment by people

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[1] We acknowledge that the heterogeneity and interdependence of various forms of natural capital make this criterion difficult to operationalize. For example, ecosystems are constantly developing and evolving, and there are many combinations of natural capital stocks that could be sustainable. However, this does not detract from the general principle that for each potentially viable combination, sustainability requires some minimal individual and aggregate quantity of these component stocks.

[2] "Natural assets" encompasses not only material resources (e.g., petroleum, the ozone layer, forests, soils) but also process resources (e.g., waste assimilation, photosynthesis, soils formation). It includes renewable as well as exhaustible forms of natural capital (Costanza and Daly 1992). Our primary interest here is in essential renewable and replenishable forms. Note that the depletion of nonrenewables could be compensated for through investment in renewable natural capital.

[3] The only ecologically meaningful interpretation of constant stocks is in terms of constant *physical* stocks as is implied here. However, some economists interpret "constant capital stock" to mean constant monetary value of stocks or constant resource income over time (for a variation on this theme, see Pearce and Atkinson 1993). These interpretations allow declining physical stocks as value and market prices rise over time.

(Catton 1986). Human load is clearly a function not only of population but also of average per capita consumption. Significantly, the latter is increasing even more rapidly than the former due (ironically) to expanding trade, advancing technology, and rising incomes. As Catton (1986) observes: “The world is being required to accommodate not just more people, but effectively ‘larger’ people . . ..” For example, in 1790 the estimated average daily energy consumption by Americans was 11,000 kcal per capita. By 1980, this had increased almost 20-fold to 210,000 kcal/day (Catton 1986). As a result of such trends, *load* pressure relative to carrying capacity is rising much faster than is implied by mere population increases.

Ecological Footprints: Measuring Human Load

By inverting the standard carrying capacity ratio and extending the concept of load, we have developed a powerful tool for assessing human carrying capacity. Rather than asking what population a particular region can support sustainably, the critical question becomes: How large an area of productive land is needed to sustain a defined population indefinitely, *wherever on Earth that land is located?* (Rees 1992; Rees and Wackernagel 1994). Most importantly, this approach overcomes any objection to the concept of human carrying capacity based on trade and technological factors. In the language of the previous section, we ask how much of the Earth’s surface is appropriated to support the “load” imposed by a referent population, whatever its dependence on trade or its level of technological sophistication.

Since most forms of natural income (resource and service flows) are produced by terrestrial ecosystems and associated aquatic ones,[4] it should be possible to estimate the area of land/water required to produce sustainably the quantity of any resource or ecological service used by a defined population or economy at a given level of technology. The sum of such calculations for all significant categories of consumption would provide a conservative area-based estimate of the natural capital requirements for that population or economy. We call this area the population’s true “ecological footprint.”

A simple two-step mental experiment serves to illustrate the ecological principles behind this approach. First, imagine what would happen to any modern city as defined by its political boundaries if it were enclosed in a glass or plastic hemisphere completely closed to material flows. This means that the human system so contained would be able to depend only on whatever remnant ecosystems were initially trapped within the hemisphere.

It is obvious to most people that the city would cease to function, and its inhabitants would perish within a few days. The population and economy contained by the capsule would have been cut off from both vital resources and essential waste sinks leaving it to starve and suffocate at the same time. In other words, the ecosystems contained within our imaginary human terrarium—and any real world city—would have insufficient carrying capacity to service the ecological load imposed by the contained population.

The second step pushes us to contemplate urban ecological reality in more concrete terms. Let’s assume that our experimental city is surrounded by a diverse landscape in which cropland and pasture, forests and water-sheds—all the different ecologically productive land-types—are represented in proportion to their actual abundance on the Earth and that adequate fossil energy is available to support current levels of consumption using prevailing technology. Let’s also assume our imaginary glass enclosure is elastically expandable. The question now becomes: How large would the hemisphere have to grow before the city at its center could sustain itself indefinitely and exclusively on

[4] Exceptions include the ozone layer and the hydrologic cycle both of which are purely physical forms of natural capital.

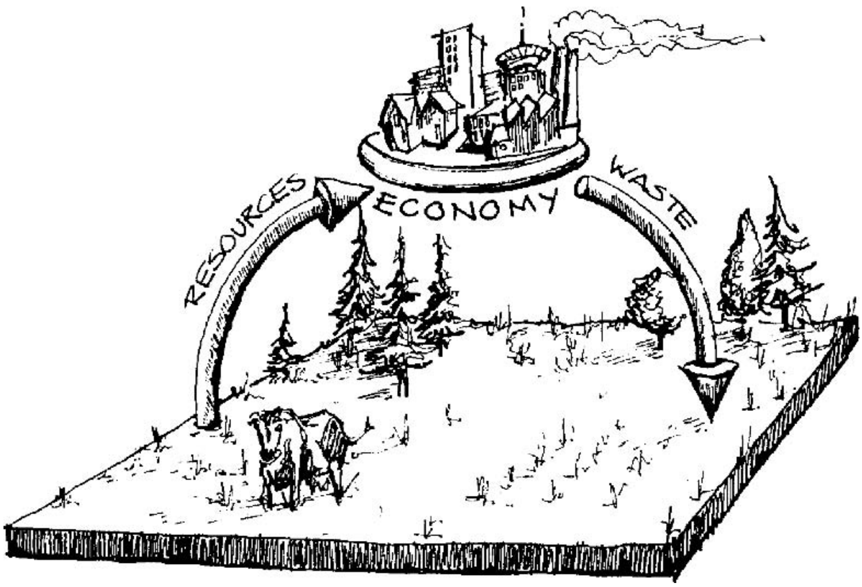

Fig. 1 What is an ecological footprint? Think of a city as having an "industrial metabolism." In this respect, it can be compared to a large animal grazing in its pasture. Just like the beast, the city consumes resources and all this energy and matter eventually passes through to the environment again. Thus, the footprint question becomes: "How large a pasture is necessary to support that city indefinitely—to produce all its 'feed' and to assimilate all its wastes sustainably" (*Source:* Wackernagel and Rees 1995)

the land and water ecosystems and the energy resources contained within the capsule?[5] In other words, what is the total area of different ecosystem types needed continuously to supply the material demands of the people of our city as they go about their daily activities (Fig. 1)?

Answering this question would provide an estimate of the de facto ecological footprint of the city. Formally defined, the ecological footprint (EF) is the total area of productive land and water required continuously to produce all the resources consumed and to assimilate all the wastes produced, by a defined population, wherever on Earth that land is located. As noted, the ecological footprint is a land-based surrogate measure of the population's demands on natural capital.

Method in Brief

The basic calculations for ecological footprint estimates are conceptually simple. First we estimate the annual *per capita* consumption of major consumption items from aggregate regional or national data by dividing total consumption by population size. Much of the data needed for preliminary assessments is readily available from national statistical tables on, for example, energy, food, or forest products production and consumption. For many categories, national statistics provide both production and trade figures from which trade-corrected consumption can be assessed:

$$trade - corrected\ consumption = production + imports - exports$$

[5] For simplicity's sake, the question as posed does not include the ecologically productive land area needed to support other species independent of any service they may provide to humans.

The next step is to estimate the land area appropriated per capita for the production of each consumption item by dividing average annual consumption of that item by its average annual productivity or yield.[6] Box 1 provides a sample calculation showing the land requirement for paper consumption by the average Canadian. A similar calculation can be made for the land required to assimilate certain individual waste products such as carbon dioxide.

Box 1. Productive Forest Area Required for Paper Production
Question: How much productive forest is dedicated to providing pulp-wood for the paper used by the average Canadian? ("Paper" includes food wrappings, other packaging, reading material and construction paper.)
Answer: Each Canadian consumes about 244 kilograms of paper products each year. In addition to the recycled paper that enters the process, the production of each metric ton of paper in Canada currently requires $1.8\,\mathrm{m}^3$ of wood. For Ecological Footprint analyses an average wood productivity of 2.3 [m^3/ha/yr] is assumed. Therefore, the average Canadian requires . . .

$$\frac{244[\mathrm{kg/cap/yr}] \times 1.8[\mathrm{m}^3/\mathrm{t}]}{1,000[\mathrm{kg/t}] \times 2.3[\mathrm{m}^3/\mathrm{ha/yr}]} = \text{0.19 [ha/capita] of forest in continuous production of paper.}$$

We then compile the total average per capita ecological footprint (ef) by summing all the ecosystem areas appropriated by an individual to fill his/her annual shopping basket of consumption goods and services.

Finally we obtain the ecological footprint ($\mathrm{EF_P}$) of the study population by multiplying the average per capita footprint by population size (N): Thus, $\mathrm{EF_P} = \mathrm{N} \times \mathrm{ef}$.

Our EF equation is structually similar to the more familiar representation of human environmental impact (I) as a product of population (P), affluence (A), and technology (T), I = PAT (Ehrlich and Holdren 1971; Holdren and Ehrlich 1974). The ecological footprint is, in fact, a measure of population impact expressed in terms of appropriated land area. The size of the per capita footprint will of course, reflect the affluence (material consumption) and technological sophistication of the subject population.

So far our EF calculations are based on five major categories of consumption—food, housing, transportation, consumer goods and services—and on eight major land-use categories. However, we have examined only one class of waste flow in detail. We account for carbon dioxide emissions from fossil energy consumption by estimating the area of average carbon-sink forest that would be required to sequester them [carbon emissions/capita]/[carbon assimilation/hectare]), on the assumption that atmospheric stability is a prerequisite of sustainability. (Ours is a relatively conservative approach. An alternative is to estimate the area of land required to produce the biomass energy equivalent [ethanol] of fossil energy consumption. Because of thermodynamic losses, this produces a much larger energy footprint than the carbon assimilation method.) Full details of EF calculation procedures and more examples can be found in Rees and Wackernagel 1994; Wackernagel and Rees 1995; and Rees 1996.

[6] We generally use world average productivities for this step in ecological footprint calculations. This is a reasonable first approximation, particularly for trade-dependent urban regions importing ecological goods and services from all over the world. Local productivities are necessary, however, to calculate actual local/regional carrying capacity.

Strengths and Limitations of Footprint Analysis

The major strength of ecological footprint analysis is its conceptual simplicity. Our method provides an intuitive and visually graphic tool for communicating one of the most important dimensions of the sustainability dilemma. It aggregates the ecological flows associated with consumption and translates them into appropriated land area, an indicator that anyone can understand. The ecological footprint of any defined population can then be compared with the available supply of productive land. Individuals can contrast their personal footprints with their ecological "fair Earthshares," national footprints can be compared to domestic territories, and the aggregate human footprint can be compared to the productive capacity of the entire planet.

In cases where the ecological footprint is significantly larger than a secure supply of productive land, the difference represents a "sustainability gap" and "ecological deficit" (Rees 1996). This is the amount by which consumption (or the measurable impact of consumption) must be reduced for longterm ecological sustainability. Thus, unlike ordinary measures of total resource use, ecological footprint analysis provides secondary indices that can be used as policy targets. The question then becomes: How large is our ecological deficit and what must be done to reduce it? (We should point out that humanity's ecological deficit may be far more important the fiscal deficit, yet the former is totally ignored in the current frenzy to reduce the latter in many countries.)

Although acknowledging its power to communicate a fundamental message, some commentators have suggested that the footprint concept is too simplistic. For example, the model is static, whereas both nature and the economy are dynamic systems. Ecological footprinting therefore cannot directly take into account such things as technological change or the adaptability of social systems.

It is true, of course, that footprint analysis is not dynamic modeling and has no predictive capability. However, prediction was never our intent. Ecological footprinting acts, in effect, as an ecological camera—each analysis provides a snapshot of our current demands on nature, a portrait of how things stand *right now* under prevailing technology and social values. We believe that this in itself is an important contribution. We show that humanity has exceeded carrying capacity and that some people contribute significantly more to this ecological "overshoot" than do others. Ecological footprinting also estimates how much we have to reduce our consumption, improve our technology, or change our behavior to achieve sustainability.

Moreover, if used in a time-series study (repeated analytic "snap-shots" over years or decades) ecological footprinting can help monitor progress toward closing the sustainability gap as new technlogies are introduced and consumer behavior changes. (After all, even a motion picture is a series of snap-shots.) Footprint analysis can also be used in static simulation studies to test, for example, the effect of alternative technologies or settlement patterns on the size of a population's ecological footprint (see Walker 1995, for an example). To reiterate, ecological footprint analysis is not a window on the future, but rather a way to help assess both current reality and alternative "what if" scenarios on the road to sustainability.

A more substantive criticism of ecological footprinting is that it ignores many other factors at the heart of sustainability. It is certainly true that the ecological footprint does not tell the entire sustainability story—indeed, any single index can be misleading (consider the problems with GDP!). In fact, our calculations to date do not even tell the whole *consumption* story. Only major categories of consumption have been included, and we are only beginning to examine the land area implications of waste discharges other than carbon dioxide. This means that our current footprint calculations are almost certainly significant *underestimates* of actual ecosystem appropriations and that improvements in the basic calculations will produce considerably larger footprint estimates. In short, improvements that increase the scope of our analyses will add to our sense of urgency but not necessarily shift the direction of needed policy change.

More important, *ecological* footprinting is precisely that—it provides an index of biophysical impacts. It therefore tells us little about the sociopolitical dimensions of the global change crisis. Of equal relevance to achieving sustainability are considerations of political and economic power, the responsiveness of the political process to the ecological imperative, and chronic distributional inequity which is actually worsening (both within rich countries and between North and South) as the market economy becomes an increasingly global affair. In fact, our current approach does not even account for myriad indirect effects of production/consumption such as the disruption of traditional livelihoods and damage to public health, which are often the most interesting local impacts of expanding economic activity on the environment. As use of the concept spreads, however, the term "footprint" is increasingly being used to encompass the *overall* impacts of high-income economies on the developing world (or of cities on the countryside) (see IIED 1995).

None of these limitations detracts from the fundamental message of ecological footprint analysis—that whatever the distribution of power or wealth, society will ultimately have to deal with the growing global ecological debt.[7] Our original objective in advancing the ecological footprint concept was to bolster our critique of the prevailing development paradigm and to force the international development debate beyond its focus on GDP growth to include ecological reality. This much seems to have been achieved. It is therefore gratifying that adherents to the ecological footprint concept are now extending it to claim even more of the conceptual jousting grounds in the quest for more holistic approaches to sustainability. The following section shows the footprint model at work.

Ecological Footprints of Modern Cities and "Developed" Regions

Canada is one of the world's wealthiest countries. Its citizens enjoy very high material standards by any measure. Indeed, ecological footprint analysis shows that the total land required to support present consumption levels by the average Canadian is at least 4.3 hectares, including 2.3 hectares for carbon dioxide assimilation alone (Fig. 2) (Wackernagel and Rees 1995). Thus, the per capita ecological footprint of Canadians (their average "personal planetoid") is almost three times their "fair Earthshare" of 1.5 hectares.[8]

Let's apply this result to a densely populated high-income region, the Lower Fraser Basin in the Province of British Columbia. Within this area, the city of Vancouver had a 1991 population of 472,000 and an area of 114 km^2 (11,400 hectares). Assuming a per capita land consumption rate of 4.3 hectares, the 472,000 people living in Vancouver require, conservatively, 2 million ha of land for their exclusive use to maintain their current consumption patterns (assuming such land is being managed sustainably). This means that the city population appropriates the productive output of a land area *nearly 180 times larger than its political area* to support its present consumer lifestyle.

We can also estimate the "marine footprint" of the city's population based on fish consumption. Available data suggest a maximum sustainable yield from the oceans of about 100 million tons of fish per year. First we divide the global fish-catch by total productive ocean area. About 96% of the world's fish catch is produced in shallow coastal and continental shelf areas that constitute only 8.2 % of the world's oceans (about 29.7 million square kilometers). Average annual production is therefore about 32.3 kg of fish per productive hectare (0.03 hectares per kilogram of fish). Since Canadians consume an average of 23.4 kg of marine fish annually (including discards?), their marine

[7] Naturally, the objective would be to achieve this fairly and equitably with a minimum of civil and geopolitical strife.

[8] There are fewer than 8.9 billion hectares of ecologically productive land on Earth (including those areas that should be left untouched to preserve biodiversity). If this were allocated evenly among the 1995 human population of 5.8 billion, each person would receive 1.5 hectares.

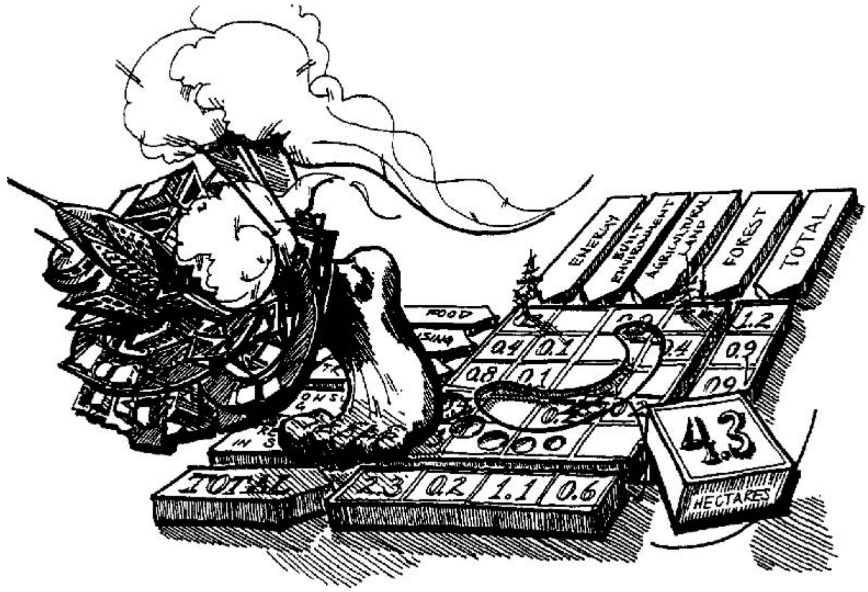

Fig. 2 The high-income footprint. The ecological footprint of the average Canadian spans several land/ecosystem types and measures over 4 hectares (*Source:* Wackernagel and Rees 1995)

footprint is about 0.7 ha each. If we add this per capita marine footprint to the terrestrial footprint, the total area of Earth needed to support Vancouver's population is 2.36 million hectares (5.83 million acres) or more than 200 times the geographic area of the city.

Although these findings might seem extraordinary to the uninitiated, other researchers have obtained similar results for other modern cities. Using our methods, British researchers have estimated London's ecological footprint for food, forest products, and carbon assimilation to be 120 times the surface area of the city proper (IIED 1995). Folke et al. (1994) report that the aggregate consumption of wood, paper, fiber, and food (including seafood) by the inhabitants of 29 cities in the Baltic Sea drainage basin appropriates an ecosystem area 200 times larger that the area of the cities themselves. (Although this study includes a marine component for seafood production, it has no energy land component.)

Extending our Canadian example to the entire Lower Fraser Basin (population = 1.78 million) reveals that even though only 18% of the region is dominated by urban land use (i.e., most of the area is rural agricultural or forested land), consumption by its human population "appropriates" through trade and biogeochemical flows the ecological output and services of a land area about 14 times larger than the home region of 5,550 square kilometres. In other words, the people of the Lower Fraser Basin, in enjoying their consumer lifestyles, have "overshot" the terrestrial carrying capacity of their geographic home territory by a factor of 14. Put another way, analysis of the ecological load imposed by the regional population shows that at prevailing material standards, *at least* 90% of the ecosystem area needed to support the Lower Fraser Basin actually lies outside the region itself. These results are summarized in Table 1.

Table 1 Estimated Ecological Footprints of Vancouver and The Lower Fraser Basin (terrestrial component only)

Geographic Unit	Population	Land Area (ha)	Ecol. Ftprnt (ha)	Overshoot Factor
Vancouver City	472, 000	11, 400	2,029,600	178.0
L. Fraser Basin	1, 780, 000	555, 000	7,654,000	13.8

It seems that the "sustainability" of the Lower Fraser Basin of British Columbia depends on imports of ecologically significant goods and services whose production requires an area elsewhere on Earth vastly larger than the internal area of the region itself. In effect, however healthy the region's economy appears to be in monetary terms, the Lower Fraser Basin is running a massive "ecological deficit" with the rest of Canada and the world.

Global Context

This situation is typical of high-income regions and even for some entire countries. Most highly urbanized industrial countries run an ecological deficit about an order of magnitude larger than the sustainable natural income generated by the ecologically productive land within their political territories (Table 2). The last two columns of Table 2 represent low estimates of these per capita deficits.

These data throw new light on current world development models. For example, Japan and the Netherlands both boast positive trade and current account balances measured in monetary terms, and their populations are among the most prosperous on earth. Densely populated yet relatively resource- (natural capital) poor, these countries are regarded as stellar economic successes and held up as models for emulation by the developing world. At the same time, we estimate that Japan has a 2.5 hectare/capita, and the Netherlands a 3.3 hectare/capita ecological footprint which gives these countries national ecological footprints about eight and 15 times larger than their total domestic territories respectively. (Note that Table 2 is based on areas of ecologically productive land only.) The marked contrast between the physical and monetary accounts of such economic success stories raises difficult developmental questions in a world whose principal strategy for sustainability is economic growth. Global sustainability cannot be (ecological) deficit-financed; simple physics dictates that *not all countries or regions can be net importers of biophysical capacity*.

Table 2 Ecological Deficits of Urban-Industrial Countries[a]

Country	Ecologically Productive land (in hectares) a	Population (1995) b	Ecol. Productive Land *per capita* (in hectares) $c = a/b$	National Ecological Deficit per capita (in hectares) $d = Ftprint - c$	(in % available) $e = d/c$
				assuming a 2.5 hectare Footprint	
Japan	30,417,000	125,000,000	0.24	2.26	940%
countries with 3–4 ha Footprints				*assuming a 3 hectare Footprint*	
Belgium	1,987,000	10,000,000	0.20	2.80	1,400%
Britain	20,360,000	58,000,000	0.35	2.65	760%
France	45,385,000	57,800,000	0.78	2.22	280%
Germany	27,734,000	81,300,000	0.34	2.66	780%
Netherlands	2,300,000	15,500,000	0.15	2.85	1,900%
Switzerland	3,073,000	7,000,000	0.44	2.56	580%
countries with 4–5 ha Footprints				*assuming Can 4.3 and US 5.1 hectare Footprints*	
Canada	434,477,000	28,500,000	15.24	(10.94)	(250%)
United States	725,643,000	258,000,000	2.81	2.29	80%

Source: Abstracted and revised from Wackernagel and Rees (1995). Ecologically productive land means cropland, permanent pasture, forests and woodlands as compiled by the World Resources Institute (1992). Semi-arid grasslands, deserts, ice-fields, etc., are not included.

[a] Footprints estimated from studies by Ingo Neumann of Trier University, Germany; Dieter Zürcher from Infras Consulting, Switzerland; and our own analysis using World Resources Institute (1992) data.

It is worth noting in this context that Canada is one of the few high (money) income countries that consumes less than its natural income domestically (Table 2). Low in population and rich in natural resources, this country has yet to exceed domestic carrying capacity. However, Canada's natural capital stocks are being depleted by exports of energy, forest, fish, agricultural products, etc., to the rest of the world. In short, the apparent surpluses in Canada are being incorporated by trade into the ecological footprints of other countries, particularly that of the United States (although the entire Canadian surplus would be insufficient to satisfy the US deficit!). How should such biophysical realities be reflected in local and global strategies for ecologically sustainable socioeconomic development?

Discussion and Conclusions: Cities and Sustainability

Ecological footprint analysis illustrates the fact that as a result of the enormous increase in per capita energy and material consumption made possible by (and required by) technology, and universally increasing dependencies on trade, *the ecological locations of high-density human settlements no longer coincide with their geographic locations*. Twentieth-century cities and industrial regions for survival and growth depend on a vast and increasingly global hinterland of ecologically productive landscapes. Cities necessarily "appropriate" the ecological output and life support functions of distant regions all over the world through commercial trade and natural biogeochemical cycles. Perhaps the most important insight from this result is that *no city or urban region can achieve sustainability on its own*. Regardless of local land use and environmental policies, a prerequisite for sustainable cities is sustainable use of the global hinterland.

The other side of this dependency coin is the impact urban populations and cities have on rural environments and the ecosphere generally. Combined with rising material standards and the spread of consumerism, the mass migration of humans to the cities in this century has turned urban industrial regions into nodes of intense consumption. The wealthier the city and the more connected to the rest of the world, the greater the load it is able to impose on the ecosphere through trade and other forms of economic leverage. Seen in this light and contrary to popular wisdom, the seeming depopulation of many rural areas does not mean they are being abandoned in any ecofunctional sense. Whereas most of the people may have moved elsewhere, rural lands and ecosystem functions are being exploited more intensely than ever in the service of newly urbanized human populations.

Cities and the Entropy Law

As noted, the populations of "advanced" high-income countries are 75% or more urban and estimates suggest that over 50% of the entire human population will be living in urban areas by the end of the century. If we accept the Brundtland Commission's estimate that the wealthy quarter of the world's population consume over three-quarters of the world's resources (and therefore produce at least 75% of the wastes), then the populations of wealthy cities are responsible for about 60% of current levels of resource depletion and pollution. The global total contribution from cities is probably 70% or more.

In effect, cities have become entropic black holes drawing in energy and matter from all over the ecosphere (and returning all of it in degraded form back to the ecosphere). This relationship is an inevitable expression of the Second Law of Thermodynamics. The second law normally states that the entropy of any isolated system increases. That is, available energy spontaneously dissipates, gradients disappear, and the system becomes increasingly unstructured and disordered in a inexorable slide toward thermodynamic equilibrium. This is a state in which "nothing happens or can happen" (Ayres 1994).

What is often forgotten is that all systems, whether isolated or not, are subject to the same forces of entropic decay. In other words, any complex differentiated system has a natural tendency to erode, dissipate, and unravel. The reason open, self-organizing systems such as modern cities do not run down in this way is that they are able to import available energy and material (essergy) from their host environments which they use to maintain their internal integrity. Such systems also export the resultant entropy (waste and disorder) into their hosts. The second law therefore also suggests that all highly-ordered systems can grow and develop (increase their internal order) only "at the expense of increasing disorder at higher levels in the systems hierarchy" (Schneider and Kay 1992). Because such systems continuously degrade and dissipate available energy and matter, they are called "dissipative structures."

Clearly, cities are prime examples of highly-ordered dissipative structures. At the same time, these nodes of intense economic activity are open sub-systems of the materially closed, nongrowing ecosphere. Thus, to grow, or simply to maintain their internal order and structure, cities necessarily appropriate large quantities of useful energy and material from the ecosphere and "dissipate" an equivalent stream of degraded waste back into it.

This means that in the aggregate, cities (or the human economy) can operate sustainably only within the thermodynamic load-bearing capacity of the ecosphere. Beyond a certain point, the cost of material economic growth will be measured by increasing entropy or disorder in the "environment." We would expect this point (at which consumption by humans exceeds available natural income) to be revealed through the continuous depletion of natural capital—reduced biodiversity, fisheries collapse, air/water/land pollution, deforestation, desertification, etc. Such trends are the stuff of headlines today. World Bank ecologist Robert Goodland uses them to argue that "current throughout growth in the global economy cannot be sustained" (Goodland 1991). It seems we have already reached the entropic limits to growth.

This brings us back to our starting premise, that with the onset of global ecological change, the world has reached an historic turning point that requires a conceptual shift from empty-world to full-world economics (and ecology). Ecological footprint analysis underscores the urgency of making this shift. As noted, the productive land "available" to each person on Earth has decreased increasing rapidly with the explosion of human population in this century. Today, there are only about 1.5 hectares of such land for each person, including wilderness areas that probably shouldn't be used for any other purpose.

At the same time, the land area appropriated by residents of richer countries has steadily increased. The present per capita ecological footprints of North Americans (4–5 ha) represents three times their fair share of the Earth's bounty. By extrapolation, if everyone on Earth lived like the average North American, the total land requirement would exceed 26 billion hectares. However there are fewer than 9 billion hectares of such land on Earth. This means that we would need three such planets to support just the *present* human family. In fact, we estimate that resource consumption and waste disposal by the wealthy quarter of world's population alone exceeds global carrying capacity and that total global overshoot is as much as 30% (Wackernagel and Rees 1995). (Again, these are underestimates based on the assumption that our present land endowment is being used sustainably, which it is not.)

Cities and Global Trade

> The structure of trade, as we know it at present, is a curse from the perspective of sustainable development (Haavelmo and Hansen 1991, p. 46).

Acknowledging the energy and material dependence of cities also forces recognition of the city's role as an engine of economic growth and global trade. According to the conventional view, trade increases both incomes and carrying capacity. Individual trading regions can export local surpluses and thereby earn the foreign exchange needed to pay for imports of locally scarce resources. Hence both the economy and the population are freed to grow beyond limits that would otherwise be imposed by regional carrying capacity. The fact that 40% of global economic growth today is sustained by trade supports this argument.

There are, however, serious flaws in the conventional interpretation. First, trade reduces the most effective incentive for resource conservation in any import region, the regional population's otherwise dependence on local natural capital. For example, the Vancouver region's seasonal access to cheap agricultural imports from California and Mexico reduces the potential money income from local agricultural land.[9] Fraser Valley farmers themselves therefore join the developers in pressing for conversion of agricultural land to urban uses which produce a higher short-term return. Because of trade, the consequent loss of foodlands in the Fraser basin proceeds without immediate penalty to the local population. Indeed, the latter are actually rewarded in the short-term by the boost to the local economy! Ironically, then, while appearing to do the opposite, trade actually *reduces* both regional and global carrying capacity by facilitating the depletion of the total stock of natural capital. By the time market prices reflect incipient ecological scarcity, it will be too late to take corrective action.

By throwing new light on commercial trade and natural flows, ecological footprint analysis also suggests a disturbing interpretation of contemporary North-South relationships. Much of the wealth of urban industrial countries comes from the exploitation (and sometimes liquidation) of natural capital, not only within their own territories, but also within their former colonies. The energy and material flows in trade thus represent a form of thermodynamic imperialism. The low cost essergy represented by commodity imports is required to sustain growth and maintain the internal order of the so-called "advanced economies" of the urban North. However, expansion of the human enterprise proceeds at the expense of "a net increase in [global] entropy as natural resource [systems] and traditional social structures are dismembered" (Hornborg 1992a, 1992b). Colonialism involved the forceful appropriation of extraterritorial carrying capacity, but today economic purchasing power secures the same resource flows. What used to require territorial occupation is now achieved through commerce (Fig. 3) (Rees and Wackernagel 1994).

In summary, to the extent that competitive open global markets and liberated trade accelerate the depletion of essential natural capital, it is counterproductive to sustainability. Trade only appears to increase carrying capacity. In fact, by encouraging all regions to exceed local limits, by reducing the perceived risk attached to local natural capital depletion, and by simultaneously exposing local surpluses to global demand, uncontrolled trade accelerates natural capital depletion, reducing global carrying capacity and increasing the risk to everyone (Rees and Wackernagel 1994).

Toward Urban Sustainability

Ecological footprint analysis not only measures the sustainability gap (Rees 1996), it also provides insight into strategies for sustainable urban development. To begin, it is important to recognize that cities are themselves vulnerable to the negative consequences of overconsumption and global ecological mismanagement. How economically stable and socially secure can a city of 10 million be if distant sources of food, water, energy or other vital resource flows are threatened by accelerating ecospheric change, increasing competition, and dwindling supplies? Does the present pattern of

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[9] The competitive advantage to imports comes from superior climate and longer growing season, abundant cheap labor, underpriced energy, and various direct and indirect subsidies (e.g., California producers pay a fraction of the real cost of providing their irrigation water).

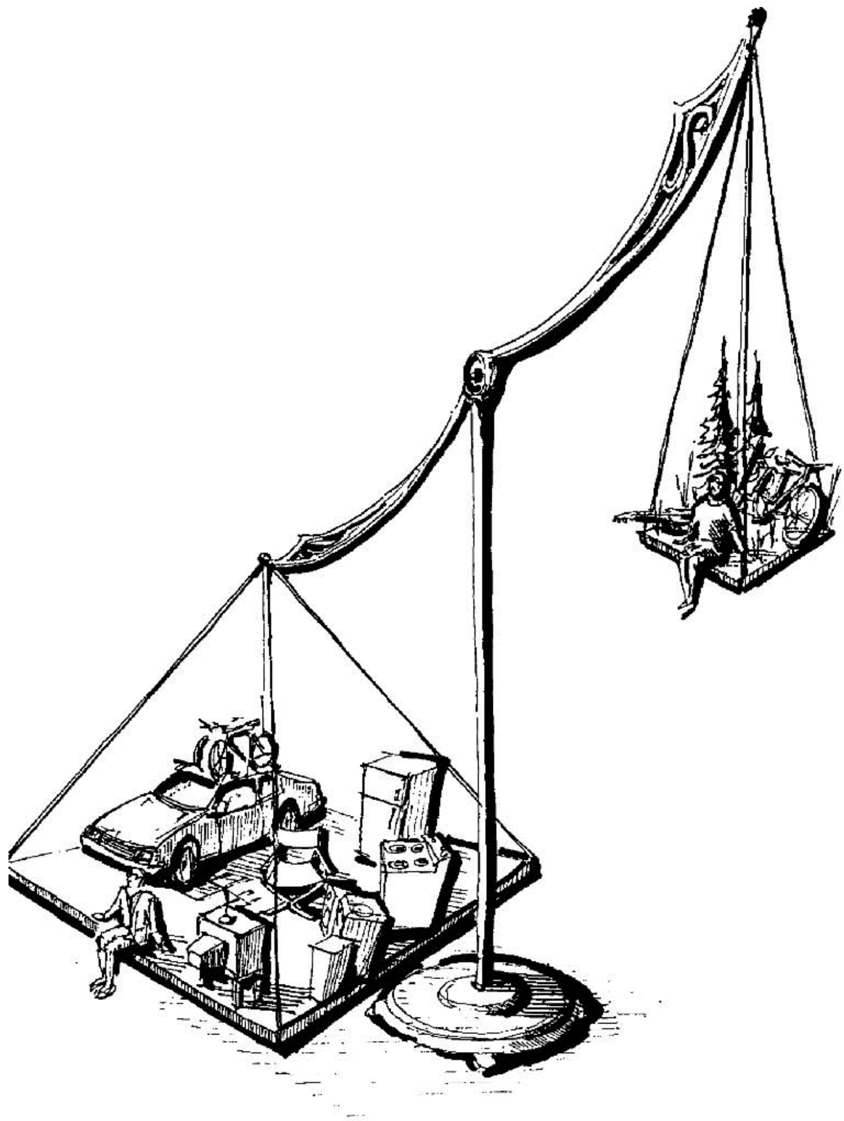

Fig. 3 Ecological inequity. In today's ecologically overloaded world, we all compete for the finite natural income flows produced by the ecosphere. The high money incomes and excessive consumption of affluent countries extends their ecological footprints into ecological space that could otherwise be occupied by the poorer nations. Even within countries, footprint sizes vary significantly as income disparity increases (*Source:* Wackernagel and Rees 1995)

global development, one that increases interregional dependence on vital natural income flows that may be in jeopardy, make ecological or geopolitical sense? If the answer is "no," or even a cautious "possibly not," circumstances may already warrant a restoration of balance away from the present emphasis on global economic integration and interregional dependency toward enhanced ecological independence and greater intraregional self-reliance. (If all regions were in ecological steady-state, the aggregate effect would be global stability.)

To reduce their dependence on external flows, urban regions and whole countries may choose to develop explicit policies to invest in rehabilitating their own natural capital stocks and to promote the use of local fisheries, forests, agricultural land, etc. This would increase regional independence thus creating a hedge against rising international demand, global ecological change, and potentially reduced productivity elsewhere.

Although greater regional self-reliance is a desirable goal on several grounds, we are not arguing for regional closure. In any event, self-sufficiency is not in the cards for most modern urban regions. The more important issue before us is to assure urban security and define an appropriate role of cities in achieving global sustainability. How can we be certain “that the aggregate performance of cities and urban systems within nations and worldwide is compatible with sustainable development goals” (Mitlan and Satterthwaite 1994) and—we would add—compatible with shrinking global carrying capacity?

Ecological Pros and Cons of Cities

A major conclusion of ecological footprint analysis and similar studies is that urban policy should strive to minimize the disruption of ecosystems processes and massively reduce the energy and material consumption, associated with cities. Various authorities share the view of the Business Council on Sustainable Development that “industrial world reductions in material throughput, energy use, and environmental degradation of over 90% will be required by 2040 to meet the needs of a growing world population fairly within the planet’s ecological means” (BCSD 1993).

Addressing these issues shows that cities present both unique problems and opportunities. First, the fact that cities concentrate both human populations and resource consumption results in a variety of ecological impacts that would not occur, or would be less severe, with a more dispersed settlement pattern. For example, cities produce locally dangerous levels of various pollutants that might otherwise safely be dissipated, diluted, and assimilated over a much larger area.

More importantly from the perspective of ecosystems integrity, cities also significantly alter natural biogeochemical cycles of vital nutrients and other chemical resources. Removing people and livestock far from the land that supports them prevents the economic recycling of phosphorus, nitrogen, other nutrients, and organic matter back onto farms and forests. As a consequence of urbanization, local, cyclically integrated ecological production systems have become global, horizontally disintegrated, throughput systems. For example, instead of being returned to the land, Vancouver’s daily appropriation of Saskatchewan mineral nutrients goes straight out to sea. As a result, agricultural soils are degraded (half the natural nutrients and organic matter from much of Canada’s once-rich prairie soils have been lost in a century of mechanized export agriculture), and we are forced to substitute nonrenewable artificial fertilizer for the once renewable real thing. All of this calls for much improved accounting for the hidden costs of cities, of transportation, and of mechanized agriculture, and a redefinition of economic efficiency to include biophysical factors.

While urban regions certainly disrupt the ecosystems of which they are a part, the sheer concentration of population and consumption also gives cities enormous leverage in the quest for global sustainability. Some of the advantages of urban settlements are as follows (based on Mitlin and Satterthwaite 1994):

- lower costs per capita of providing piped treated water, sewer systems, waste collection, and most other forms of infrastructure and public amenities;
- greater possibilities for, and a greater range of options for, material recycling, re-use, remanufacturing, and the specialized skills and enterprises needed to make these things happen;
- high population density, which reduces the per capita demand for occupied land;
- great potential through economies of scale, co-generation, and the use of waste process heat from industry or power plants, to reduce the per capita use of fossil fuel for space-heating;
- great potential for reducing (mostly fossil) energy consumption by motor vehicles through walking, cycling, and public transit.

For a fuller appreciation of urban leverage, let us examine this last point in more detail. It is commonplace to argue that the private automobile must give way to public transportation in our cities and just as commonplace to reject the idea (at least in North America) as politically unfeasible. However, political feasibility depends greatly on public support. The popularity of the private car for urban transportation is in large part due to underpriced fossil fuel and numerous other hidden subsidies (up to $2500 per year per vehicle). Suppose we gradually move toward full cost pricing of urban auto use and reallocate a significant proportion of the considerable auto subsidy to public transit. This could make public transportation faster, more convenient, and more comfortable than at present, and vastly cheaper than private cars. Whither political feasibility? People would demand improved public transit with the same passion they presently reserve for increased road capacity for cars.

Most importantly, the shift in incentives and modal split would not only be ecologically more sustainable but also both economically more efficient and socially more equitable. (It *should* therefore appeal to both the political right and left.) Over time, it would contribute to better air quality, improved public health, greater access to the city, more affordable housing, more efficient land use, the hardening of the urban fringe, the conservation of food lands, and levels of urban density at which at least direct subsidies to transit become unnecessary. In short, because of complex systems linkages, seriously addressing even a single issue in the city can stimulate change in many related factors contributing to sustainability. Rees (1995) has previously called this the "urban sustainability multiplier."

Note, in this context, that ecological footprint analysis provides a tool to compare the relative effectiveness of alternative urban development patterns, transportation technologies, etc., in reducing urban ecological impacts. For example, Walker (1995) has shown that the increased density associated with high-rise apartments, compared to single-family houses, reduces those components of the per capita ecological footprint associated with housing type and urban transportation by 40%. There is little question that urban structure and form can have a significant impact on individual resource consumption patterns (Fig. 4) (see also Kenworthy and Laube, this issue, pp. 279–308).

At the same time, we should recognize that many consumption-related human impacts that can be traced *to* cities have little to do with the structure, form, or other properties *of* cities per se. Rather, they are a reflection of societal values and behavior and of individual activities and habits. For example, the composition of one's diet may not be much related to place of residence. Similarly, that component of a dedicated audiophile's ecological footprint related to his/her use of stereo equipment will be virtually the same whether s/he resides in a farming village or industrial metropolis. In short, if the fixed elements of an individual's footprint require the continuous output of two hectares of land scattered about the globe, it doesn't much matter where that individual resides. This impact would occur regardless of settlement pattern.

There are, of course, other complications. People often move to cities because of greater economic opportunities. To the extent that the higher incomes associated with urban employment result in increased average personal consumption (*net* of any savings resulting from urban agglomeration economies), the urban ecological footprint may well expand beyond the base case. Many categories of elevated urban consumption may not even contribute to improved material well-being. Higher clothing bills, cleaning costs, and increased expenditures on security measures are all necessitated by urban life. However, they contribute nothing to relative welfare while adding to the city's total ecofootprint.

To reiterate, the real issue is whether the material concentrations and high population densities of cities make them inherently more or less sustainable than other settlement patterns. What is the materially optimal size and distribution of human settlements? We cannot say on the basis of the mixed evidence to date. Until we know the answer to this question, we cannot know on ecological grounds whether policy should encourage or discourage further urbanization. In the meantime, we in the wealthiest cities must do what we can to create cities that are more ecologically benign

Fig. 4 The urban sustainability multiplier. High density urban living significantly shrinks our per capita ecological footprints by reducing our energy and material needs. We may also find that through improved urban design, our cities can become more accessible and community-oriented places that are safer and healthier for their residents (*Source:* Wackernagel and Rees 1995)

(including, perhaps, learning to live more simply, that others may live at all). Fortunately, ecological footprinting can be use to monitor general progress toward sustainability.

Epilogue

Cities are among the brightest stars in the in the constellation of human achievement. At the same time, ecological footprint analysis shows that they act as entropic black holes, sweeping up the output of whole regions of the ecosphere vastly larger than themselves. Given the causal linkage between global ecological change and concentrated local consumption, national and provincial/state governments should assess what powers might be devolved to, or shared with, the municipal level to enable cities better to cope with the inherently urban dimensions of sustainability.

At the same time, international agencies and national powers must recognize that policies for local, provincial, or national sustainability have little meaning without firm international commitment to the protection and enhancement of remaining common-pool natural capital and global life support services. There can be no ecological sustainability without international agreement on the nature of the sustainability crisis and the difficult solutions that may be necessary at all spatial scales. The prognosis here is not encouraging. As Lynton Caldwell observes:

> The prospect of worldwide cooperation to forestall a disaster . . . seems far less likely where deeply entrenched economic and political interests are involved. Many contemporary values, attitudes, and institutions militate against international altruism. As widely interpreted today, human rights, economic interests, and national sovereignty would be factors in opposition. The cooperative task would require behavior that humans find most difficult: collective self-discipline in a common effort (Caldwell 1990).

This statement suggests that as a result of political inertia, the world may well simply stay its present development course in the blind hope that things will all work out. If so, and the analysis presented in this article is correct, humans may well become the first species to document in exquisite detail the factors leading to its own demise (without acting to prevent it).

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Health, Supportive Environments, and the Reasonable Person Model

Stephen Kaplan and Rachel Kaplan

Abstract The Reasonable Person Model is a conceptual framework that links environmental factors with human behavior. People are more reasonable, cooperative, helpful, and satisfied when the environment supports their basic informational needs. The same environmental supports are important factors in enhancing human health.

We use this framework to identify the informational requirements common to various health-promoting factors that are realizable through well-designed physical environments. Environmental attractors, support of way-finding, and facilitation of social interaction all contribute to the health-relevant themes of community, crime, and mode of transportation. In addition, the nearby natural environment, although often neglected, can serve as a remarkably effective resource.

Keywords: environmental factors · human behavior · human health · human satisfaction

Urban Ills are All Too familiar, as is their capacity to undermine health. The results of numerous studies have increased our understanding of pieces of these problems but have been less effective in drawing attention to their interrelatedness. In this commentary, we suggest a conceptual framework that provides a broader view, embracing domains currently addressed by disparate fields of study. The Reasonable Person Model (RPM) bridges environmental factors and public health domains by focusing on meeting people's informational needs. RPM posits that people are more reasonable—cooperative, helpful, constructive—when the environment satisfies such needs.

Crime, lack of community, and dependence on motorized transportation serve as pertinent examples of rampant urban ills. They have a number of commonalties: clear links to public health, strong social components, and major ties to the built environment. Crime and its associated fears have pervasive and damaging influences on people's well-being and physical activity [1]. Fear of crime has also been an important factor in the flight from the cities and the resulting proliferation of sprawl. At the same time, the residential patterns that have mushroomed across the country in the last half century have reduced the sense of community, leading to social isolation, to "disconnection and fragmentation" [2, p.13]. The same development patterns have vastly increased reliance on motor vehicles, leading to reduced physical activity and numerous other physical health problems [3, 4]. These highly interrelated domains of urban and regional planning are thus intimately related to physical, mental, and social well-being.

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Does reasonableness have anything do with these urban problems? Clearly, connecting with others requires reasonableness. Many exchanges among people do not fare well; incivility knows all too many forms. It is hardly a big leap to propose that unreasonableness undermines trust among people. However, the RPM addresses more than such social patterns. It also links human behavior with environmental factors. It is useful, then, to explore what qualities encourage people to be more reasonable with each other, with themselves, and with the places they depend upon.

The Reasonable Person Model

The RPM posits that people are more reasonable when their environment supports their basic informational needs [5, 6]. To appreciate the importance of such needs, one must consider the role of information in human evolution. Lacking the speed and strength of other species, humans have depended on their capacity to seek, store, and share information [7, 8]. However, at the same time information can be the bane of human functioning. An overwhelming amount of information, confusing information, and untrustworthy information can all readily threaten reasonableness.

We have organized people's core informational needs into 3 categories. The first, *exploration and understanding*, focuses on the acquisition and comprehension of information, both basic survival mechanisms for our species. The second, *meaningful action*, involves acting effectively based on the information one has. *Restoration*, the third category, deals with maintaining the capacity to focus on, select, and respond appropriately to the information in the environment. The 3 categories are highly interrelated. Meaningful action often requires understanding and invites exploration. Exploration can facilitate restoration. A more restored individual, in turn, can more effectively maneuver in complex settings.

Exploration and Understanding

Research on preference for outdoor environments, yielding results contrasting strongly with both theory and traditional practice at the time, led to us to propose these tandem concepts [9]. Across numerous studies using scenes of diverse outdoor environments, the single most important factor in predicting preferences turned out to be the content of the scene, and more particularly, the presence of natural elements. Beyond that, we found that 2 content-independent predictors played important roles. First, the results showed preferences for scenes that were not confusing and where it seemed possible to wander without getting lost. This component came to be called *understanding*. Secondly, preferences were greatest if the scenes offered the possibility of discovery and learning, and especially the promise of more information as one imagined oneself walking further into the scene. This was called *exploration*.

Although originating from research on landscape preferences, these 2 themes–to make sense of and acquire new information–represent enduring inclinations in many domains. For an information-based animal, survival requires the mental capacity to recognize what is going on and to figure out what might happen next while there is still time to take appropriate action [10]. This requires a high priority on exploration to learn about the environment, while at the same time requiring that the animal not stray far so that it can understand the situation. As a result, humans are eager to explore but quick to retreat to the familiar, leading to a chronic restlessness characteristic of the species.

People want to make sense of what is going on and have a strong aversion to being confused. At the same time, they prefer and benefit from acquiring, at their own pace, information that is relevant to their concerns. Exploration provides a potent means of achieving understanding.

What properties of an environment can help support exploration and understanding?

1. The amount and rate of information should be manageable. Ideally, the individual has some role in deciding how much and how quickly to explore at any one time [11].
2. Understanding requires building a cognitive map. This takes time and repeated exposures [11].
3. In any environment, some things (like landmarks) are important; others (like advertising) may be less so. The bigger, brighter, and more distracting the less important aspects are, the harder it will be to build a mental map of the environment [11].

Meaningful Action

The meaningful action component of RPM arose from the compelling findings about the harmful effects of feelings of helplessness [12, 13], along with the misguided notion that control offers an appropriate antidote (for detailed critiques, see references 14 and 15). Essentially, control is an unsatisfactory antidote for several reasons. (1) People only want control in certain circumstances; much more often they do not want the responsibility that comes with control, but rather want things to be *under control.* (2) Control is readily a zero-sum situation–when one party has more control, the other has less. (3) Control is often unrealistic; the forces of nature are typically not under human control. Human efforts to control nature have had many disastrous consequences. By contrast, participation is more realistic, less demanding, and far less likely to be harmful. People often want to be heard and to be a part of the process.

Helplessness has strong parallels to feeling "out of the loop," being disregarded, not mattering–all qualities that undermine reasonableness. By contrast, opportunities for exercising one's effectiveness serve as important examples of meaningful action. By achieving and enhancing competence, being useful to others, and gaining their respect [16], one is less likely to feel helpless and worthless. Such motivations make good evolutionary sense. Being effective is adaptive; being known for one's effectiveness helps secure one's place in the group.

We propose participation as an important corrective to helplessness. Participation responds to people's strong motivation to be heard, to make a difference, to feel that they are needed [17]. It involves being part of the action, providing input or helping to do something that needs to be done. In the urban context especially, participation can link the individual to both the physical and the social environment [18], ensuring that the person remains a functioning member of the local community. At the same time, activities that enhance a person's effectiveness can be health promoting in themselves and increase the likelihood of living in an environment that is compatible with the person's needs [19]. However, such occasions for participation can be meaningful even if they entail activities a person might not have chosen or find appealing. In his inaugural address in 1994, Detroit mayor Dennis Archer offered no promises, but rather challenged citizens to do their part–to "clean the rubbish from the storm sewer on YOUR street. Pick up the broken glass in YOUR alley . . ." [20, p.A7]. This call for participation was answered by a standing ovation.

Restoration

The third component of RPM deals with the decline in effectiveness and reasonableness because of mental fatigue. Deficits in understanding and exploration as well as the lack of opportunities for meaningful action can lead to such declines. However, even environments that are supportive in these respects can result in ineffectiveness and irritability if they contain large amounts of distracting information.

Although mental fatigue describes a very familiar phenomenon, it is a misleading label, as the mind per se is not fatigued. Rather it is a particular aspect of mental functioning, more appropriately described as the fatigue of directed attention [21]. Directed-attention fatigue makes it difficult to continue to pay attention to the many complex and competing demands in one's environment. In addition to irritability, characteristic symptoms of directed-attention fatigue are distractibility, impulsiveness, and impaired capacity to make and follow plans.

In the course of human evolution, directed attention was presumably needed far less than in modern times. A key function of directed attention is to pay attention to things that are important but not inherently interesting. For early humans, most of the things that were important–potential game, potential mates, potential dangers–were also innately interesting (just as they are to modern humans). However, for modern humans many things that are important are not particularly interesting, and many that are interesting (such as commercial advertising) are not important. Thus, directed attention is used far more extensively and is more likely to be fatigued in our contemporary world.

Many of the most effective settings for recovering from directed-attention fatigue involve the natural environment [22]. Such restorative environments are in short supply in many urban contexts. Unfortunately, environments that have the opposite effect are rampant. Transportation offers many examples of settings that can contribute to mental fatigue. Attentional resources are drained by the demands of traffic whizzing by or of navigating jammed highways lined with distracting billboards. Waiting endlessly for a bus in a place that is exposed to constant traffic as well as the elements can make mass transit no less demanding. The very notion of traffic-calming patterns acknowledges the widespread fatiguing influences of our daily means of locomotion. Road rage may be one of the more widely publicized symptoms of the resulting mental fatigue.

Information and the Physical Environment

A vast literature links the physical environment to community, crime, and transportation modes. In this section we look at some of the research findings in the context of the RPM. In particular, we take an informational perspective in examining the environmental factors to assess their impacts on human informational needs, and, in turn, on issues of health.

The arrangement of space conveys information that can make environments more interesting and attractive, facilitate way-findings, and enhance opportunities for exchange among individuals. In addition to environmental factors that are based on the way the space is organized, this section also highlights the particular role played by natural environments. Here it is the content–the trees, water, vegetation–that has strong health impacts.

Attractions

Gaining understanding generally requires repeated contact. In the neighborhood context, such repetition requires that people get out of their houses and move through their environment, ideally on foot. Exploration is encouraged when there are interesting, diverse, safe, and accessible routes and reasons for being outside. Booth et al. [23] found that among 2300 elderly persons in Australia, physical activity was significantly influenced by the availability of safe footpaths and access to facilities such as a park or recreation center. The importance of esthetic factors, including enjoyable scenery, in encouraging physical activity has been found in studies by King et al. [24] as well as studies reviewed by Humpel et al [25]. Attractive tree-lined sidewalks and functionally useful destinations (such as shops, parks, or a library) can thus contribute to health both by encouraging physical exercise and by fostering community as people become acquainted.

Benefits of having schools within walking distance have been documented in a variety of places. The Ontario Walkability Study [26], for example, has shown that a vast majority of more than

6000 elementary school–aged children would prefer to walk or bicycle to school. One could easily have assumed that the comfort and convenience of being driven to school would have been their preference. The walk to school helps children understand their local environment. In a study of 6- and 7-year-old children in 57 schools in England, Lee [27] found that those who walked to school fared better than their peers who were bused, according a variety of "adjustment" measures including concentration, response to affection, and aggression. Lee interprets this finding in terms of the inability of the bused children to make a comfortable connection between home and school; rather, he suggests, they experience a "semi-permeable barrier" between these 2 environments.

As we have seen, an object, scene, or environment that fosters understanding and exploration is more likely to be preferred. Reactions to efforts by planners during the 1960s to achieve "slum clearance" provide useful evidence of such preferred environments. The intense and persisting grief Fried [28] described after the dislocation of Boston West End residents is clearly based on the loss of their social community; however, at the same time, their deep understanding of the physical structure of the community was also shattered. Similarly, Jacobs' [29] perceptive work provides vivid imagery of the attractions and opportunities for exploration offered by sidewalks, multiple alternative routes, and diversity of kinds and ages of structures.

People are attracted to environments that permit exploration and understanding and that offer nature with its restorative properties. Destinations that allow people to carry out meaningful actions, even purposes as simple as obtaining groceries or a library book, are also attractive. Thus, knowing what people prefer [17] is important to each aspect of RPM and more likely to provide settings that encourage active engagement.

Way-Finding

In the context of his incisive analysis of urban form, Lynch [30] wrote long ago of people's profound terror of being lost or disoriented. Such fears are by no means restricted to making one's way in cities. In 1981, Reizenstein and Vaitkus (cited in reference 31, p. 67) reported that when visitors and patients were asked about their sources of stress in the hospital environment, getting lost was highest on the list. Given the many other stresses associated with hospitals, this is all the more remarkable. Fear of being lost, in turn, can reduce the likelihood of exploration.

Exploration depends on way-finding. Way-finding can be assisted by signage and maps; it is also more directly enhanced by the way the environment is structured. Lynch [30] found that some cities were far more supportive of effective way-finding than others. He identified distinctive landmarks as an important factor in reducing the danger of disorientation. Diversity of land-use patterns as well as the styles and ages of buildings also support the ease of finding one's way. The sameness of many recently built housing developments and shopping strips fail to offer such guidance.

Way-finding can also be strongly impacted by street patterns. In many newer communities, cul-de-sacs reduce automobile traffic but at the cost of discouraging walking and bicycling and making way-finding more difficult. Neotraditional designs have thus returned to the grid pattern. A fascinating synthesis of the 2 approaches, combining cul-de-sac and loop patterns at a neighborhood scale with a grid pattern at the community scale, offers the promise of supporting both local nonmotorized transportation and way-finding at the larger scale [32].

Fellow Humans

The attractions offered by the environment and the environmental patterns that support needs for way-finding strongly impact how people relate to each other. The previously mentioned works by Jacobs [29] and Fried [28] provide classic examples of urban patterns that encourage social bonds. Fried interpreted the enduring grief experienced by the relocated residents of Boston's West End in terms of their loss of social networks that had been formed in a setting that fostered knowing each other. Such networks are less likely to develop where there are no sidewalks, where one sees one's neighborhood only as one drives through it to get to the nearest main road. Duany et al. [33] offer ample imagery of the social consequences that traditional subdivisions have for young people, elderly people, commuters, "soccer moms," poor people, and the surrounding cities. As Engwicht [34] shows, the simple act of reducing vehicular traffic opens the way to a multitude of environmental solutions for creating "vibrant communities."

Interestingly, the anonymity and lack of social bonds attributed to suburbia have also been negative forces in public housing. Pruitt-Igoe, a public housing project in St. Louis, served its tenants so badly that it was ultimately razed to the ground. The failure of this costly project can be understood in terms of the absence of "defensible space" and "semi-public space "[35, 36]. These concepts emphasize that community and trust require places where neighbors can meet to become acquainted and where surveillance is easily possible. In apartments for elderly people, for example, the area where residents get their mail can serve these functions. Front porches, a theme that has been revived by new urbanist designers, similarly offer a transitional space that allows information to be exchanged and encourages people to get to know each other [37]. Oldernburg's [38] concept of "third places," such as pubs, drugstores, and cafés, provides a similar example of the way a setting can help create familiarity and hence community.

Special Role of Nature

The powerful effects of the natural environment are striking because they apply so broadly yet do not require extensive exposure in terms of either time or space. Even the minimal encounters with nature afforded by the view from the window have been shown to be related to health benefits in the context of hospitals [39, 40] and prisons, [41, 42] as well as the workplace [43] and home environment [44]. In a large-scale, 5-year follow-up cohort study of older people, perceived access to walkable green space was found to predict longevity, even after controlling for age, socioeconomic status, gender, and marital status [45]. Frumkin [46] discusses findings of numerous other studies that document the health benefits of the nearby natural environment.

The work of Kuo and Sullivan has been particularly important in showing the dramatic role played by the availability of vegetation in the context of public housing. In a series of studies they have shown the presence of nearby natural areas to be related to reduced crime, aggression, and violence [47, 48] as well as increased civility and neighborliness [49]. As these researchers indicate, the results strongly support the restoration portion of RPM [21]; their careful statistical analyses showed the effect to be attributable to the greater attentional capacity of residents in apartments with natural areas nearby.

Strong preference for nearby natural settings is evidenced not only in many empirical studies but by countless ballot outcomes showing citizens' willingness to be taxed for urban green spaces and for the preservation of nearby farms and forests [50]. Municipalities that offer opportunities for public participation often hear that citizens desire more natural areas and trails. Having natural areas nearby can provide incentives for walking and bicycling; increased pedestrian activity can enhance the likelihood that people will become familiar with each other [51]. Participation in local nature activities can increase the sense of pride in one's community [52] and strengthen urban neighborhoods [53].

Natural areas have the potential to be both attractive and restorative. They encourage outdoor activities and have the potential for making one's neighbors more reasonable and one's community safer. They can thus enhance exploration and understanding as well as facilitating meaningful action in the form of community participation.

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Small Things that Make Big Differences

Recognition of the multifaceted and extensive connections between the public health domain and issues related to land use and planning has been growing, [2, 54] with this issue of the *American Journal of Public Health* taking a further step in that direction. However, identifying these connections may not be sufficient for finding workable solutions. We offer the RPM as a way to provide general principles for identifying needed changes by addressing both environmental factors and broad health issues in the context of human informational needs. Although many studies support the appropriateness of such an approach, many opportunities for directly testing this model remain to be explored.

In some ways, what we are proposing may be seen as a radical change; however, the factors we have identified can be implemented in many small and inexpensive steps. Such changes involve making the environment more understandable, creating interesting but reassuring opportunities for exploration, providing settings that offer restorative experiences, and incorporating processes that include people in decisionmaking.

Natural environments can bring a remarkable range of benefits. Preservation and enhancement of small pieces of nature in the urban environment can be achieved at little cost. Incorporating volunteers in these efforts, as has been the case with respect to urban tree planting and natural area maintenance, does much more than reduce the cost. It provides health benefits and opportunities for meaningful action for the participants and is a source of pride to their community.

The economic perspective, currently dominant in planning, views many environmental changes in terms of amenities, failing to recognize their health implications and significance in terms of less tangible yet far more vital consequences. RPM shifts the emphasis from economics to a concern for meeting human needs. Such an approach can be implemented through numerous small changes that can make big differences. Such changes can offer even greater benefits when they are made by, and not for, communities. Through their participation, community members gain meaning while contributing to their own health and that of their community.

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Relationship Between Urban Sprawl and Physical Activity, Obesity, and Morbidity

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Abstract ***Purpose*** To determine the relationship between urban sprawl, health, and health-related behaviors. ***Design*** Cross-sectional analysis using hierarchical modeling to relate characteristics of individuals and places to levels of physical activity, obesity, body mass index (BMI), hypertension, diabetes, and coronary heart disease. ***Setting*** U.S. counties (448) and metropolitan areas (83). ***Subjects*** Adults (n = 206,992) from pooled 1998, 1999, and 2000 Behavioral Risk Factor Surveillance System (BRFSS). ***Measures*** Sprawl indices, derived with principal components analysis from census and other data, served as independent variables. Self-reported behavior and health status from BRFSS served as dependent variables. ***Results*** After controlling for demographic and behavioral covariates, the county sprawl index had small but significant associations with minutes walked ($p = .004$), obesity ($p < .001$), BMI ($p = .005$), and hypertension ($p = .018$). Residents of sprawling counties were likely to walk less during leisure time, weigh more, and have greater prevalence of hypertension than residents of compact counties. At the metropolitan level, sprawl was similarly associated with minutes walked ($p = .04$) but not with the other variables. ***Conclusion*** This ecologic study reveals that urban form could be significantly associated with some forms of physical activity and some health outcomes. More research is needed to refine measures of urban form, improve measures of physical activity, and control for other individual and environmental influences on physical activity, obesity, and related health outcomes.

Keywords: Physical Activity · Urban Design · Sprawl · Obesity · Prevention Research

Introduction

The links between physical activity and health outcomes are well established. At the time of the Surgeon General's *Report on Physical Activity and Health* in 1996, hundreds of research studies were amassed providing evidence of these links [1]. Physical inactivity contributes to increased risk of many chronic diseases and conditions, including obesity, hypertension, non–insulin-dependent diabetes, colon cancer, osteoarthritis, osteoporosis, and coronary heart disease. Despite the health benefits of physical activity, 74% of U.S. adults do not get enough physical activity to meet public health recommendations and about one in four U.S. adults remains completely inactive during their leisure time [2, 3].

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One consequence of physical inactivity—obesity—has reached epidemic proportions across age, race/ethnic, and socioeconomic groups [4, 5]. Recent data from the National Health and Nutrition Examination Survey (NHA-NES) found that 64.5% of the U.S. adult population is overweight and almost one in three is obese (30.5%) [6]. Excess weight and physical inactivity are reported to account for over 300,000 premature deaths each year, second only to tobacco-related deaths among preventable causes of death [7, 8].

There is growing interest in how physical inactivity, obesity, and related chronic health problems are affected by environmental factors. Public health researchers are expanding their horizons, moving beyond individual models of behavior to more inclusive ecologic models that recognize the importance of both physical and social environments as determinants of health [9–14]. For physical activity researchers, this interest is relatively new. A review published in 1998 found only seven such studies [14]. Since then, several studies have researched environmental determinants of physical activity [15–19]. One such study found that urban and suburban residents living in homes built before 1946 (a proxy for older neighborhoods) were more likely to walk long distances with some frequency than those living in newer homes [17]. This result was attributed to the greater likelihood of sidewalks, denser interconnected streets, and a mix of business and residential uses in older neighborhoods.

Urban planning and transportation researchers are also expanding their horizons, giving increased attention to how their fields affect human behavior and health [20]. In the past decade, more than 50 studies have related aspects of the built environment to travel for utilitarian purposes [21]. Utilitarian travel is travel not for its own sake but, rather, to engage in activities at the trip end, such as going to work, shopping, or school. It is distinct from leisure time physical activity such as walking for exercise, an end in itself. Several recent studies have focused on the relationship between the built environment and the choice of travel mode (e.g., driving a car, taking a bus, or walking) [22–29]. Walking for utilitarian purposes is consistently found to be more prevalent in dense, mixed-use neighborhoods when compared to lower density, exclusively residential neighborhoods [30]. For example, a study of two pairs of neighborhoods in the San Francisco Bay Area concluded that walking trips to commercial areas were more frequent in the older neighborhoods with nearby stores and grid-like street networks than in the newer more homogeneous neighborhoods [29].

The quantity and quality of such studies, although based on cross-sectional and case study designs, are increasing and some of these studies are now being reviewed as part of the evidence base for the *Guide for Community Preventive Services*. Developed by public health experts, this guide will recognize the importance of community design in promoting leisure time physical activity [31].

The study reported in this paper measured urban form at the county and metropolitan levels. Urban form at these levels is often characterized as more or less "sprawling." *Poor accessibility* is the common denominator of urban sprawl—nothing is within easy walking distance of anything else [32]. Although variously defined by others, we consider sprawl to be any environment characterized by (1) a population widely dispersed in low-density residential development; (2) rigid separation of homes, shops, and workplaces; (3) a lack of distinct, thriving activity centers, such as strong downtowns or suburban town centers; and (4) a network of roads marked by large block size and poor access from one place to another. Compact development is the antithesis of sprawl, keeping complementary uses close to one another.

Our working hypotheses, based on the planning and public health literature, were that residents of sprawling places would (1) walk less, (2) weigh more, and (3) have higher prevalence of health problems linked to physical inactivity than those living in more compact places. These hypotheses were tested using data from the Behavioral Risk Factor Surveillance System (BRFSS) for 1998 to 2000.

Methods

Design

The research design in this study was cross-sectional and ecologic. The degree of sprawl within counties or metropolitan areas was related to levels of physical activity, obesity, body mass index (BMI), hypertension, diabetes, and coronary heart disease (CHD) for BRFSS respondents from these particular counties or metropolitan areas. Hierarchical linear and nonlinear modeling (HLM) methods were used to control for covariates, such as age, race/ethnicity, and education, at the individual level while examining the effects of sprawl at the population level.

Behavioral and health status variables extracted from BRFSS are listed in Table 1. All data are self-reported. A condition was assumed to exist if a health care practitioner had told the respondent that he or she had the condition.

Three leisure time physical activity variables served as dependent or outcome variables: *any physical activity*, reporting any amount of leisure time physical activity over the past month; *recommended physical activity*, getting the recommended levels of physical activity in the past month; and *minutes walked*, total minutes of walking as leisure time physical activity in the past month. A person was considered to have met the physical activity recommendations if she or he reported ≥30 minutes of moderate-level physical activity on ≥5 days of the week or if he or she reported ≥20 minutes of vigorous activity on ≥3 days of the week. Walking was emphasized because of its documented relationship to urban form and its dominance as a leisure time activity (reported with almost six times the frequency of the next most common leisure time activity, gardening).

Table 1 Sample Sizes (n), Means, and Standard Deviations (SD) for Health Behavior and Health Status Variables, 1998 to 2000*

	n for County Models With All Covariates (N = 206,992)†	n for Metropolitan Models With All Covariates (N = 175,609)	Means (SD) for County Models	Means (SD) for Metropolitan Models
Any physical activity‡	149,835	126,893	0.730 (0.444)	0.733 (0.442)
Recommended physical activity§	135,344	115,006	0.268 (0.443)	0.273 (0.445)
Minutes walked‖	147,305	124,764	247.8 (493.3)	251.2 (499.6)
Body mass index (BMI, kg/m^2)	137,263‡‡	116,779‡‡	26.06 (5.15)	26.03 (5.15)
Obesity¶	137,409‡‡	116,913‡‡	0.181 (0.385)	0.181 (0.385)
Hypertension#	85,465	68,927	0.239 (0.426)	0.235 (0.424)
Diabetes**	142,685‡‡	121,292‡‡	0.056 (0.230)	0.055 (0.228)
Coronary heart disease††	40,651	31,563	0.042(0.201)	0.041 (0.197)

* For exact wording of Behavioral Risk Factor Surveillance System (BRFSS) questions and to see how calculated variables were determined, go to http://www.cdc.gov/brfss/calcvars.htm.
† N, initial sample before any BRFSS variables entered.
‡ Reported any leisure time physical activity in the last month.
§ Met recommended level of physical activity in the last month: Recommended amount is 30 minutes of moderately intense physical activity at least 5 days per week and/or 20 minutes of vigorously intense physical activity at least 3 days per week.
‖ Minutes walked for leisure during last month.
¶ BMI ≥ 30.
# Ever been told had hypertension.
** Ever been told had diabetes.
†† Ever been told had coronary heart disease.
‡‡ Includes fruit and vegetable consumption as a covariate, which reduced sample size.

Table 2 Sociodemographic and Behavioral Covariates From BRFSS Surveys*

Gender	Male (dichotomous)
Age	Ages 18 to 29, 30 to 44, 45 to 64, 65 to 74, 75+ (categorical)
Race/ethnicity	White non-Hispanic, black non-Hispanic, Hispanic, other race (categorical)
Education	College graduate, some college, high school graduate, less than high school (categorical)
Smoking	Currently smoke (dichotomous)
Diet	Fruit or vegetable consumption three or more times per day (dichotomous)

* For exact wording of Behavioral Risk Factor Surveillance System (BRFSS) questions and to see how calculated variables were determined, go to http://www.cdc.gov/brfss/calcvars.htm.

Two weight-related measures were included as outcome variables: *BMI* and *obesity*. BMI was defined as weight in kilograms divided by height in meters squared (kg/m^2) and obesity was defined as a BMI of $\geq$30.0.

Three health status variables were also modeled: *hypertension, diabetes*, and *coronary heart disease* (CHD). These three were selected for their known relationships to inactivity and obesity.

Unless otherwise noted, gender, race/ethnicity, education, age, smoking status, and fruit and vegetable consumption were included in models as individual-level covariates. The reference groups for sociodemographic variables were females, white non-Hispanics, college graduates, and persons aged 18 to 30 years. Race/ethnicity, for example, was represented by three dummy variables (1 if yes, 0 otherwise): black non-Hispanic, Hispanic, and Asian or other race. In this case, white non-Hispanics were the reference group (for other covariates, see Table 2).

A *metropolitan sprawl index*, developed for Smart Growth America (SGA), was used in this study to measure urban form at the metropolitan level. The metropolitan sprawl index is a linear combination of 22 land use and street network variables. A simpler *county sprawl index* was used to measure urban form at the county level. It is a linear combination of six variables from the larger set, these six being available for counties, whereas many of the larger set are available only for metropolitan areas. The derivation of these indices is described in the "Measures" section.

Sample

BRFSS surveys for 1998, 1999, and 2000 provided data on leisure time physical activity levels, BMI and obesity, hypertension, diabetes, and CHD [33]. Our samples consisted of 206,992 respondents from 448 counties and 175,609 respondents from 83 metropolitan areas for the pooled 1998, 1999, and 2000 BRFSS surveys. These respondents were selected from the larger BRFSS samples because they had known places of residence for which urban sprawl indices were available. Hence, it was possible to link urban sprawl indices directly to health data for all respondents. Data for 3 years were pooled to increase the statistical power of the analysis.

Metropolitan areas, as defined by the U.S. Office of Management and Budget, consist of one or more counties having a high degree of economic and social integration with one another. Our sample of respondents was smaller at the metropolitan than county level because metropolitan sprawl indices were available only for the largest metropolitan areas (500,000 population or more) with complete urban form datasets (all 22 variables that make up the metropolitan sprawl index). The sample of respondents at the county level included residents of counties that are part of smaller metropolitan areas, metropolitan areas with only partial datasets (although always with all six variables that make up the county sprawl index), or both.

As illustrated in Table 1, actual sample sizes varied among BRFSS outcome measures because of missing responses and exclusion of certain questions in certain years. For instance, physical activity data were collected by all states in 1998 and 2000 but only by certain states in 1999. Although diabetes data were gathered by all states for all 3 years, many cases were lost because fruit and vegetable consumption was included as an explanatory variable in the diabetes analysis. Fruit and vegetable consumption data were collected by all states in 1998 and 2000 but by only a small subset of states in 1999.

Sample sizes for individual counties ranged from 6 to 6429, with 353 counties having samples of 50 or more. Sample sizes were more than adequate to support stable and powerful statistical analysis. HLM uses the method of maximum likelihood to optimally combine information from different samples. In this study, counties with small samples contributed less information to the estimation of parameters than counties with large samples. Because maximum likelihood took into account the information from each county and because the number of counties in this study was large ($n = 448$), counties with small samples were not problematic from a statistical standpoint [34–36].

Measures

BRFSS is a population-based, random digit–dialed telephone survey administered to U.S. civilian noninstitutionalized adults aged $\geq$18 years. For the years under study, BRFSS collected data from 150,000 to 185,000 respondents in the 50 states and the District of Columbia. Surveys consisted of a core module of questions asked annually, a rotating core asked every other year, and optional modules asked at states' discretion. A recent review found high reliability and validity for demographic questions (e.g., age, sex, race) and moderate to high reliability and validity for behavioral and health status questions (e.g., hypertension, diabetes, level of physical activity, weight, BMI, fruit and vegetable consumption) [37]. Further information on specific questions and how variables were calculated can be found at http://www.cdc.gov/brfss/index.htm.

Smart Growth America's *metropolitan sprawl index*, used in this study to represent urban form at the metropolitan level, is the most comprehensive representation of sprawl for metropolitan areas yet developed. Technical details, including operational definitions, are available in the full technical report at the SGA web site (www.smartgrowthamerica.org).

To construct the index, 83 metropolitan areas in the United States with a total population of more than 150 million people in 2000, over half the U.S. population, were rated in four urban form dimensions. For each dimension, a composite factor was extracted from several observed variables via principal components analysis.

- *Residential density* was defined in terms of gross and net densities and proportions of population living at different densities; seven variables made up the metropolitan density factor.
- *Land use mix* was defined in terms of the degree to which land uses are mixed and balanced within subareas of the region; six variables made up this factor.
- *Degree of centering* was defined as the extent to which development is focused on the region's core and regional subcenters; six variables made up this factor.
- *Street accessibility* was defined in terms of the length and size of blocks; three variables made up this factor.

The four factors were combined into an overall index by summing them and then adjusting for the size of the metropolitan area. The four were given equal weight in the overall index. Scores were then converted to a scale with a mean of 100 and standard deviation of 25. The bigger the value of the index, the more compact the metropolitan region. The smaller the value, the more sprawling the metropolitan region. A few metropolitan regions are compact in all dimensions; New York, New York; San Francisco, California; Boston, Massachusetts; and Portland, Oregon rank near the top in overall score. Others near the top, despite one factor score below average, include Jersey City, New Jersey; Providence, Rhode Island; Honolulu, Hawaii; and Omaha, Nebraska. A few regions sprawl badly in all dimensions. These include Atlanta, Georgia; Raleigh-Durham and Greens-boro-Winston-Salem-High Point, North Carolina; and Riverside-San Bernardino, California. They rank at or near the bottom in overall score.

In an earlier study, the metropolitan sprawl index was found to have good explanatory power. The index explained a significant proportion of the variance across metropolitan areas in percent walking or taking transit to work, average vehicle ownership, vehicle miles traveled per capita, traffic fatality rates, and ground-level ozone concentration [38].

In order to examine the effects of urban form at a finer geographic scale, we developed a *county sprawl index* using a process similar to that used to develop the *metropolitan sprawl index*. The county is the smallest geographic unit that can be matched to BRFSS data. The index was estimated for 448 metropolitan counties or statistically equivalent entities (e.g., independent towns and cities). These counties comprised the 101 most populous metropolitan statistical areas, consolidated metropolitan statistical areas, and New England county metropolitan areas in the United States as of the 1990 census, the latest year for which metropolitan boundaries were defined as this study began. Nonmetropolitan counties and metropolitan counties in smaller metropolitan areas were excluded from the sample. More than 183 million Americans, nearly two thirds of the U.S. population, lived in these 448 counties in 2000.

Although sprawl has the four characteristics noted above, only two of these could be measured at the county level: low residential density and poor street accessibility. Six variables became part of the county sprawl index (as shown in Table 3). We used U.S. Census data [39] to derive three population density measures for each county: (1) gross population density (persons per square mile); (2) percentage of the county population living at low suburban densities, specifically, densities between 101 and 1499 persons per square mile, corresponding to less than one housing unit per acre; and (3) percentage of the county population living at moderate to high urban densities, specifically, more than 12,500 persons per square mile, corresponding to about eight housing units per acre, the lower limit of density needed to support mass transit. When deriving these county population density measures, we excluded census tracts with fewer than 100 inhabitants per square mile (corresponding to rural areas, desert tracts, and other undeveloped lands) located within the county, because we were only concerned about sprawl in developed areas where the vast majority of residents live. A fourth density variable, the net density in urban areas, was derived from estimated urban land area for each county from the Natural Resources Inventory of the U.S. Department of Agriculture [40].

Data reflecting street accessibility for each county were obtained from the U.S. Census, based on information concerning block size [41]. A census block is defined as a statistical area bounded on all sides by streets, roads, streams, railroad tracks, or geopolitical boundary lines, in most cases. A traditional urban neighborhood is composed of intersecting bounding roads that form a grid, with houses built on the four sides of the block, facing these roads. Therefore, the length of each side of that block, and therefore its block size, is relatively small. By contrast, a contemporary suburban

Table 3 County Sprawl Index Variables and Factor Loadings

Observed Variable	Factor Loading*
Gross population density in persons per square mile	0.846
% of population living at densities <1500 persons per square mile	–0.698
% of population living at densities $>12,500$ persons per square mile	0.846
County population divided by the amount of urban land in square miles	0.849
Average block size in square miles	–0.698
% of blocks 1/100 of a square mile or less in size (about 500 feet on a side, a traditional urban block)	0.821

* Correlation with county sprawl index.

neighborhood does not make connections between adjacent cul-de-sacs or loop roads. Instead, local streets only connect with the road at the subdivision entrance, which is on one side of the block boundary. Thus, the length of a side of this block is quite large, and the block itself often encloses multiple subdivisions to form a superblock a half mile or more on a side. Large block sizes indicate a relative paucity of street connections and alternate routes.

For each county, we calculated (1) average block size and (2) percentage of blocks with areas less than 1/100 square mile, the size of a typical traditional urban block bounded by sides just over 500 feet in length. Tracts with blocks larger than 1 square mile were excluded from these calculations because they were likely to be in rural or other undeveloped areas.

The six variables were combined into one factor representing degree of sprawl within the county. This was accomplished via principal components analysis. The principal component selected to represent sprawl was the one capturing the largest share of common variance among the six variables (i.e., the one on which the observed variables loaded most heavily; see Table 3). This one component explained almost two thirds of the variance in the dataset (63.4%). Because this component captured the majority of the combined variance of these variables, no subsequent components were considered.

To derive a county sprawl index, we transformed the principal component, which had a mean of 0 and standard deviation of 1, to a scale with a mean of 100 and standard deviation of 25. This transformation produced a more familiar metric (like an IQ scale) and ensured that all values would be positive, thereby enhancing our ability to test nonlinear relationships.

The bigger the value of the index, the more compact the county. The smaller the value, the more sprawling the county. Scores ranged from a high of 352 to a low of 63. At the most compact end of the scale were four New York City boroughs—Manhattan, Brooklyn, The Bronx, and Queens; San Francisco County, California; Hudson County (Jersey City), New Jersey; Philadelphia County, Pennsylvania; and Suffolk County (Boston), Massachusetts. At the most sprawling end of the scale were outlying counties of metropolitan areas in the Southeast and Midwest, such as Goochland County in the Richmond, Virginia, metropolitan area and Geauga County in the Cleveland, Ohio, metropolitan area. The county sprawl index is positively skewed. Most counties clustered around intermediate levels of sprawl. In the United States, few counties approach the densities of New York or San Francisco counties. (A complete list of counties and their respective "sprawl" scores is available on request.)

Analysis

In this cross-sectional, ecologic study, relationships between urban sprawl and leisure time physical activity levels, BMI and obesity, hypertension, diabetes, and CHD were estimated with HLM 5 (Hierarchical Linear and Nonlinear Modeling) software [42]. Many BRFSS respondents share characteristics of a given place, which tends to produce a dependence among respondents, violating the independence assumption of ordinary least squares (OLS) regression. Standard errors of regression coefficients associated with place characteristics based on OLS will consequently be underestimated. Moreover, OLS regression coefficient estimates will be inefficient. Hierarchical (multilevel) modeling overcomes these limitations, accounting for the dependence among respondents residing in a given place and producing more accurate standard error estimates [43].

The hierarchical models estimated in this study can be characterized as pairs of linked statistical models. At the first level, respondent health status or behavior were modeled within each place as a function of respondent characteristics plus a random error. Thus, each place had a place-specific regression equation that described the association between respondent characteristics and respondent health status or behavior within that place. At the second level, the place-specific intercept and coefficients were conceived as outcomes and were modeled in terms of place characteristics plus random effects.

In some HLM models, only the place-specific intercepts vary across places, and all of the place-specific regression coefficients are invariant across places. These are often termed "random intercept" models to denote that only the intercept randomly varies. In other HLM models, the place-specific regression coefficients randomly vary as well. These are often termed "random coefficient" models.

In this study, all models initially assumed the random intercept form. Only the intercept term in the place-specific model was allowed to vary, and all place-specific coefficients were taken as fixed. Then this assumption was relaxed, and coefficients were allowed to vary as a function of place characteristics, effectively permitting interactions between place and respondent characteristics.

Interactions between place and respondent characteristics were seldom significant and never sufficiently large to appreciably affect the relationships between place characteristics and outcome variables. Hence, the only results reported are for random intercept models.

Linear models were estimated for continuous outcome variables such as minutes of walking per month. Nonlinear models were estimated for the binary outcomes such as meeting or failing to meet recommended physical activity levels; specifically, the log-odds of the outcome was equated to a linear function of the explanatory variables.

Using HLM software, we were able to apply BRFSS final weights to observations, thereby partially accounting for different probabilities of sample selection and survey response. However, we were unable to account for the complex cluster and stratified sample survey designs used by state health departments when conducting the health surveys on which this study relies. This capability lies beyond the current HLM software.[1]

Results

County-level Analysis

Physical Activity Outcomes

As has been found in previous research, the likelihood of engaging in any leisure time physical activity in the past month was greater for males than females and for white non-Hispanics than other races/ethnicities. The likelihood declined with age and increased with educational attainment (see Table 4 for regression coefficients, t-ratios, and significance levels).

The likelihood of engaging in recommended levels of physical activity in the past month followed a similar pattern, with one exception. Those age 65 or older were more likely to meet recommended levels than were younger adults because of the greater amount of leisure time walking they do.

The amount of leisure time walking was greater for females than males and increased with age up to 75 years. Education was positively associated with minutes of walking and being physically active in general.

Controlling for these covariates, the likelihood of reporting any leisure time physical activity was not significantly related to the county index ($t = 1.01$, $p = .313$). The likelihood of getting recommended levels of physical activity was related to the county index, but just short of the traditional .05 probability level ($t = 1.94$, $p = .052$). The number of minutes walked varied directly with the county index, with residents of more compact places reporting more leisure time walking than residents of more sprawling places. The difference was not large but was statistically significant ($t = 2.95$, $p = .004$).

[1] To account for the complexities of the BRFSS sample survey design, the package of choice is SUDAAN. However, SUDAAN software is not capable of multilevel modeling, a significant short-coming in a study of this sort.

Table 4 Relationship Between Individual Characteristics, County Sprawl Index, and Leisure Time Physical Activity, 1998 to 2000 (With Coefficients, *t*-ratios, and Significance Levels)

	Any Physical Activity			Recommended Physical Activity			Minutes Walked		
	Coefficient	t	p	Coefficient	t	p	Coefficient	t	p
Male	0.246	12.1	<0.001	0.087	4.44	<0.001	−82.5	−22.1	<0.001
Age 30 to 44	−0.396	−14.7	<0.001	−0.228	−8.17	<0.001	39.4	7.95	<0.001
Age 45 to 64	−0.596	−17.5	<0.001	−0.159	−5.68	<0.001	102.2	14.9	<0.001
Age 65 to 74	−0.639	−13.6	<0.001	0.054	1.38	0.167	139.7	16.4	<0.001
Age 75+	−1.067	−26.7	<0.001	0.187	4.78	<0.001	74.1	6.65	<0.001
Black non-Hispanic	−0.322	−10.9	<0.001	−0.176	−4.96	<0.001	4.24	0.62	0.537
Hispanic	−0.625	−14.7	<0.001	−0.217	−6.15	<0.001	−27.6	−3.58	0.001
Other race	−0.553	−9.43	<0.001	−0.276	−4.49	<0.001	−37.8	−3.26	0.001
Some college	−0.417	−13.3	<0.001	−0.226	−10.3	<0.001	−8.33	−1.66	0.097
High school graduate	−0.854	−31.8	<0.001	−0.525	−21.1	<0.001	−19.8	−3.74	<0.001
Less than high school	−1.353	−39.6	<0.001	−0.946	−20.9	<0.001	−65.3	−9.24	<0.001
Currently smoke	−0.357	−15.7	<0.001	−0.273	−11.0	<0.001	−5.65	−1.16	0.245
County sprawl index	0.000552	1.01	0.313	0.000872	1.94	0.052	0.275	2.95	0.004

All else being equal, residents of a county one standard deviation (25 units) above the mean county index would be expected to walk for leisure 14 minutes more each month compared to residents of a county one standard deviation below the mean (i.e., 50 units $\times$ 0.275 minutes per unit). Comparing the extremes (New York County with an index of 352 and Geauga County with an index of 63), New York residents would be expected to walk for leisure 79 minutes more each month.

Weight-related Outcomes

BMI was higher for males than females; increased with age up to middle age (45 to 64 years), and then declined; was higher for blacks and Hispanics than for whites and lower for other races (primarily Asian); was higher for the less educated relative to the college educated; was lower for smokers than nonsmokers; and was lower for those who consume three or more servings of fruits and vegetables daily (Table 5).

After controlling for these covariates, the county index was related to BMI in the expected direction and at a highly significant level ($t = -2.84$, $p = .005$). Residents of a more compact county, one standard deviation above the mean county index, would be expected to have BMIs 0.17 kg/m^2 lower than residents of a more sprawling county, one standard deviation below the mean (i.e., 50 $\times$ $-.00344$). Again, comparing the extremes, New York residents would have BMIs almost 1 kg/m^2 less than their counterparts in Geauga County. For the BRFSS sample mean BMI (26.1 kg/m^2), this translates into 6.3 fewer pounds of body weight.

The binary variable obesity was also modeled and had a highly significant relationship to the county index ($t = 4.24$, $p < .001$). The odds of being obese in a more compact county, one standard deviation above the mean county index, were 0.90 times the odds in a more sprawling county, one standard deviation below the mean index (95% CI, 0.86 to 0.95). Table 6 reports odds ratios and confidence intervals for all binary outcome variables.

Morbidity Outcomes

Males were more likely to report having diabetes and coronary heart disease than were females. The probability of having these conditions, as well as hypertension, generally increased with age. The

Table 5 Relationship Between Individual Characteristics, County Sprawl Index, and Weight, 1998 to 2000 (With Coefficients, *t*-ratios, and Significance Levels)

	Body Mass Index			Obesity		
	Coefficient	t	p	Coefficient	t	p
Male	1.190	22.4	<0.001	0.0535	2.07	0.038
Age 30 to 44	1.696	27.7	<0.001	0.578	16.0	<0.001
Age 45 to 64	2.547	43.0	<0.001	0.852	24.2	<0.001
Age 65 to 74	1.995	23.5	<0.001	0.574	12.3	<0.001
Age 75+	0.517	6.29	<0.001	0.0542	0.98	0.327
Black non-Hispanic	1.604	20.1	<0.001	0.563	17.5	<0.001
Hispanic	0.744	8.71	<0.001	0.308	6.45	<0.001
Other race	−1.075	−10.2	<0.001	−0.448	−7.32	<0.001
Some college	0.818	14.7	<0.001	0.397	13.7	<0.001
High school graduate	1.102	17.9	<0.001	0.520	17.0	<0.001
Less than high school	1.693	19.7	<0.001	0.758	17.4	<0.001
Currently smoke	−0.985	−16.6	<0.001	−0.381	−11.4	<0.001
Fruit/vegetable consumption	−0.327	−7.54	<0.001	−0.154	−5.94	<0.001
County sprawl index	−0.00344	−2.84	0.005	−0.00212	−4.24	<0.001

Table 6 Odds of Leisure Time Physical Activity, Obesity, and Morbidity One Standard Deviation Above the Mean County Sprawl Index Compared to One Standard Deviation Below, 1998 to 2000

	Odds Ratio (95% Confidence Interval)
Any physical activity	1.028 (0.974–1.084)
Recommended physical activity	1.045 (0.996–1.092)
Obesity	0.899 (0.856–0.945)
Hypertension	0.942 (0.897–0.990)
Diabetes	0.971 (0.930–1.014)
Coronary heart disease	0.994 (0.988–1.000)

probability of having hypertension and diabetes generally decreased with educational attainment. Probabilities varied with race in more complex ways (Table 7).

The only morbidity outcome statistically linked to sprawling places was hypertension ($t = -2.37$, $p = .018$). The odds of suffering from hypertension in a more compact county, one standard deviation above the mean sprawl index, was 0.94 times the odds in a more sprawling county, one standard deviation below the mean index (95% CI, 0.90 to 0.99). As for diabetes and coronary heart disease, the county index had the expected sign in both equations, but the relationships were not statistically significant.

Direct and Indirect Effects on BMI and Obesity

To explore the mechanisms by which sprawl affects BMI and obesity, additional analyses were conducted that included *minutes walked* as an independent variable in the level-1 equations for both BMI and obesity. We wanted to see whether living in compact counties was independently related to weight, after controlling for the amount of reported leisure time walked. Results are presented in Table 8. Both variables—minutes walked and county index—were significantly (and independently) associated with BMI. BMI declined as leisure time walking increased at the individual level, and BMI declined as the county index increased at the population level. The same pattern applied to the binary variable obesity.

Thus, sprawl appears to have direct relationships to BMI and obesity, plus indirect relationships through the number of minutes walked, which varies with the county sprawl index. A portion of the overall sprawl-weight relationship is mediated through the amount of time people spend walking for leisure. The direct effect is much stronger. A 25-unit increase in the county index (1 SD) is associated directly with a .085 kg/m^2 ($25 \times .00338$) decrease in BMI. The same 25-unit increase is associated indirectly with only a .001 kg/m^2 ($25 \times 0.275 \times .000128$) decrease in BMI through its effect on leisure time walking.

Metropolitan-level Analysis

We also examined relationships between sprawl at the metropolitan level and health and health-related behaviors (see Table 9). The metropolitan sprawl index proved significantly related to only one outcome variable, *minutes walked* as a leisure time activity ($t = 2.09$, $p = .04$). Model coefficients for the county and metropolitan sprawl indices can be compared because they were standardized on the same basis, with means of 100 and standard deviations of 25. In most cases, the county index was more strongly associated with outcomes than was the metropolitan index.

Table 7 Relationship Between Individual Characteristics, County Sprawl Index, and Morbidity, 1998 to 2000 (With Coefficients, *t*-ratios, and Significance Levels)

	Hypertension			Diabetes			Coronary Heart Disease		
	Coefficient	t	p	Coefficient	t	p	Coefficient	t	p
Male	0.0191	0.74	0.46	0.221	7.32	<0.001	0.0207	13.1	<0.001
Age 30 to 44	0.689	16.8	<0.001	1.064	12.8	<0.001	0.0164	10.4	<0.001
Age 45 to 64	1.778	44.5	<0.001	2.435	31.7	<0.001	0.0594	31.8	<0.001
Age 65 to 74	2.435	52.2	<0.001	2.958	37.2	<0.001	0.0949	18.3	<0.001
Age 75+	2.456	48.9	<0.001	2.736	34.8	<0.001	0.123	19.8	<0.001
Black non-Hispanic	0.597	15.5	<0.001	0.731	18.5	<0.001	0.0167	−4.19	<0.001
Hispanic	−0.101	−1.10	0.27	0.413	7.38	<0.001	−0.0304	−5.40	<0.001
Other race	0.0203	0.24	0.81	0.284	3.06	0.003	−0.0168	−3.07	0.003
Some college	0.253	7.41	<0.001	0.361	8.33	<0.001	0.0162	7.47	<0.001
High school graduate	0.287	7.73	<0.001	0.383	9.66	<0.001	0.0128	5.93	<0.001
Less than high school	0.427	11.4	<0.001	0.869	18.2	<0.001	0.0680	8.62	<0.001
Currently smoke	−0.0454	−1.51	0.13	−0.232	-5.68	<0.001	−0.00087	−0.43	0.67
Fruit/vegetable consumption	—	—	—	0.0909	2.63	0.009	—	—	—
County sprawl index	−0.00119	−2.37	0.018	−0.00059	−1.32	0.19	−0.00011	−1.82	0.069

Table 8 Relationship of County Sprawl Index and Leisure Time Walking to Body Mass Index (BMI) and Obesity, 1998 to 2000*

	County Index			Minutes Walked		
	Coefficient	t	p	Coefficient	t	p
BMI	–0.00338	–2.87	0.005	–0.000128	–2.93	0.004
Obesity	–0.00216	–4.35	<0.001	–0.000061	–2.30	0.022

* Models included gender, age, race, education, smoking status, fruit and vegetable consumption, and minutes of walking for leisure as level-1 covariates.

Table 9 Comparison of Relationships of County and Metropolitan Sprawl Indices to Leisure Time Physical Activity, Obesity, and Morbidity Outcomes, 1998 to 2000*

	County Index			Metropolitan Index		
	Coefficient	t	p	Coefficient	t	p
Any physical activity	0.000552	1.01	0.313	0.000760	0.83	0.411
Recommended physical activity	0.000872	1.94	0.052	0.00141	1.49	0.139
Minutes Walked	0.275	2.95	0.004	0.338	2.09	0.040
BMI	−0.00344	−2.84	0.005	−0.00142	−1.03	0.307
Obesity	−0.00212	−4.24	<0.001	−0.000800	−1.02	0.312
Hypertension	−0.00119	−2.37	0.018	−0.000325	−0.49	0.626
Diabetes	−0.000586	−1.32	0.187	−0.000400	−0.60	0.548
Coronary heart disease	−0.000113	−1.82	0.069	na		

* Models included gender, age, race, education, and smoking status as level-1 covariates. Models for body mass index (BMI), obesity, and diabetes also included fruit and vegetable consumption.

Discussion

This ecologic study reveals that urban form could be significantly associated with some forms of physical activity and with some health outcomes. After controlling for demographic and behavioral covariates, the county sprawl index had small but significant associations with minutes walked ($p =$ .004), obesity ($p <$ .001), BMI ($p =$ .005), and hypertension ($p =$ .018). Those living in sprawling counties were likely to walk less, weigh more, and have greater prevalence of hypertension than those living in compact counties. At the metropolitan level, sprawl was similarly associated with minutes walked ($p =$ .04) but not with the other variables.

Although the magnitude of the effects observed in this study are small, they do provide added support for the hypothesis that urban form affects health and health-related behaviors. Furthermore, as Geoffrey Rose has pointed out, even a small shift in the distribution at the population level can have important public health implications [44].

Heretofore, BRFSS data have not generally been used to examine county- or metropolitan-level relationships. In this study, the consistency of findings with those generally found in previous research on associations between health outcomes and covariates, such as gender, age, and race/ethnicity, provides some assurance that our observations on health and urban form also have validity.

Our finding that relationships are stronger for the county index than for the larger scale metropolitan index is not surprising. Most metropolitan areas consist of multiple counties whose built environments vary significantly between central and outlying counties. The county environment might be more representative of what is actually experienced on a day-to-day basis by residents than is the overall metropolitan environment. By implication, as research shifts from the macroscale (metropolitan and county) to the meso- and microscales (community and neighborhood), we might expect that the explanatory power of environmental variables to predict outcomes will improve.

This study is exploratory and subject to important limitations that call for additional research.

- Because this study is ecologic and cross-sectional in nature, it is premature to imply that sprawl causes obesity, hypertension, or any other health condition. Our study simply indicates that sprawl is associated with certain outcomes. Future research using quasi-experimental designs is needed to tackle the more difficult job of testing for causality.
- As shown in Fig. 1, the presumptive relationships between environment (urban form), physical activity, and health are multiple and complex. In particular, leisure time physical activity constitutes only one of four major sources of physical activity, the others being related to occupation, household, and transportation. Greater precision in characterizing physical activity will help disentangle the effects of urban form on health. Recognizing the need to monitor more than just leisure time physical activity, the 2001 BRFSS questions were modified to include transportation-, household-, and work-related physical activity.
- In this study, we were not able to account for the complex nature of the BRFSS sampling design, reinforcing the need for cautious interpretation of these early findings. There is growing interest in using BRFSS at the local level, and CDC is in the process of developing methods to adjust the state-based weights for use at the local level.
- Better measures of walking are needed to improve our ability to trace potential differences that are attributable to urban form. The variable *minutes walked* is based on people who reported walking as one of their top two forms of leisure time activity. It excludes walking as a less frequent form of leisure time activity or walking for other purposes. The new BRFSS questions should help produce more comprehensive measures of walking.
- We recognize that the relationships between sprawl and behavior or weight are probably not completely linear. It might be that certain thresholds or critical levels of "compactness" are needed before community design begins to have a palpable influence on physical activity—increasing density from one or two houses per acre to three or four might not meet the threshold needed for change. Subsequent research will have to explore threshold effects.
- This study relates physical activity and health to the built environment at the county and metropolitan levels, which are large areas compared to the living and working environments of most residents. If environmental effects are felt most strongly at the community or neighborhood level, at least for walking, this study needs to be supplemented with research at a finer geographic scale. Future research will need to use geographic information system (GIS) data to hone in on the specific living and working environments of individuals.
- Because they are not directly measured in either of the sprawl indices, many other environmental variables that might act directly or interact to influence physical activity, such as availability and quality of parks, sidewalks, and bike trails, are not accounted for in this study. Also missing from this analysis are potentially important environmental variables such as climate, topography, and crime. Future research will have to fill this void by specifying more complete outcome models.
- By focusing on physical activity, this study largely ignores the other side of the energy equation—calories consumed as opposed to calories expended. In this study, leisure time walking accounts for only a small portion of the relationship between urban form and BMI. Although we expect other forms of physical activity to fill some of this gap, differing patterns of food consumption must also be explored. Only our fruit and vegetable consumption variable begins to get at that

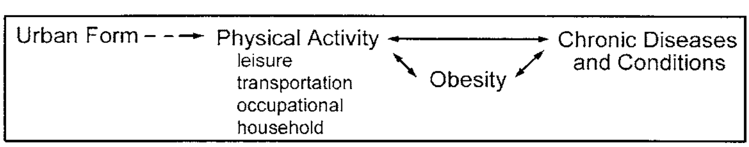

Fig. 1 Established (Solid) and Speculative (Dashed) Relationships

dimension of the problem. Caloric intake could have a spatial component. Future research could, for example, relate the density of fast food restaurants and availability of food choices to diet and obesity.

The growing interest in how policies and the environment serve to encourage or discourage health-related behaviors is attested to by the new focus on these issues in journals such as this one and by new initiatives of governmental and nongovernmental organizations such as the CDC, with its Active Community Environments (ACES) research group, and Robert Wood Johnson Foundation, with its commitment of more than 70 million dollars to promote active living. Over the past several decades, we have engineered much of the physical activity out of our daily lives. Now our task is to understand how opportunities for physical activity can be revived.

SO WHAT: Implications for Health Promotion Practitioners and Researchers

This exploratory study seems to indicate that, after controlling for individual differences, those living in sprawling counties are likely to walk less in their leisure time, weigh more, and have greater prevalence of hypertension than those living in more compact places. Combined with other research from public health and urban planning, there is moderate support for the assertion that urban form can have significant (positive or negative) influences on health and health-related behaviors.

If this assertion holds true, health practitioners can improve public health by advocating for more compact development patterns. Public health researchers can refine their understanding of physical activity, obesity, and morbidity by including urban form variables in their analyses.

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Megacities as Global Risk Areas

Frauke Kraas

Abstract In the last few decades a striking world-wide trend towards rising fatalities and economic losses due to natural and man-made hazards can be observed. One major influencing factor is growing urbanization, megacities being particularly prone to supply crises, social disorganisation, political conflicts and natural disasters. They can be both victims and producers of risks. This article concentrates on major risks and gives examples of a) environmental hazards (such as earthquakes and volcanic eruptions, storms, floods, droughts and heat waves, snowfall, frost and avalanches as well as global sea-level rise). Furthermore b) man-made hazards such as air, water and soil pollution, accidents, fires, industrial explosions, sinking land levels, diseases and epidemics, socio-economic crises, civil riots and terror attacks, nuclear accidents as well as war, germ and nuclear warfare are addressed. Finally, the most remarkable deficits in research are summarized, as well as future tasks.

Keywords: Megacities · urbanization · hazard · risk · prevention · management · governability

1 Introduction

In the last few decades a striking world-wide trend towards rising fatalities and economic losses due to natural and man-made hazards can be observed. Although there is a broadly reverse relationship between disaster-related deaths and damages in the developed world, the number of people affected in the developing world is increasing. One major influencing factor is growing urbanization, and above all megacities are particularly prone to supply crisis, social disorganization, political unrest, natural and man-made disasters due to their highest concentration of people and extreme dynamics of development. Therefore we have to consider megacities as global risk areas; their vulnerability is high. Megacity research is about to become a central element of global peace policy.

In this context it is the prime objective of the following contribution to give an overview of the specific risks for megacities, the most remarkable gaps in research and tasks for the future concerning strategies for risk minimisation and prevention.

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2 Growing Risks and Global Urbanization Processes

The major reasons for increasing disaster-related fatalities and damages, even if the frequency of geophysical events remains unchanged and despite a number of efforts for disaster reduction, are to be found in the following processes (Smith 1996, p. 36 ff.):

1. *Population growth:* The number of people likely to be affected by hazards and their vulnerability are growing, owing to decreasing security of food supplies, malnutrition, inadequate health care and fragile livelihoods. Conversely, the growing elderly population needs special support.
2. *Rural land pressure:* Insufficient access to land resources and environmental degradation force people to adopt unsustainable land-use practices, whereby further areas are becoming regions at risk (e.g. with deforestation, over-cultivation, soil erosion; Kasperson, Kasperson & Turner 1995.
3. *Urbanization:* Mainly rural-urban migration leads to the concentration of growing numbers of people in often unsafe, overcrowded, badly built, predominantly coastal cities.
4. *Inequality:* Disparities and fragmentation continue to increase, thereby exacerbating the vulnerability of these societies.
5. *Climate change:* The consequences of global warming are destabilizing ecological and economic systems.
6. *Political change:* The developed countries appear to be reducing their commitment to internal welfare and development aid.
7. *Economic growth:* The increasing amount of built property, the complexity of economic dynamics, shortages of building land and growing spatial demand contribute to growing exposure to catastrophic property damage.
8. *Technological innovation:* Technology offers better forecasting, safer construction techniques and immediate reaction, but also leads to growing dependency and additional potential for hazard.
9. *Social expectations:* Wealthier societies in particular expect absolute security of supply and services, thus relying more on public systems than on their own coping strategies in case of an emergency (Laquian 1994).
10. *Global interdependence:* The functioning of the world economy is reinforcing hazard vulnerability and growing interdependence effects others far outside the immediate area of impact.

In the context of these developments current urbanization processes play a key role. Until World War II urbanization had primarily been a feature of developed countries, only since then has rapid urban growth also begun in developing countries, encouraged by intensified industrialization and migration to the cities. For the first time in the history of man, more than half of the world's population will live in cities in the year 2007 (UN 2002, p. 1). World-wide, the proportion of the population as a whole living in cities rose from 29.8 % (1950) to 37.9 % (1975) to 47.2 % (2000), and it will probably increase to 57.2% in 2010 or 60.2 % in 2030 (UN 2002, p. 4). in the industrialized countries 73 % of the population was living in cities by 1990 (ca. 877 million), while in developing countries the corresponding figure was only 37 %, although in absolute figures was it 1,357 million. It is assumed that the rate of urbanization in industrialized countries will only increase slightly to 78%, i.e. 1,087 million people, while in developing countries the increase will be enormous, although it may vary from state to state. With an estimated 57 % of the total population, probably more than 3,845 million people will live in cities here in 2025 (HABITAT 2001, Coy & Kraas 2003).

3 Megacities: The Growing Phenomena of World-wide Urbanization

Megacities have particular significance in this worldwide process of urbanization. New scales have evolved ("mass matters"), including new dimensions of large high-density concentrations of population with immense sprawl and a serious increase in infrastructural, socioeconomic and ecological overload (Fig. 1). Furthermore these may develop extreme dynamism in demographic, economic, social and political processes (background information: Doga & Kasarda 1988, Aguilar & Escamilla 1999, Alonso-Villar 2001, Beckel 2001). Both phenomena – the new scale and dynamism – make megacities vulnerable, especially where administrative direction is absent or weak.

According to different definitions, megacities are on a purely quantitative level those metropolises with a population of over 5 million (Bronger 1996a, 1996b), more than 8 million (UN 1987, p. iii; Fuchs et al. 1994, p. 1, 42, 43; Chen & Heligman 1994) or more than 10 million inhabitants (Mertins 1992). Some authors also set a minimum level for population density (at least 2,000 persons/km^2) and only include cities with a single dominant centre (Bronger 1996a, 1996b), whereby polycentric agglomerations such as the Rhein-Ruhr area in Germany, for example, with 12.8 million inhabitants, are excluded. Others include this polycentric mega-urban region (UN 2002, p. 116 ff.). Ultimately it is futile to fight over a fixed definition of megacities, as any setting of minimum/maximum values is subjective and thus open to debate. Furthermore, there are the problems of inconsistent spatial boundaries for administrative districts, as well as the reliability of up-to-date population figures given inconsistent censuses, projections and estimations. International statistics are not based on similar areas of reference, so that the figures given for the size of cities and megacities are generally not comparable.

While in the 1950s there were only four cities with a population greater than 5 million, by 1985 there were already 28 and in 2000 39. Depending on the threshold accepted as a lowest population value for a megacity, there are currently world-wide 16, 24 or 39 megacities (Fig. 2); in the year 2015 there will probably be almost 60 (Fig. 3). Before World War II megacities were a phenomenon of industrialized countries; today by far the greater number are concentrated in developing countries and *N*ewly *I*ndustrializing *C*ountries (NICs). Two thirds of the megacities are now in developing

Fig. 1 Tokyo: Sea of houses in the centre of the city (Photo: Kraas 2000)

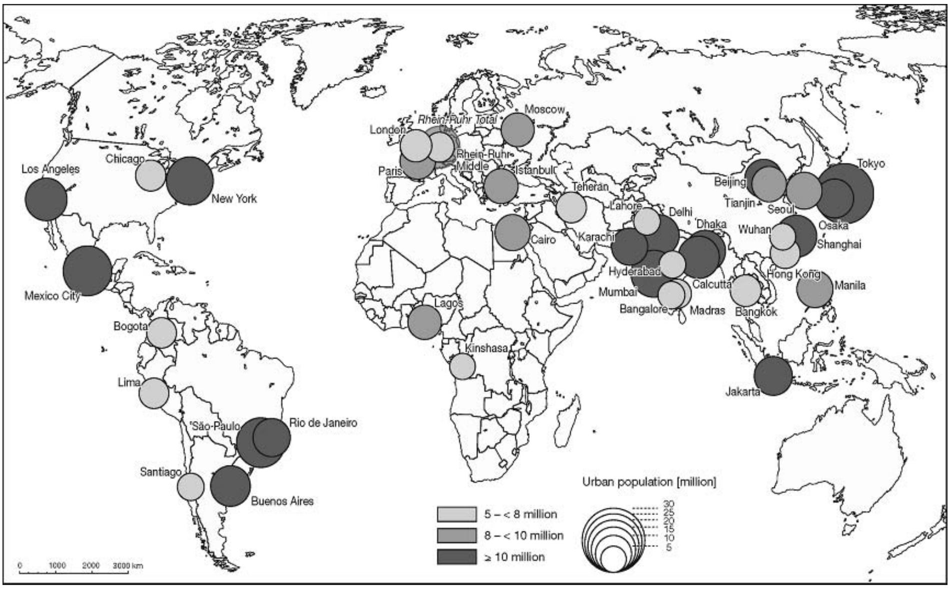

Fig. 2 Megacities with 5, 8 and 10 million inhabitants in the year 2000 (Source: UN 2002; Cartography: SPOHNER)

countries, most of them in East and South Asia. At the moment just under 394.2 million people live in megacities, 246.4 million of them in developing countries, more than 214.5 million in Asia. In 2015 there will be about 604.4 million people living in megacities (Fig. 4; UN 2002, p. 120 ff.). In some of these – Mexico City, São Paulo, Seoul, Mumbai, Jakarta and Teheran – the population figures have almost trebled between 1970 and 2000 (Bronger 1997, Heintel & Spreitzhofer 1998, Lo & Yeung 1998, p. 7; Dege 2000, Coy 2001).

Megacities have a large number of specific problems with occasionally striking structural similarities. Fuchs et al. (1994, p. 7) emphasize: “While it is true that megacity development is rooted in

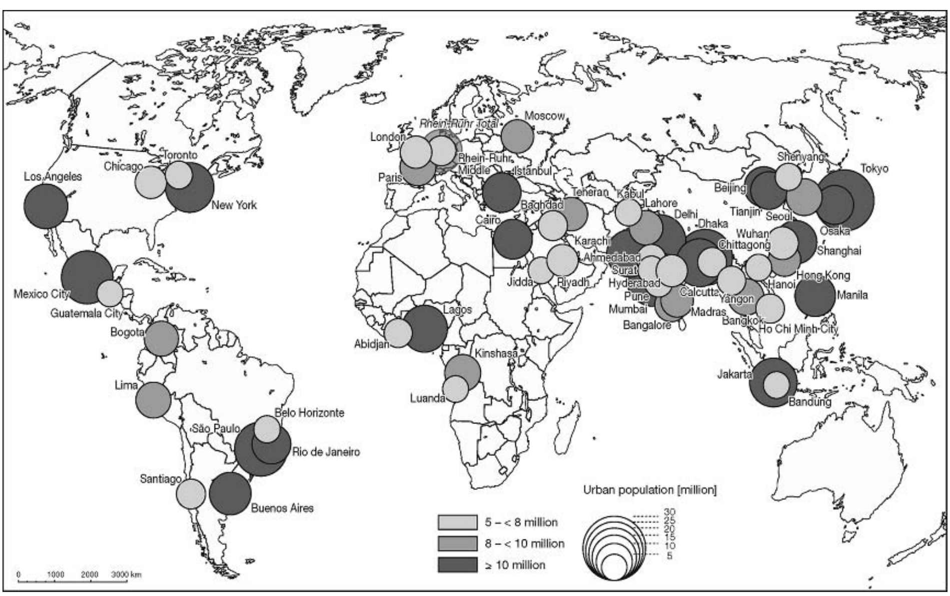

Fig. 3 Megacities with 5, 8 and 10 million inhabitants in the year 2015 (Source: UN 2002; Cartography: Spohner)

Size of megacities [million inhabitants]	Inhabitants [million] in the year 2000	2015	increase 2000–2015
5 to < 8	95.1	156.5	61.4
8 to <10	74.1	107.4	33.3
≥10	225.0	340.5	115.5
Altogether	*394.1*	*604.4*	*210.2*
Number of megacities	39	58	19

Fig. 4 Population in megacities in the years 2000 and 2015 (Source: UN 2002, p. 120 ff.)

its specific country or regional context, . . . megacities have more in common with each other than with their own hinterlands". Among the most important common characteristics are high population concentration and density, with extreme levels in some cases, largely uncontrolled spatial expansion, high traffic levels, in some cases severe infrastructural deficits, high concentrations of industrial production, signs of ecological strain and overload, unregulated and disparate land and property markets and insufficient housing provision, in some cases extreme socio-economic disparities – as well as a high level of dynamism in all demographic, social, political, economic and ecological processes. One must nevertheless be wary of generalizing statements, for among megacities there are clear differences in infrastructural quality, the level of economic development (e.g. transformation processes; Gaubatz 1999, Han 2000, Lentz 2000, Stadelbauer 2000), social polarization or political leadership and governability, differences which should not be ignored.

4 Megacities as Global Nodes and Action Spaces

Undisputedly megacities are economically active areas and nodal points of globalization (Korff 1996, Feldbauer 1997). Here are concentrated the economic activities of the national and the global economy; they are a pioneering driving force for developments at a national and broader regional level (Lo & Marcotullio 2001). The special potential of megacity economies is to be found in the broad spectrum and high dynamism of their economic activities; national and broader regional potential is maximised here and connected into international economic cycles. A high concentration of trained specialized workers as well as a large reservoir of "cheap" labour make possible the diversified development of labour markets with a broad spectrum of supply and demand for the formal and the informal sector (Meyer 1994, Gertel 2002), above-average capital accumulation as well as a generally high capacity for innovation and attractiveness for international direct investment.

Many megacities are pronounced primate cities, i.e. a disproportionately high percentage of the national population is concentrated in them – producing as a result an extremely polarized urban system, where other cities in the state are of almost no significance (Fig. 5; Bähr & Wehrhahn 1995, Bronger & Strelow 1996, Kraas 1996). This is also true for functional primacy, i.e. as regards the dominant role of the megacity in national administrative, political, economic, social and cultural affairs (Gormsen & Thimm 1994). As a rule megacities in developing countries ultimately do not play a significant political or economic role in the global urban system, in spite of their enormous populations, and thus – unlike the megacities in the industrialized countries such as Tokyo, New York or Paris – do not belong among the World Cities or Global Cities with global political and economic centres of power and influence (Sassen 1991, GaWC 2003). To a certain extent even the Newly Industrializing *Countries* (NICs) could more adequately be considered as Newly Industrialized *Cities* as only the cities are connected to the global economy.

Fig. 5 Seoul: Public housing within traditional living quarters (urban re-densification) (Photo: Kraas 2000)

5 Megacities as Global Risk Areas

Megacities are particularly endangered as they are increasingly affected by natural and man-made hazards, and they are – as highly complex and vulnerable systems – ever more exposed to global change as well as contributing to it themselves: Megacities can thus be both victims and producers of risks. Still, it must be kept in mind that until now only a few megacities have experienced disasters, and anticipative projections are necessarily speculative (Blaikie et al. 1994, p. 37 ff.; Mitchell 1999, p. 22 ff.).

Due to their particular characteristics and problems mentioned above, megacities prove to be highly vulnerable in crises and disasters: sudden supply shortages, heavy environmental burdens or major catastrophes can quickly lead to serious bottlenecks or emergencies for a vast number of people, or aggravate further those of the socially weakest groups among the population (Wisner 1999). Constraints and conflicts may acquire multiple dimensions, as they arise amid poorly co-ordinated administration and planning, the growing influence of an increasingly globalized economy, growing socio-economic disparities and intensifying environmental burdens. Risks are therefore related to complex sources, factors and networks.

Generally it is necessary to distinguish between the terms "hazard (or cause) as a 'potential threat to humans and their welfare' and risk (or consequence) as 'the probability of a specific hazard occurrence' " (Smith 1996, p. 5). When large numbers of people are killed, injured or affected, the event is termed a "disaster", without a universal definition of its scale; more qualitatively it is defined to be "an event, concentrated in time and space, in which a community undergoes severe danger and incurs such losses to its members and physical appurtenances that the social structure is disrupted and the fulfilment of all or some of the essential functions of the society is prevented" (Smith 1996, p. 20).

The two general complexes of environmental and man-made hazards and risks (Fig. 6) need to be examined at least to an extent under the growing influence of global change processes. As far as the impacts of hazards with sudden or slow onset are concerned, one has – in a not entirely static way – to consider gains and losses as well as direct and indirect, tangible and intangible, short-term and long-term effects (Smith 1996, Plate & Merz 2001). In the following several examples of cities

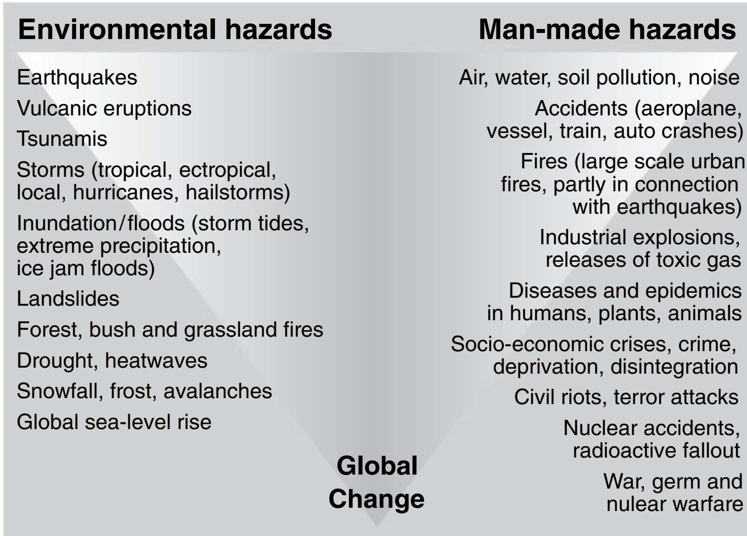

Fig. 6 Environmental and man-made hazards

and megacities are given (note: only some had been megacities at the time of the hazard events and only severe "big event" hazards are included – excluding the "normal" risks of malnutrition and poor health; Blaikie et al. 1994, Wisner 1999):

- *Environmental hazards* include *earthquakes* and *volcanic eruptions* (e.g. earthquakes near Calcutta 1737 and the Kanto earthquake in Tokyo 1923 killed up to 300,000 and 140,000 people, in Mexico 1985 more than 10,000 people were killed, nearly 41,000 injured and about 80,000 became homeless; Puente 1999, p. 306 f.; Blaikie et al. 1994, p. 174 ff.; in Kobe 1995 about 6,300 people were killed and the damages went up to more than 100 billion US-$; Fig. 7; Mitchell 1999, p. 2, 13, 510), *tsunamis* (cities in Java and Sumatra by Krakatoa eruption 1833; Kondratyev, Grigoryev & Varotsos 2002, p. 57 f.), *storms* (e.g. cyclones in Calcutta 1864, Haiphong [near Hanoi] 1881 and in Mumbai 1882 killed "tens of thousands", 300,000 and 100,000 people, the Ise Bay typhoon in Japan 1959 killed more than 5,000 people; Mitchell 1999, p. 1, 510), *inundation and floods* (e.g. floods in Dhaka 1988; Blaikie et al. 1994, p. 138 ff.), *landslides* (e.g. in Rio de Janeiro 1988 277 people were killed, 735 injured and more than 22,000 displaced; Blaikie et al. 1994, p. 183), *forest, bush and grassland fires* (e.g. the so-called "haze" in 1997 strongly affected Jakarta and Singapore; Goldammer 2001), *droughts* and *heat waves* (e.g. urban microclimate with heat islands; Jauregui 1993), intensive *snowfall, frost* and *avalanches* as well as the *global rise in sea-level* (for New York: Rosenzweig & Solecki 2001).
- *Man-made hazards* include *air, water* and soil *pollution* (including aspects of hydropolitics [Fig. 8], inadequate or non-existing provision for waste disposal and sewage treatment, degradation and contamination of soils), *accidents, fires* (e.g. fires in Los Angeles 1993 burned more than 200,000 acres, destroyed over 1,000 houses and caused losses exceeding 500 millions US-$; Wisner 1999, p. 403), *industrial explosions* (e.g. the release of toxic gas in Bhopal 1984 caused between 2,000 and 6,400 deaths, 34,000 eye defects and 200,000 people's migration; Smith 1996, p. 321 ff.), *sinking land levels* (e.g. Bangkok, Delhi), *diseases* and *epidemics* (e.g. the black death, great plague, cholera and typhoid in London 1349–1861 caused more than 250,000 fatalities; Parker 1999, p. 189 ff.; SARS in Guangzhou, Hong Kong 2003; Tanaka et al. 1996; Krafft, Wolf & Aggarwal 2003), *socio-economic crises* (e.g. the so-called Asian crisis 1997; Douglass 2000, Kraas 2000), *civil riots* and *terror attacks* (e.g. Los Angeles 1992,

Fig. 7 The earthquake of Kobe in 1995 damaged many infrastructure lines (Photo: © dpa-Bildarchiv)

Jakarta 1998, New York 2001; Wisner 1999, Gamerith 2002), *nuclear accidents* (e.g. Chernobyl 1986; Smith 1996, p. 328 ff.) as well as *war, germ and nuclear warfare* (in World War II a high number of large cities were effected, like London, Paris, Moscow, as well Dresden 1944, Hiroshima 1945; Blaikie et al. 1994, p. 43 f.).

Water is a key, not only in arid areas but also seasonally issue in the humid tropics; water is as bottleneck factor in more than one respect:

- Sinking land levels on the coast and in delta areas
- Floods and inundation
- Adequate supply of drinking water
- Waste water management
- Health risks: water-borne diseases

Fig. 8 Aspects of hydropolitics in megacities

Under the dynamics of global (environmental and human) change megacities will face growing risks: As well as the above-mentioned hazards, symptoms of ecological overload and "consumption" of space will further concentrate in urban areas, resources (e.g. energy, water) are used up at rising rates, sinking land levels become more of a problem (as most megacities are located on coasts and flood plains; background information: Hardoy, Mitlin & Satterthwaite 2001, Satterthwaite 2001). As far as global societal changes are concerned, megacities are prone to growing socio-economic vulnerability because of pronounced poverty, socio-spatial and political fragmentation, sometimes with extreme forms of segregation, disparities and conflicts. Uncontrolled sprawling as well as the absence of land-use planning and control result mainly from the enormous dynamism of growth. The juxtaposition of very different local lifeworlds, lifeforms and lifestyles (including ethnic, social and behavioural groups) play a significant differentiating role. Socio-economic polarization and fragmentation as well as social disintegration in megacities endanger the stability and development, especially when these are made even more unstable and prone to disruption because of large socio-economic disparities (Coy & Pöhler 2002). The cumulative result of different causes, effects and feedback effects in problem areas interconnected at many levels reinforce each other, which impedes the analysis of material flows and their management. Thus the risk potential increases rapidly in a

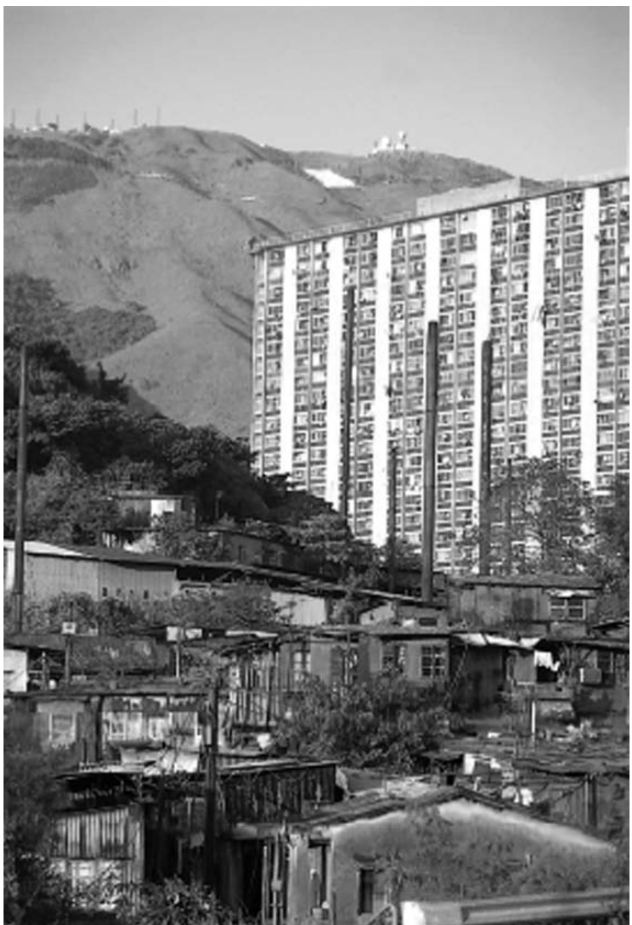

Fig. 9 Hong Kong: New Town Sha Tin (Photo: Kraas 1994)

complex manner. On the other hand, megacities offer positive potential for global transformation (e.g. minimization of "space consumption", high effectivity of resources applied, efficient disaster prevention – insofar as corresponding strategies for direction and provision have been developed; e.g. Tokyo: Togo 1995, Sorensen 1999, Taniguchi 1999, Flüchter 2000).

The above-mentioned high risk potential is already beyond the reach of direction and governability. For many megacities, not for all (Fig. 9), governability is de facto no longer given, and this loss of governability affects planning and control as much as the comprehensive organization and management of urban responsibilities, the establishment of general order, and control over development processes (Pile, Brook & Mooney 1999, Kraas 2000). This is due to (weak) political-administrative decision makers and heterogenous political-administrative organizations which are not horizontally interconnected (large numbers of independent departments as well as separate municipalities within the mega-agglomeration). Likewise central development and environmental planning as well as their implementation are impossible – especially as the megacities' own budgets are not even sufficient for the minimization of problems, let alone their solution (see for example studies of Rio de Janeiro, Manila, Delhi, Bangkok; Roy 1994, Shaw 1995, Rakodi 1997, Ribbeck 2002). A particular problem lies in the fact that the existing administrations and their organizational structures are not designed for the scale which these cities have meanwhile attained; studies in Tokyo, Bangkok, Shanghai and Beijing also confirm this (Kraas 1996, 1997; Hohn 2000).

Finally, it must be born in mind that megacities cannot be regarded as isolated entities as they are nodal points of national and global systems and economies, which radiate far beyond the cities' direct hinterlands and spheres of influence into other economic activity spaces as well as development peripheries. A major earthquake in Toyko or Los Angeles, for instance, would affect national and global economies in a much more far-reaching way than the terror attacks of September 11th 2001 did – directly and indirectly.

6 Gaps in Research

Given the insufficiency of broader and detailed studies world-wide, especially concerning comparative work, there is a considerable need for research, especially as regards the above-mentioned man-made hazards, their implications, the actors involved (hazard and risk politics) and causal networks. The same is true for the following themes of risk factors: land-use dynamism, resource consumption, deficits in water supply, rubbish and sewage disposal, the securing of energy supplies, insufficient transport infrastructure, human security and health, social vulnerability, functional interconnections in megaurban economics, crisis- and disaster prevention-planning, the investigation and development of systems for administrative direction (best practice models); extended primary research, the implementation and improvement of complex methods of steering and management, the best of these with the aid of improved highest resolution satellite images, GIS and modelling and monitoring systems.

There are furthermore clear deficits in research into the relationship between the megacities and world cities as regards their competitive position within the global economic system. From an urban economic perspective, the question arises as to which economic development strategies could help peripheral megacities to achieve global significance (e.g. the location of transnational businesses, super-regional financial and service centres or global transport, research and technology centres). On the other hand, beyond any economic influences, the cultural significance of megacities and its development potential must be examined more closely. Finally, the relationships between the influence of the areas of responsibility and the specific scope for action of administration, economics and politics on controlling factors, as well as good governance and self-organization must be analysed in greater detail.

Structures and processes in megacities in developing countries have a different expression, dimension and dynamism: They often grow significantly more quickly than the smaller cities within the national urban systems. Within a few years, or at most decades, housing, infrastructure, jobs, public utilities as well as health and education services have to be provided for hundreds of thousands of people. Often a disproportionate growth of marginal settlements results. But the percentage of the population living in marginal settlements varies considerably; also public utilities systems, legal frameworks and internal organizational structures of the marginal settlements are very heterogenous (Bähr & Mertins 1995, 2000). Living conditions for lower and middle income groups in particular are impaired by markedly class-specific processes of displacement in the market for land, housing and capital and by sometimes extreme disparities in income levels and property ownership as well as in the educational system and health care. Growth in the informal sector outside of state regulated economic activities is also characteristic. Informal employment now contributes considerably to the GDP in many megacity economies (Feldbauer & Parnreiter 1997, p. 17).

In the context of the risks and problem situations discussed here, the need for a definition of different risk types and profiles among individual megacities becomes apparent. In insurance, for example, the Munich Re Group allocates so-called risk indices for megacities and agglomerations via the components of "endangerment, proneness to damage, exposed values", e.g. for Tokyo (Index 710), San Francisco (167), Los Angeles (100), Osaka-Kobe-Kyoto (92), Miami (45), New York (42), Hong Kong Perlflussdelta (41), Manila (31), London (30), Paris (25); it is not possible to provide a detailed discussion of the construction of these indices here; Munich Re Group 2003, p. 34 ff. Beyond such a systematic identification of risk factors and a purely economic perspective in megacities (which comprises the amounts of significant damage, persons affected and numbers of casualties and fatalities), scientific research should also be concerned with the analysis of vulnerability (as a cross-cutting issue and priority), which is increasingly affecting the urban poor in the face of multifarious "internal" and "external" vulnerability factors (Morrow 1999, Sanderson 2000).

The issues cited above with a considerable need for increased research, should remain at the forefront. A dynamic, process oriented approach should be preferred (decision-support-system). Actor- or action-oriented components and long-term perspectives must be included. Approaches previously neglected (e.g. material flow models – also transposed to communication flows and socio-cultural steering models) must be taken into account. An integrated, holistic approach which integrates the human dimension into the hitherto mainly physical aspects of man-environment science, must include early warning systems (Zschau & Küppers 2003) as well as modern problem-solving techniques (integrated monitoring and modelling of material flows, highest resolution satellite imaging and multidimensional GISs). In this context analyses of existing solution strategies must be undertaken and new models developed. The extent to which potential for sustainable development specific to megacities exists must also be examined (Satterthwaite 2001).

7 Tasks for the Future: Risk Minimization and Risk Prevention

Systematic risk minimization and risk prevention are essential in the light of the expected global consequences of megacity risks and impacts. The areas with the greatest need for action, on which strategies should concentrate, are as follows:

- In the area of the environment and health, problems of emission reduction, the provision of clean drinking water as well as sewage and rubbish disposal are the most important issues. The inadequate environmental situation is already directly responsible for more than a quarter of avoidable health problems.

- The problems of habitat and spatial expansion associated with dynamic population growth, together with inadequate land-use planning and poor achievability continue to be unsolved problems.
- In the case of the rapidly increasing concentration of (international) economic activities, conflict arises between urban economies and national economic interests. Power and its social and spatial effects create polarized active and marginal economic spaces, at a national, regional and local level. The megaurban economies with their multi-layered interconnections with increasing globalization and the expansion of the informal sectors have hitherto been little researched.
- Already, existing symptoms of economic, ecological, infrastructural and socio-economic overload are increasing dramatically and are thus extreme urban security risks at a global level.
- Increasing disparities and sometimes extreme socio-economic fragmentation with serious social and spatial segregation are sources of social and political centres of conflict.
- Natural and man-made catastrophic events are an increasing threat for the world's megacities; disaster prevention planning is increasing in significance.
- Poor governability and directability inhibit controlling and correcting intervention on the part of state and local authorities in order to minimize or indeed prevent poor conditions.

Thus the improved direction and management of megacities are central tasks in global conflict research and work for peace: It is essential to extend the administrations' analytical and planning competence as well as their direction capacity and scope for action. Specifically, prevention plans and action programs in the case of crises, conflicts, and catastrophes should be developed. Direction "from the top down" increasingly needs a counterbalance "from the bottom up" through civil society organizations which take on responsibility for the common good at a neighbourhood and urban district level and develop culturally adapted, local strategies for improving the quality of life. The analysis of such networks, their power structures, institutional integration and forms of direction as well as the development of prevention strategies are central tasks for (geographical) megacity research.

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Why Is Understanding Urban Ecosystems Important to People Concerned About Environmental Justice?

Bunyan Bryant and John Callewaert

Keywords: environmental justice · culture · values · motivation · affordable housing · poverty

Ecosystems, and more specifically urban ecosystems, represent important models for understanding particular places, environments, or regions. Even though ecologists generally view ecosystems as functional and geographic units, we suggest that ecosystems should also be viewed as cultural constructs. By this we mean that understandings of ecosystems exist within a cultural context, and meanings assigned to ecosystems cannot help but reflect this cultural context. Thus, understandings of nature are themselves cultural constructions, even though their referents have independent standing as biological realities (Kirsch 1999).

Environmental justice is both a field of study and a social movement that seeks to address the unequal distribution of environmental benefits and harms and asks whether procedures and impacts of environmental decision making are fair to the people they affect. A primary issue for people concerned about environmental justice is that some groups, most often communities of color and low-income communities, face a disproportionate exposure to environmental health risks such as air and water pollution, and environmental hazards such as landfills, incinerators, sewage treatment plants, and polluting industries. As with ecosystems, environmental justice can also be understood as a cultural construct—one that focuses on the class and racial aspects of environmental concerns.

This chapter begins by examining in more detail the perspective of ecosystems and environmental justice as cultural constructs. Understanding the connections between urban ecosystems and environmental justice concerns is an important first step and will prove helpful in identifying common areas of knowledge in supported sustainability. Following these conceptual perspectives, specific reasons are presented as to why an understanding of urban ecosystems is important to people with environmental justice concerns. Finally, three strategies are offered to strengthen the connection between an understanding of urban ecosystems and environmental justice.

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Understanding Ecosystems and Environmental Justice as Cultural Constructs

Urban Ecosystems

While it is true that biological realities such as species present, the amount of water available, climatic conditions, flows and patterns of resource exchange, and so on ultimately set the limit for a region's political, economic, and social institutions, we hypothesize that if ecosystems, be they urban or rural, are not understood within a cultural context, then we fail to fully understand them. An ecosystem as a culturally defined construct says more about ourselves than perhaps about ecosystems, and we must therefore understand the values and belief systems that shape and motivate behavior toward ecosystems, particularly if we hope to explain why people protect or exploit the Earth (Cronon 1996).

Cultural constructs may be defined as mental representations of external reality that are unique to the human species (White 1949). Humans have an extraordinary ability to construct, symbolize, and name the world. Language or combinations of symbolic constructions are used to organize thoughts for understanding and meaning, for organizing behavior and management, and for envisioning and planning the future. Humans name elements of the world for specific purposes. Terms such as wildlife, park, virgin forest, externality, carbon sink, and brownfield are examples of how we construct conceptions of the world. Such conceptions are often for the selfinterests of certain groups and their use or application can influence the building and maintaining of urban ecosystems.

Our speech, our work, our play, and our social life, our ideas about ourselves and nature all exist within a cultural context that is historically, geographically, and culturally determined and cannot be understood apart from that context. Thus, the way we understand an ecosystem, the way we see and value an ecosystem is a construct of a particular culturally determined context (Cronon 1996). When we think of ecosystems or modify them, however, we think of nature—not culture. Cities are more visible cultural constructions; they are places where ecosystems have been transformed by humans to support urban habitats that bear little resemblance to nature.

We contend that conceptions of ecosystem education, management, policy, planning, and design are based in cultural values of efficiency, beauty, convenience, and utility. Decisions about ecosystems are therefore value-laden. Forests cannot be managed or planned unless decisions are made about whom they will serve. Will they serve industry, local human communities, or non-human species? More specifically will they serve the spotted owl or the English sparrow; hikers or hunters; naturalists or lumbermen or some combination of the above? Will forests be managed for native oaks or Norway maples, jack pines or walnuts (Cronon 1996)?

In an urban environment we also need to consider the parts of an ecosystem that are managed for affordable housing, the business community, industrial production, landfills, incinerators, sewage treatment plants, urban parks, and recreational facilities. What do the spatial relations of these entities say about other cultural constructs such as race, class, and gender? We must ask ourselves the question: Who benefits and who loses from these culturally defined constructions? The answers to these questions depend upon the cultural values and belief systems of a particular place and people. In essence, we need to deconstruct and examine our notions of ecosystems to discover their core meanings.

To understand the values and motivations that shape our actions toward an ecosystem and to explain our actions that abuse that system, we should be more concerned about the impact of culture. Many of our values and motivations are steeped in the marketplace and the immense power of the accumulation system. Culturally transformed and commodified ecosystems are another extension of the market, producing both "social goods" and "social bads" and alienation from the natural world in which we live. Externalities such as hazardous waste are traditionally ignored by the market system and often find their way into neighborhoods with high proportions of low-income residents or people of color; these communities, themselves struggle to be valued and fully respected by the market system.

Environmental Justice

Environmental justice as a cultural construct challenges the absolute authority of the market system and places emphasis on the interconnections between environmental quality, social justice, and civil rights. With a specific focus on distributional equity, environmental justice adds new layers of analysis to the field of environmental science. Just as environmental scientists examine how human actions can alter local, regional, and global ecological systems, environmental justice advocates call attention to the environmental repercussions of human actions that threaten and disrupt particular social systems.

Environmental injustice can cover a very broad range of environmental disparities and the unequal enforcement of environmental regulations (Goldman 1994; Lavelle and Coyle 1992). In an analysis of 64 empirical studies, Benjamin Goldman (1994) found an overwhelming body of empirical evidence that people of color and lower incomes face disproportionate environmental impacts in the United States. All but one of the 64 studies found environmental disparities either by race or income, regardless of the kind of environmental concern or the level of geographic specificity examined. One of the most influential investigations of environmental injustice was a national study on the distribution of hazardous waste sites that was conducted by the Commission for Racial Justice (CRJ) of the United Church of Christ (1987). The CRJ study revealed that the proportion of minorities residing in communities with a commercial hazardous waste facility is about double the proportion of minorities in communities without such a facility. Where two or more facilities are located, the proportion of residents who are minorities is more than triple. Furthermore, the CRJ study and others have shown that race is often the single best predictor of where commercial hazardous waste facilities are located (Commission for Racial Justice of the United Church of Christ 1987; Bryant and Mohai 1992).

Today people of color and low-income communities across the country are rebelling against the siting of locally undesirable land uses in their communities (Taylor 2000; Tesh and Williams 1996). Through these struggles, people concerned about environmental justice are deconstructing the belief that such communities are valueless. They are seeking to make their communities safe, healthy, viable, and productive. Often these activists are focused on specific places within urban ecosystems that experience the brunt of toxic and hazardous waste and polluting industries; they decry environmental racism and distrust government and the scientific community because neither provides answers to their demands for certainty or immediate solutions. As a result, many community groups are doing their own research in order to find answers to their questions, and to reconstruct their communities to be more viable and livable places.

The struggle of two community groups—the Alum Crest Acres Association and the South Side Community Action Association—representing a predominantly middle-class African American neighborhood on the south side of Columbus, Ohio clearly demonstrates such concerns. Since the mid-1980s the community has voiced numerous environmental and health complaints about a Georgia-Pacific resins facility in the neighborhood. Community concern about the facility peaked in 1997 when chemicals were improperly mixed and exploded violently, leaving one worker dead, several others injured, parts of the facility in ruins, and many residents upset about property damage and a host of alleged health impacts (Edwards 1997). Frustrated with the lack of response from the Columbus Health Department, the community groups applied for and received funding from the United Way to conduct their own health study. The funding for the study, however, was temporarily suspended due to the influence of local government officials (Columbus Dispatch 1999). The community groups have also filed a complaint under Title VI of the 1964 Civil Rights Act with the

Office of Civil Rights of the U.S. Environmental Protection Agency (USEPA) alleging a discriminatory impact from permit decisions by the Ohio Environmental Protection Agency concerning the Georgia-Pacific facility. The civil rights complaint was recently accepted for investigation by USEPA. Ohio EPA is also under investigation currently by USEPA for failing to adequately enforce environmental regulations (Edwards 2000). The above represents only one of many communities where people of color and low-income groups are disproportionately impacted by environmental hazards.

Connecting an Understanding of Urban Ecosystems with Concerns About Environmental Justice

A deeper and more comprehensive understanding of urban ecosystems will perhaps provide the incentive for a paradigm shift to knowledge that is more sustainable and that will change how we build and reconstruct healthy and livable urban ecosystems. When we speak of sustainable knowledge, we use "sustainable" as an adjective to describe knowledge just as others use the term in sustainable development. Sustainable knowledge is broader than sustainable development in that the former is knowledge that guides our behavior and our understanding of nature. When we speak of sustainable knowledge, it is not knowledge that will remain static, but it is knowledge that mimics nature. It is knowledge that is consistent with and not disruptive of the Earth's life cycles, and it is knowledge that will sustain plant and animal species (Hawken 1993). In nature, the waste of one life form becomes food for another life form. In the same way we need to create knowledge so that the waste from one industry will become the raw materials for another (Anderson 1998). Such a sustainable knowledge conception of urban ecosystems is needed to help eliminate the environmental injustices present in so many cities.

An urban ecosystem built upon injustice will not survive. When people are not allowed their fair share of market benefits but are saddled with more than their fair share of environmental burdens, an ecosystem view tells us that such disparities and imbalances will eventually create problems for the entire system. This emphasis on social dimensions such as race, class, and justice adds important new dimensions of analysis that have not yet been considered in current understandings of humans as components of ecosystems.

Environmental justice often involves the struggle of a particular neighborhood or community against a local polluting industry or facility. A better understanding of ecosystems can help environmental justice advocates connect their specific concerns to broader, regional issues that may reveal significant environmental and/or health concerns. For instance, besides having impact on people of color in a low-income neighborhood, emissions or waste from a facility also may be harming a preserved area or estuary. The work of Walsh, Warland, and Smith (1997) has shown that when environmental justice advocates establish coalitions and partnerships with other groups and institutions, they are much more successful than if they had only focused on the environmental justice aspects of the problem.

For people concerned about environmental justice, knowledge of an ecosystem's characteristics is very important. For example, after one community on the south side of Chicago learned how emissions from a proposed incinerator would combine with the prevailing wind patterns to disproportionately impact their neighborhood, a new environmental justice organization was formed (Schwab 1994). In the Columbus, Ohio, example cited earlier, there have been numerous concerns expressed about contamination of the underground aquifer. These concerns, however, have not been fully explored in terms of what an ecosystem perspective can reveal regarding water flows and other vital characteristics.

Another way to understand and develop the connections between urban ecosystems and environmental justice is through geographic information system (GIS) applications. Such techniques have become an important tool for those with environmental justice and ecosystem concerns. Combining economic, social and environmental data will support better-coordinated efforts by all involved parties. GIS can help environmental justice advocates better understand the characteristics and dimensions of ecosystems and it also can help ecosystem scientists become more fully aware of the important overlap between physical, ecological, and social dimensions of an ecosystem.

Strategies

In order to strengthen the connections between environmental justice and understanding ecosystems, we offer the following three strategies: (1) promoting community-based research initiatives; (2) incorporating environmental justice concerns within a sustainable knowledge construct of urban ecosystems; and (3) supporting the formation of a new type of professional that will be able to forge the connections between understanding urban ecosystems and concerns about environmental justice.

Promoting Community-Based Research

There must be a vigorous effort to increase community involvement in designing initiatives that promote the understanding of urban ecosystems and environmental justice. This emphasis on participatory research or community-based research is highlighted in the recent Institute of Medicine (1999) report, *Toward Environmental Justice: Research, Education, and Health Policy Needs* and has been supported by other leading research institutions. Our emphasis here on community-based research is not to exclude other research approaches, but to suggest that given particular settings and desired outcomes, some approaches are more appropriate than are others. Table 1 offers a modified version of Patton's (1990) typology of research purposes and explains some of the differences in research approaches based on a number of variables.

We emphasize a community-based research approach for the following three reasons: (1) it focuses the locus of control of knowledge within the community; (2) people feel they have more control over their lives by being actively engaged in a democratic process of creating knowledge for sustainable and viable communities; and (3) by understanding the role of knowledge and culture. A fundamental difference between community-based research and both action research and basic research is that rather than seeking simply to resolve a problem or to expand knowledge, community-based research involves participants in challenging basic cultural constructs and knowledge that may support unsustainable practices or conditions. In analyzing data from a national study of community-based research in the United States, Sclove, Schammell, and Holland (1998) note that community-based research processes differ fundamentally from mainstream research in being coupled relatively tightly with community groups that are eager to know the research results and to use them in practical efforts to achieve constructive social change. Community-based research is not only usable, it is actually used and, more than that, used to good effect.

In many cases community groups concerned about environmental justice and involved in participatory research have been very successful in problem solving (Schafer, et al. 1993). This process does not mean, however, that they would do a better job than a researcher from a university community—this is hardly the point. The point is that they feel that have control over what happens in their community by being involved in a participatory process. Most importantly, community-based research provides the opportunity for people to learn about their communities (Israel, et al. 1998). This is particularly important in terms of understanding urban ecosystems as cultural constructs, with all strengths and weaknesses that such a concept presents.

Table 1 A Typology of Research Purposes

	Basic Research	Action Research	Community-Based Research
Focus of research	• Questions deemed important by one's discipline or personal intellectual interest	• Organization and community problems	• Solve problems and identify societal causes of problems
Goals	• Knowledge as an end in itself; discover truth	• Solve problems in a program, organization, or community	• Advance practical knowledge • Solve problems and create systemic change • Empower participants and strengthen capacities
Key assumptions	• The world is patterned; those patterns are knowable and explainable	• People in a setting can solve problems by studying themselves	• People in a setting can understand, confront, and change oppressive forces
Desired results	• Contribution to theory	• Solving problems as quickly as possible	• Changing societal structures that created problems
Investigator's relationship with providers of data	• Subjects/Objects • Detached and external	• Clients/subjects • Agency control • Internal or external	• Participant and researcher co-control • Responsive to community needs • Internal priority with external help
Utility of research for providers of data	• Low likelihood (atleast not directly or soon)	• Low to medium depending on agency status and role	• High
Who benefits from research	• University • Scientific community or other researchers •"Trickle down" to policy makers	• Client agency • Clients of agency • Policymakers, community leaders	• Participants and community members • Total system (conflicting parts and interest groups) • Constituency

Source: Adapted from Patton (1990) and Chesler. *Personal communication*.

Community-based research can also strengthen or build new social relationships and enhance social trust. This is essential in situations that are complex or involve controversial and value-laden issues. There is a long history of outside researchers producing work that has had devastating impacts on people of color such as the Tuskegee Study (Hatch, et al. 1993; Thomas 1991), Jensen's (1968) research on black children, Schockley's (1992) work on intelligence, and Moynihan's (1965) report on black families.

Community-based research, though, is not at present a prominent form of research in the United States. Fig. 1 clearly shows that community-based research accounts for only a small fraction of research expenditures in the United States. It is not the type of research that usually gets funded and it may require many years of work in order to establish the necessary community trust and participation. Furthermore, many of the results of community-based research—such as community empowerment—are not standard research outcomes and are therefore difficult to quantify. Despite the lack of attention given to community-based research, we still believe it offers the most appropriate methodology that can enable people to deconstruct the cultural conceptions of urban ecosystems while empowering them to use an understanding of urban ecosystems to address environmental injustices. The Loka Institute in Amherst, Massachusetts has spent several years studying the idea

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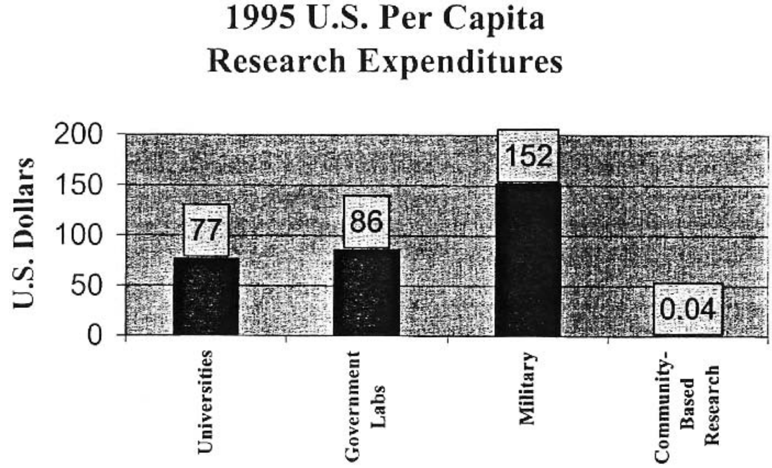

Fig. 1 Comparison of Research Expenditures. Adapted from Sclove, Schammell, and Holland (1998)

of community-based research and suggests that the university-affiliated community research centers in Holland, popularly known as “Dutch Science Shops,” offer one approach to more successfully promote community-based research in the United States. Through such centers, the Dutch are able to invest in community-based research at 37 times the U.S. rate (Sclove, et al. 1998).

Incorporating Environmental Justice in Urban Ecosystem Understandings

Our second strategy of incorporating environmental justice concerns within the context of understanding urban ecosystems builds directly on the opportunities for local learning emphasized with community-based research. Although people concerned about environmental justice often place health and survival issues as top community priorities, they must place these priorities in the context of the failure of urban ecosystems; they must make the connection between healthy ecosystems that mimic nature and just social systems. Those gathered at the First National People of Color Leadership Summit understood this when they established the 17 Principles of Environmental Justice and acknowledged that environmental justice affirms the ecological unity and interdependence of all species, and affirms the need for urban ecological policies to clean up and rebuild cities in balance with nature (Newton 1996). These principles challenge the unsustainable aspects of urban ecosystems and suggest ways in which such systems can be more sustainable.

When environmental justice struggles join with wider regional environmental coalitions, there is greater overall success than if each issue group works independently (Walsh, et al. 1997). It is also important for those working to advance the understanding of urban ecosystems to reach out to environmental justice advocates. Connecting an understanding of an urban ecosystem with a desire for environmental justice can help identify the wider social and environmental implications of a particular concern (landfill, incinerator, industrial facility, etc.). This connection can lead to stronger networks providing greater overall resource mobilization and support. An important tool to help incorporate environmental justice within an understanding of urban ecosystems is GIS (e.g., the “Environmental Mapper” website www.epa.gov/compliance/whereyoulive.html of the USEPA is one option for working with GIS that is accessible to anyone with access to the Internet). Having a visual representation of the overlap of social and environmental concerns is key to building these important partnerships.

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A New Type of Professional

Our third strategy, calling for the formation of a new type of professional, is the most important one of all. This type of person is needed in communities, government agencies, and university research institutions. They will need to understand the culturally constructed dimensions of urban ecosystems and be able to forge connections with a variety of groups with environmental justice concerns. Only recently have humans been recognized as components of ecosystems (McDonnell and Pickett 1993). This recognition was seen as a fundamental shift in the understanding of ecosystems. A similar fundamental shift is now needed to promote sustainable knowledge and to fully appreciate the complexity of the human dimension of ecosystems.

Such professionals need to accept the challenges of working directly with communities and should be able to use participatory and community-based research methods to involve community members in the design, implementation, data collection, and analysis of research initiatives connecting environmental justice with a better understanding of urban ecosystems. Institutions also need to recognize the difficulty of such work as it reaches across disciplines and challenges many of the assumptions of scientific inquiry. In a recent analysis of adaptive strategies for ecosystem management, Aley, et al. (1999) provide helpful examples of how some natural resource professionals are successfully integrating social dimensions into natural resource initiatives.

Conclusions

We have attempted to explore the importance of understanding urban ecosystems from the perspective of people concerned about environmental justice. By understanding ecosystems as cultural constructs, we are pointed in the direction of intentional cultural change to help ameliorate environmentally unjust conditions. Understanding the complexities of race, class, and justice is key to understanding the complexity of urban ecosystems as culturally defined constructs. If we fail to fully understand urban ecosystems, the urban environment will continue to decline and be made more unhealthy by policy decisions that disproportionately affect people of color and low-income groups. An understanding of urban ecosystems also can provide opportunities for additional networking and information exchange that can be very helpful to environmental justice initiatives.

To achieve these ends we have stressed the need for participatory or community-based research initiatives, the importance of placing concerns about environmental justice within the context of urban ecosystems, and finally we have to called for a new type of professional that will be able to use sustainable knowledge to help us reconstruct urban ecosystems to be more livable. The results of such efforts would hopefully be better community-based initiatives that are informed by economic, social, and ecosystem realities. There would also be stronger, more successful coalitions working on environmental justice and expanding the understanding of urban ecosystems. The time for such action is now.

Acknowledgments We are grateful for the insights on the issue of community-based research provided by Dr. Mark Chesler, Sociology Department, University of Michigan—Ann Arbor.

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Section VI
The Anthropposphere: Planning and Policy

This section attempts to put into perspective the reasons why quick remedies to environmental problems are often difficult to find. The selection of eleven papers delineate the relationships among the critical components necessary for addressing contemporary urban-ecological problems. These include the institutional, social, economic and biological systems that must interact in connecting science with policy and development.

There is still no generally accepted decision support system for even basic aspects of urban ecological knowledge. What exists is simply the experimental approaches of cities and towns worldwide. Political decision and planning tools have improved the environmental conditions of urban areas. But, the results show many cases where cities have inadequate instruments to implement urban ecological knowledge. Often the administrative forms of governance are inefficient environmental guides. In many cities steering institutions do not exist or are without power to influence proper urban development. Sometimes the participation of the urban dwellers in these processes is only marginal or of little effect. But everywhere urban planning attempts to improve its efficiency in urban ecological aspects and its basic understanding of this complex system approach. Together with scientists, planners work to refine models of well functioning cities and include into these research and urban design models the urban ecological system approach to discover an "ideal form" of urban development. Scientists and planners must work together and listen to each other to bridge the still existing gap between idealistic visions and traditional practicability. Good results are reflected in a number of local and often spatially limited examples of districts and patterns in cities. These examples are visible facts of putting urban ecology into practice and to stimulate researchers, citizen leaders and communities to step forward to improve the urban ecological development. The ambitious idea to build an ideal (ecological) city is more and more being replaced by visionary improvement of existing internal urban pattern and by well designed new urban quarters following the "Leitbild" of the Ecological City.

We begin with the assumption that in order to address environmental problems, there must be institutions with the ability to accurately identify and prioritize problems, know what might be feasible and effective solutions, and have the authority to actually effect change (the last is the rarest and most difficult). Change in urban and urbanizing environments may come in the form of proposing ideal human settlement patterns that accommodate the needs of both humans and the natural world. In order to offer such proposals, we must understand the complex interaction among all aspects of the human and natural environment and the varied effects that they have on one another. Knowing these effects allows us to link those understandings to broad policy directives, as well as to more specific planning and design prescriptions for on-the-ground development. The papers in this section tell this story and, in the process, provide us with an appreciation for the difficulty and complexity of implementing effective programs for addressing environmental problems.

Dietz et al. (2003) offer an excellent discussion of institutions and why in some cases they are ill-equipped to address many of our contemporary environmental problems. The question of boundaries of authority is most important to many of today's problems, where our institutions were historically created along political boundaries and for the most part do not conform to, or include the larger environmental systems that must now be monitored and managed. Ecosystem and watershed boundaries that serve as the basis for much of our integrated planning today do not conform to the straight lines common to the boundaries of most cities, counties, states and countries. Indeed, many of our more persistent problems of global consequence are beyond the confines of more traditional institutions and call for cooperation and innovation in creating institutions that are inclusive of all relevant stakeholder parties, foster new arrangements among governing bodies and adhere to the strictest notion of adaptive management to deal with the complexity and uncertainty of multi-scale problems over a variety of time periods.

We selected a few passages from Lynch's (1981) classic book on city form so that readers might become acquainted with a standard planning perspective. However, any notion of "ideal" form presumes that we know how to operationalize the ideal settlement patterns that are vital for both human and biological purposes. The papers by Alberti (1999), Ryan & Jensen (2002), and Roundtree (1995) are illustrative for their discussions of both the difficulties and promise of working toward effective integration of human and natural systems and in the development of policies leading to pragmatic impacts. Alberti's (1999) comments focus on providing a conceptual framework for modeling the urban ecosystem and presents the challenge faced in attempting to quantitatively model complex urban systems. Issues of problem definition, multiple actors, time, space, scale, feedback and uncertainty are the necessary requirements to effective modeling. Each remains a challenge of great proportion. Assuming we are able to model relevant human and natural system relationships, there remains the issue of making the link between knowledge and action. The science/policy debate is one of long standing and Ryan & Jensen (2002) offer an important perspective on why this is such a persistent problem and what is necessary to close the gap. A conceptual framework assisting scientists and policy-makers to forge new dialogues is provided. At the core of this discussion is the examination of the two spheres of science and policy and their differences as evidenced by their internal complexities. The framework suggested calls for greater institutional incentives for cooperation and integration and the need for leadership and risk-taking in developing solutions. Roundtree amplifies these points by casting the debate in what he calls the "core and the context" where the ecosystem management, or the scientific core is necessarily dealt with in the policy-management context.

The next four papers in this section provide both general and specific direction for effectively integrating the built environment with ecological aspects of the landscape. For the most part these papers are concerned with problems of site and regional scales that surround urban and urbanizing environments. Their local perspective does not diminish there significance if we are to adhere to the adage of "think globally, act locally". The newly translated Wittig (1995) article asks the question of the form of the ideal city from an ecologic view and provides general guidelines for incorporating ecological knowledge into urban planning development. It provides an entry into the large amount of work German ecologists have done on this topic (e.g., Sukopp et al. 1995). Shafer (1997) focuses on the question of reserves, and their optimum size and configuration for sustaining minimum viable populations, while Soulé (1991) and Marzluff & Ewing (2001) provide specific suggestions for maintaining landscapes for birds and animals. These authors focus on the settlement impacts of fragmentation and the effect that this has on indigenous species and the increased risk for species extinction.

Finally, we present two papers that explore the integration of human and natural systems through the planning and design processes. Arendt (1996) offers a Western perspective on the design of conservation subdivisions that has been the focus of discussion by environmental planners for several decades. What distinguishes Arndt's approach from previous work is not so much the specifics of what to consider and how it should be incorporated into planning and design solutions, but rather how today's institutional environments are more receptive to these solutions, indeed, in many cases requiring the spatial arrangements suggested by Arendt and supported by the science reported in other papers in this compendium. Yokohari et al. (2000) consider open space planning in Asian megacities and note that Western concepts of zoning and greenbelts may not fare well in these cultures. They suggest a careful balance of urban and rural landscapes is necessary to build sustainable Asian megacities. They point out that vegetated urban fringe areas including agriculture provide human sustenance, cultural stimulation, health benefits, and wildlife habitat. They return to early Western views of the "Garden City" and suggest that this old view may in fact be newly appropriate, with slight modification, as a planning guide for today's Asia.

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The Struggle to Govern the Commons

Thomas Dietz, Elinor Ostrom and Paul C. Stern

Abstract Human institutions—ways of organizing activities—affect the resilience of the environment. Locally evolved institutional arrangements governed by stable communities and buffered from outside forces have sustained resources successfully for centuries, although they often fail when rapid change occurs. Ideal conditions for governance are increasingly rare. Critical problems, such as transboundary pollution, tropical deforestation, and climate change, are at larger scales and involve nonlocal influences. Promising strategies for addressing these problems include dialogue among interested parties, officials, and scientists; complex, redundant, and layered institutions; a mix of institutional types; and designs that facilitate experimentation, learning, and change.

Keywords: tragedy of commons · government · human institutions · fisheries · sustainability

In 1968, Hardin [1] drew attention to two human factors that drive environmental change. The first factor is the increasing demand for natural resources and environmental services, stemming from growth in human population and per capita resource consumption. The second factor is the way in which humans organize themselves to extract resources from the environment and eject effluents into it—what social scientists refer to as institutional arrangements. Hardin's work has been highly influential [2] but has long been aptly criticized as oversimplified [3–6].

Hardin's oversimplification was twofold: He claimed that only two state-established institutional arrangements—centralized government and private property—could sustain commons over the long run, and he presumed that resource users were trapped in a commons dilemma, unable to create solutions [7–9]. He missed the point that many social groups, including the herders on the commons that provided the metaphor for his analysis, have struggled successfully against threats of resource degradation by developing and maintaining self-governing institutions [3,10–13]. Although these institutions have not always succeeded, neither have Hardin's preferred alternatives of private or state ownership.

In the absence of effective governance institutions at the appropriate scale, natural resources and the environment are in peril from increasing human population, consumption, and deployment of advanced technologies for resource use, all of which have reached unprecedented levels. For example, it is estimated that "the global ocean has lost more than 90% of large predatory fishes" with an 80% decline typically occurring "within 15 years of industrialized exploitation" [14]. The threat of massive ecosystem degradation results from an interplay among ocean ecologies, fishing technologies, and inadequate governance.

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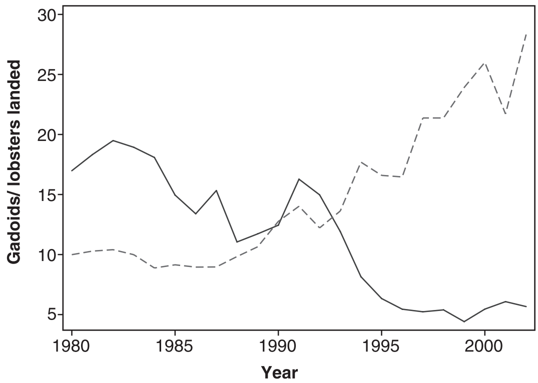

Fig. 1 Comparison of landings of ground fish (gadoids, solid line) and lobster (dashed line) in Maine from 1980 to 2002. Measured in millions of kilograms of ground fish and lobsters landed per year. International fishing in these waters ended with the extended jurisdiction that occurred in 1977 [155]

Inshore fisheries are similarly degraded where they are open access or governed by top-down national regimes, leaving local and regional officials and users with insufficient autonomy and understanding to design effective institutions [15, 16]. For example, the degraded inshore ground fishery in Maine is governed by top-down rules based on models that were not credible among users. As a result, compliance has been relatively low and there has been strong resistance to strengthening existing restrictions. This is in marked contrast to the Maine lobster fishery, which has been governed by formal and informal user institutions that have strongly influenced state-level rules that restrict fishing. The result has been credible rules with very high levels of compliance [17–19]. A comparison of the landings of ground fish and lobster since 1980 is shown in Fig. 1. The rules and high levels of compliance related to lobster appear to have prevented the destruction of this fishery but probably are not responsible for the sharp rise in abundance and landings after 1986.

Resources at larger scales have also been successfully protected through appropriate international governance regimes such as the Montreal Protocol on stratospheric ozone and the International Commission for the Protection of the Rhine Agreements [20–24]. Figure 2 compares the trajectory of atmospheric concentrations of ozone-depleting substances (ODS) with that of carbon dioxide since 1982. The Montreal Protocol, the centerpiece of the international agreements on ozone depletion, was signed in 1987. Before then, ODS concentrations were increasing faster than those of CO_2; the increases slowed by the early 1990s and the concentration appears to have stabilized in recent years. The international treaty regime to reduce the anthropogenic impact on stratospheric ozone is widely considered an example of a successful effort to protect the global commons. In contrast, international efforts to reduce greenhouse gas concentrations have not yet had an impact.

Knowledge from an emerging science of human-environment interactions, sometimes called human ecology or the "second environmental science" [25, 26], is clarifying the characteristics of

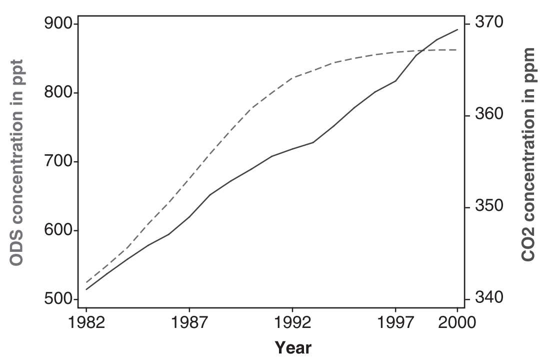

Fig. 2 Atmospheric concentration of CO_2 (solid line, right scale) and three principal ODS (dashed line, left scale). The ODS are chlorofluorocarbons (CFCs) 11, 12, and 113 and were weighted based on their ozone-depleting potential [156]. Data are from [157]. ppt, parts per trillion; ppm, parts per million

institutions that facilitate or undermine sustainable use of environmental resources under particular conditions [6, 27]. The knowledge base is strongest with small-scale ecologies and institutions, where long time series exist on many successes and failures. It is now developing for larger-scale systems. In this review, we address what science has learned about governing the commons and why it is always a struggle [28].

Why a Struggle?

Devising ways to sustain the earth’s ability to support diverse life, including a reasonable quality of life for humans, involves making tough decisions under uncertainty, complexity, and substantial biophysical constraints as well as conflicting human values and interests. Devising effective governance systems is akin to a coevolutionary race. A set of rules crafted to fit one set of socioecological conditions can erode as social, economic, and technological developments increase the potential for human damage to ecosystems and even to the biosphere itself. Furthermore, humans devise ways of evading governance rules. Thus, successful commons governance requires that rules evolve.

Effective commons governance is easier to achieve when (i) the resources and use of the resources by humans can be monitored, and the information can be verified and understood at relatively low cost (e.g., trees are easier to monitor than fish, and lakes are easier to monitor than rivers) [29]; (ii) rates of change in resources, resource-user populations, technology, and economic and social conditions are moderate [30–32]; (iii) communities maintain frequent face-to-face communication and dense social networks—sometimes called social capital—that increase the potential for trust, allow people to express and see emotional reactions to distrust, and lower the cost of monitoring behavior and inducing rule compliance [33–36]; (iv) outsiders can be excluded at relatively low cost from using the resource (new entrants add to the harvesting pressure and typically lack understanding of the rules); and (v) users support effective monitoring and rule enforcement [37–39]. Few settings in the world are characterized by all of these conditions. The challenge is to devise institutional arrangements that help to establish such conditions or, as we discuss below, meet the main challenges of governance in the absence of ideal conditions [6, 40, 41].

Selective Pressures

The characteristics of resources and social interaction in many subsistence societies present favorable conditions for the evolution of effective self-governing resource institutions [13]. Hundreds of documented examples exist of long-term sustainable resource use in such communities as well as in more economically advanced communities with effective, local, self-governing rights, but there are also many failures [6, 11,42–44]. As human communities have expanded, the selective pressures on environmental governance institutions increasingly have come from broad influences. Commerce has become regional, national, and global, and institutions at all of these levels have been created to enable and regulate trade, transportation, competition, and conflict [45, 46]. These institutions shape environmental impact, even if they are not designed with that intent. They also provide mechanisms for environmental governance (e.g., national laws) and part of the social context for local efforts at environmental governance. Larger scale governance may authorize local control, help it, hinder it, or override it [47–52]. Now, every local place is strongly influenced by global dynamics [48, 53–57].

The most important contemporary environmental challenges involve systems that are intrinsically global (e.g., climate change) or are tightly linked to global pressures (e.g., timber production for the world market) and that require governance at levels from the global all the way down to the local [48, 58, 59]. These situations often feature environmental outcomes spatially displaced from their causes and hard-to-monitor, larger scale economic incentives that may not be closely aligned with the condition of local ecosystems. Also, differentials in power within user groups or across scales allow some to ignore rules of commons use or to reshape the rules in their own interest, such as when global markets reshape demand for local resources (e.g., forests) in ways that swamp the ability of locally evolved institutions to regulate their use [60–62].

The store of governance tools and ways to modify and combine them is far greater than often is recognized [6, 63–65]. Global and national environmental policy frequently ignores community-based governance and traditional tools, such as informal communication and sanctioning, but these tools can have significant impact [63, 66]. Further, no single broad type of ownership—government, private, or community—uniformly succeeds or fails to halt major resource deterioration, as shown for forests in multiple countries (supporting online material text, figs. S1 to S5, and table S1).

Requirements of Adaptive Governance in Complex Systems

Providing information. Environmental governance depends on good, trustworthy information about stocks, flows, and processes within the resource systems being governed, as well as about the human-environment interactions affecting those systems. This information must be congruent in scale with environmental events and decisions [48, 67]. Highly aggregated information may ignore or average out local information that is important in identifying future problems and developing solutions.

For example, in 2002, a moratorium on all fishing for northern cod was declared by the Canadian government after a collapse of this valuable fishery. An earlier near-collapse had led Canada to declare a 200-mile zone of exclusive fisheries jurisdiction in 1977 [68, 69]. Considerable optimism existed during the 1980s that the stocks, as estimated by fishery scientists, were rebuilding. Consequently, generous total catch limits were established for northern cod and other ground fish, the number of licensed fishers was allowed to increase considerably, and substantial government subsidies were allocated for new vessels [70]. What went wrong? There were a variety of information-related problems including: (i) treating all northern cod as a single stock instead of recognizing distinct populations with different characteristics, (ii) ignoring the variability of year classes of northern cod, (iii) focusing on offshore-fishery landing data rather than inshore data to “tune” the stock assessment, and (iv) ignoring inshore fishers who were catching ever-smaller fish and doubted the validity of stock assessments [70–72]. This experience illustrates the need to collect and model both local and aggregated information about resource conditions and to use it in making policy at the appropriate scales.

Information also must be congruent with decision makers’ needs in terms of timing, content, and form of presentation [73–75]. Informational systems that simultaneously meet high scientific standards and serve ongoing needs of decision makers and users are particularly useful. Information must not overload the capacity of users to assimilate it. Systems that adequately characterize environmental conditions or human activities with summary indicators such as prices for products or emission permits, or certification of good environmental performance can provide valuable signals as long as they are attentive to local as well as aggregate conditions [76–78].

Effective governance requires not only factual information about the state of the environment and human actions but also information about uncertainty and values. Scientific understanding of coupled human-biophysical systems will always be uncertain because of inherent unpredictability in the systems and because the science is never complete [79]. Decision makers need information that characterizes the types and magnitudes of this uncertainty, as well as the nature and extent of scientific ignorance and disagreement [80]. Also, because every environmental decision requires tradeoffs, knowledge is needed about individual and social values and about the effects of decisions on various valued outcomes. For many environmental systems, local and easily captured values (e.g., the market value of lumber) have to be balanced against global, diffuse, and hard-to-capture values (e.g., biodiversity and the capability of humans and ecosystems to adapt to unexpected events).

Finding ways to measure and monitor the outcomes for such varied values in the face of globalization is a major informational challenge for governance.

Dealing with conflict. Sharp differences in power and in values across interested parties make conflict inherent in environmental choices. Indeed, conflict resolution may be as important a motivation for designing resource institutions as is concern with the resources themselves [81]. People bring varying perspectives, interests, and fundamental philosophies to problems of environmental governance [74, 82–84], and their conflicts, if they do not escalate to the point of dysfunction, can spark learning and change [85, 86].

For example, a broadly participatory process was used to examine alternative strategies for regulating the Mississippi River and its tributaries [87]. A dynamic model was constructed with continuous input by the Corps of Engineers, the Fish and Wildlife Service, local landowners, environmental groups, and academics from multiple disciplines. After extensive model development and testing against past historical data, most stakeholders had high confidence in the explanatory power of the model. Consensus was reached over alternative management options, and the resulting policies generated far less conflict than had existed at the outset [88].

Delegating authority to environmental ministries does not always resolve conflicts satisfactorily, so governments are experimenting with various governance approaches to complement managerial ones. They range from ballots and polls, where engagement is passive and participants interact minimally, to adversarial processes that allow parties to redress grievances through formal legal procedures, to various experiments with intense interaction and deliberation aimed at negotiating decisions or allowing parties in potential conflict to provide structured input to them through participatory processes [89–93].

Inducing rule compliance. Effective governance requires that the rules of resource use are generally followed, with reasonable standards for tolerating modest violations. It is generally most effective to impose modest sanctions on first offenders, and gradually increase the severity of sanctions for those who do not learn from their first or second encounter [39, 94]. Community-based institutions often use informal strategies for achieving compliance that rely on participants' commitment to rules and subtle social sanctions. Whether enforcement mechanisms are formal or informal, those who impose them must be seen as effective and legitimate by resource users or resistance and evasion will overwhelm the commons governance strategy.

Much environmental regulation in complex societies has been "command and control." Governments require or prohibit specific actions or technologies, with fines or jail terms possible for punishing rule breakers. If sufficient resources are made available for monitoring and enforcement, such approaches are effective. But when governments lack the will or resources to protect "protected areas" [95–97], when major environmental damage comes from hard-to-detect "nonpoint sources," and when the need is to encourage innovation in behaviors or technologies rather than to require or prohibit familiar ones, command and control approaches are less effective. They are also economically inefficient in many circumstances (98–100).

Financial instruments can provide incentives to achieve compliance with environmental rules. In recent years, market-based systems of tradable environmental allowances (TEAs) that define a limit to environmental withdrawals or emissions and permit free trade of allocated allowances under those limits have become popular [76, 101, 102]. TEAs are one of the bases for the Kyoto agreement on climate change.

Economic theory and experience in some settings suggest that these mechanisms have substantial advantages over command and control [103–106]. TEAs have exhibited good environmental performance and economic efficiency in the U.S. Sulfur Dioxide Allowance Market intended to reduce the prevalence of acid rain [107, 108] and the Lead Phasedown Program aimed at reducing the level of lead emissions [109]. Crucial variables that differentiate these highly successful programs from less successful ones, such as chlorofluorocarbon production quota trading and the early EPA emission trading programs, include: (i) the level of predictability of the stocks and flows, (ii) the number of users or producers who are regulated, (iii) the heterogeneity of the regulated users, and (iv) clearly defined and fully exchangeable permits [110].

TEAs, like all institutional arrangements, have notable limitations. TEA regimes tend to leave unprotected those resources not specifically covered by trading rules (e.g., bycatch of noncovered fish species) [111] and to suffer when monitoring is difficult (e.g., under the Kyoto protocol, the question of whether geologically sequestered carbon will remain sequestered). Problems can also occur with the initial allocation of allowances, especially when historic users, who may be called on to change their behavior most, have disproportionate power over allocation decisions [76, 101]. TEAs and community-based systems appear to have opposite strengths and weaknesses [101], suggesting that institutions that combine aspects of both systems may work better than either approach alone. For example, the fisheries tradable permit system in New Zealand has added comanagement institutions to complement the market institutions [102, 112].

Voluntary approaches and those based on information disclosure have only begun to receive careful scientific attention as supplements to other tools [63, 77, 113–115]. Success appears to depend on the existence of incentives that benefit leaders in volunteering over laggards and on the simultaneous use of other strategies, particularly ones that create incentives for compliance [77, 116–118]. Difficulties of sanctioning pose major problems for international agreements [119–121].

Providing infrastructure. The importance of physical and technological infrastructure is often ignored. Infrastructure, including technology, determines the degree to which a commons can be exploited (e.g., water works and fishing technology), the extent to which waste can be reduced in resource use, and the degree to which resource conditions and the behavior of humans users can be effectively monitored. Indeed, the ability to choose institutional arrangements depends in part on infrastructure. In the absence of barbed-wire fences, for example, enforcing private property rights on grazing lands is expensive, but with barbed wire fences, it is relatively cheap [122]. Effective communication and transportation technologies are also of immense importance. Fishers who observe an unauthorized boat or harvesting technology can use a radio or cellular phone to alert others to illegal actions [123]. Infrastructure also affects the links between local commons and regional and global systems. Good roads can provide food in bad times but can also open local resources to global markets, creating demand for resources that cannot be used locally [124]. Institutional infrastructure is also important, including research, social capital, and multilevel rules, to coordinate between local and larger levels of governance [48, 125, 126].

Be prepared for change. Institutions must be designed to allow for adaptation because some current understanding is likely to be wrong, the required scale of organization can shift, and biophysical and social systems change. Fixed rules are likely to fail because they place too much confidence in the current state of knowledge, whereas systems that guard against the low probability, high consequence possibilities and allow for change may be suboptimal in the short run but prove wiser in the long run. This is a principal lesson of adaptive management research [31, 127].

Strategies for Meeting the Requirements of Adaptive Governance

The general principles for robust governance institutions for localized resources (Fig. 3) are well established as a result of multiple empirical studies [13, 39, 128–137]. Many of these also appear to be applicable to regional and global resources [138], although they are less well tested at those scales. Three of them seem to be particularly relevant for problems at larger scales.

Analytic deliberation. Well-structured dialogue involving scientists, resource users, and interested publics, and informed by analysis of key information about environmental and human-environment systems, appears critical. Such analytic deliberation [74, 139, 140] provides improved information and the trust in it that is essential for information to be used effectively, builds social capital, and can

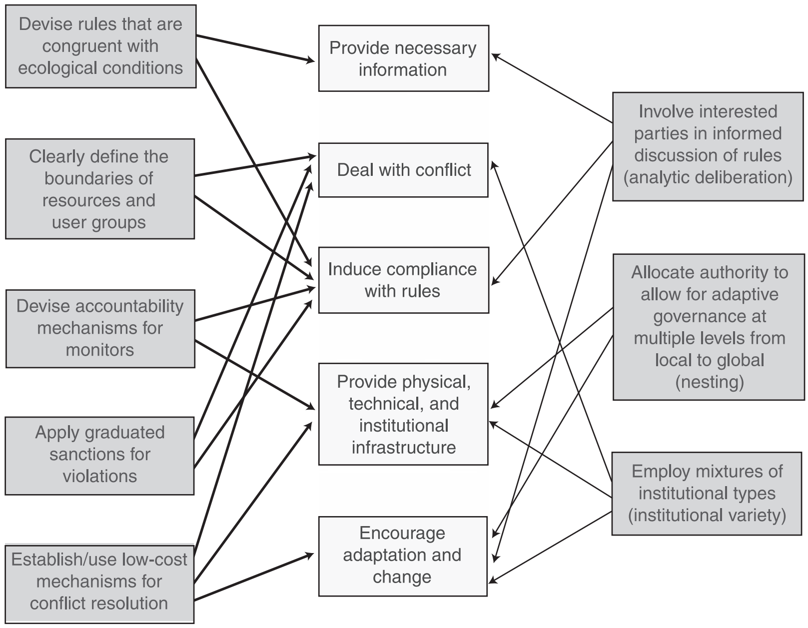

Fig. 3 General principles for robust governance of environmental resources (left and right columns) and the governance requirements they help meet (center column) [13, 158]. Each principle is relevant for meeting several requirements. Arrows indicate some of the most likely connections between principles and requirements. Principles in the right column may be particularly relevant for global and regional problems

allow for change and deal with inevitable conflicts well enough to produce consensus on governance rules. The negotiated 1994 U.S. regulation on disinfectant by-products in water that reached an interim consensus, including a decision to collect new information and reconsider the rule on that basis [74], is an excellent example of this approach.

Nesting. Institutional arrangements must be complex, redundant, and nested in many layers [32, 141, 142]. Simple strategies for governing the world's resources that rely exclusively on imposed markets or one-level, centralized command and control and that eliminate apparent redundancies in the name of efficiency have been tried and have failed. Catastrophic failures often have resulted when central governments have exerted sole authority over resources. Examples include the massive environmental degradation and impoverishment of local people in Indonesian Borneo [95], the increased rate of loss and fragmentation of high-quality habitat that occurred after creating the Wolong Nature Reserve in China [143], and the closing of the northern cod fishery along the eastern coast of Canada partly attributable to the excessive quotas granted by the Canadian government [70].

Institutional variety. Governance should employ mixtures of institutional types (e.g., hierarchies, markets, and community self-governance) that employ a variety of decision rules to change incentives, increase information, monitor use, and induce compliance [6, 63, 117]. Innovative rule evaders can have more trouble with a multiplicity of rules than with a single type of rule.

Conclusion

Is it possible to govern such critical commons as the oceans and the climate? We remain guardedly optimistic. Thirty-five years ago it seemed that the "tragedy of the commons" was inevitable everywhere not owned privately or by a government. Systematic multidisciplinary research has, however, shown that a wide diversity of adaptive governance systems have been effective stewards of many resources. Sustained research coupled to an explicit view of national and international policies as experiments can yield the scientific knowledge necessary to design appropriate adaptive institutions.

Sound science is necessary for commons governance, but not sufficient. Too many strategies for governance of local commons are designed in capital cities or by donor agencies in ignorance of the state of the science and local conditions. The results are often tragic, but at least these tragedies are local. As the human footprint on the Earth enlarges [144], humanity is challenged to develop and deploy understanding of large-scale commons governance quickly enough to avoid the large-scale tragedies that will otherwise ensue.

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Modeling the Urban Ecosystem: A Conceptual Framework

M. Alberti

Abstract In this paper I build on current research in urban and ecological simulation modeling to develop a conceptual framework for modeling the urban ecosystem. Although important progress has been made in various areas of urban modeling, operational urban models are still primitive in terms of their ability to represent ecological processes. On the other hand, environmental models designed to assess the ecological impact of an urban region are limited in their ability to represent human systems. I present here a strategy to integrate these two lines of research into an urban ecological model (UEM). This model addresses the human dimension of the Puget Sound regional integrated simulation model (PRISM)—a multidisciplinary initiative at the University of Washington aimed at developing a dynamic and integrated understanding of the environmental and human systems in the Puget Sound. UEM simulates the environmental pressures associated with human activities under alternative demographic, economic, policy, and environmental scenarios. The specific objectives of UEM are to: quantify the major sources of human-induced environmental stresses (such as land-cover changes and nutrient discharges); determine the spatial and temporal variability of human stressors in relation to changes in the biophysical structure; relate the biophysical impacts of these stressors to the variability and spatial heterogeneity in land uses, human activities, and management practices; and predict the changes in stressors in relation to changes in human factors.

Keywords: urban model · ecological model · integrated model · puget sound · simulation · land cover change

1 Introduction

Planning agencies worldwide are increasingly challenged by the need to assess the environmental implications of alternative urban growth patterns—and policies to control them—in a comprehensive manner. Urban growth leads to rapid conversion of land and puts increasing pressure on local and global ecosystems. It causes changes in water and energy fluxes. Natural habitats are reduced and fragmented, exotic organisms are introduced, and nutrient cycles are severely modified. Although impacts of urban development often seem local, they cause environmental changes at larger scales. Assessments of urban growth that are timely and accurate, and developed in a transparent manner, are crucial to achieve sound decisions. However, operational urban models designed to analyze or predict the development of urban areas are still primitive in their ability to represent ecological processes and urban ecosystem dynamics. Though important progress has been made in various

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areas of urban modeling (Wegener, 1994; 1995), only a few scholars have attempted to integrate the environmental dimension. The majority of these models are designed to answer a set of fundamental but limited planning questions relevant to housing (Anas, 1995; Anas and Arnott, 1991; Kain and Apgar, 1985), land use (Landis, 1992; 1995; Prastacos, 1986; Waddell, 1998), transportation (Boyce, 1986; Kim, 1989) and in some cases the interactions among them (de la Barra, 1989; Echenique et al, 1990; Mackett, 1990; Putman, 1983; 1991; Wegener, 1983).

On the other hand, the environmental models designed to assess the ecological impact of an urban region are limited in their ability to represent human systems. These models represent people as static scenarios of land uses and economic activities and predict human-induced disturbances from aggregated measures of economic development and urban growth. Only with the increasing attention paid to the role of human activities in global environmental change has the need emerged to represent more explicitly human systems in environmental models. Whereas integrated assessment modeling can be traced back to the late 1960s (Forrester, 1969; Meadows et al, 1972), the first generation of operational integrated models has emerged only in the mid-1980s. During the last decade, integrated assessment modeling has been proposed as a new approach to link biophysical and socioeconomic systems in assessing climate change (Dowlatabadi, 1995). At present more than thirty integrated assessment models (IAMs) have been developed (Alcamo, 1994; Dowlatabadi, 1995; Rotmans et al., 1995). The focus of current IAMs is global; however, a new generation of spatially explicit regional integrated models is now emerging (Maxwell and Costanza, 1995). These models have started to treat human decisions explicitly but are still too limited in the representation of human behavior and the heterogeneity of urban land uses (Alberti, 1998).

Recent progress in the study of complex systems (Schneider and Kay, 1994) and the evolution of computer modeling capabilities (Brail, 1990) have made possible a more explicit treatment of the link between human and ecological systems. The development of GIS has provided the capability to integrate spatial processes. However, the greatest challenge for integrating urban and environmental modeling will be in interfacing the various disciplines involved. Urban subsystems have been studied for several decades but progress in urban-ecological modeling has been limited because of the difficulty in integrating the natural and social sciences. A recent National Science Foundation workshop on urban processes pointed out that ecologists, social scientists, and urban planners will need to work together to make their data, models, and findings compatible with one another and to identify systematically where fruitful clusters of multidisciplinary research problems can be developed (Brown, 1997). Such an approach can offer a new perspective on modeling urban systems.

In this paper I build on research in urban and ecological simulation modeling to develop an integrated urban-ecological modeling framework. This framework is part of a current effort to develop an urban – ecological model (UEM) at the University of Washington as part of the Puget Sound regional integrated simulation model (PRISM). UEM simulates the environmental impacts associated with human activities under alternative demographic, economic, policy, and environmental scenarios. Its objectives are to:

(1) Quantify the major sources of human-induced environmental stresses (such as land-cover changes and nutrient discharges);
(2) Determine the spatial and temporal variability of human stressors in relation to changes in the biophysical structure;
(3) Relate the biophysical impacts of these stressors to the variability and spatial heterogeneity in land uses, human activities, and management practices; and
(4) Predict the changes in stressors in relation to changes in human factors.

The development of an integrated urban-ecological framework has both scientific and policy relevance. It provides a basis for developing integrated knowledge of the processes and mechanisms that govern urban ecosystem dynamics. It also creates the basis for modeling urban systems and provides planners with a powerful tool to simulate the ecological impacts of urban development patterns.

2 The Urban Ecosystem

Early efforts to understand the interactions between urban development and environmental change led to the conceptual model of cities as urban ecosystems (Boyden et al, 1981; Douglas, 1983; Duvigneaud, 1974; Odum, 1963; 1997; Stearns and Montag, 1974). Ecologists have described the city as a heterotrophic ecosystem highly dependent on large inputs of energy and materials and a vast capacity to absorb emissions and waste (Boyden et al, 1981; Duvigneaud, 1974; Odum, 1963). Wolman (1965) applied an 'urban metabolism' approach to quantify the flows of energy and materials into and out of a hypothetical American city. Systems ecologists provided formal equations to describe the energy balance and the cycling of materials (Douglas, 1983; Odum, 1983). Although these efforts have never been translated into operational simulation models, they have laid out the basis for urban-ecological research. Urban scholars were rightly skeptical about the attempts to integrate biological and socioeconomic concepts into system dynamics models. None of these models represented explicitly the processes by which humans affect or are affected by the urban environment. At best, human behavior was reduced to a few differential equations. These models simplified the interactions of natural and social systems so much that they could provide little useful insight for planners and decisionmakers. Since then, however, urban and ecological research has made important progress with respect to understanding how urban ecosystems operate and how they differ from natural ecosystems.

Urban-ecological interactions are complex. Urban ecosystems consist of several interlinked subsystems—social, economic, institutional, and environmental—each representing a complex system of its own and affecting all the others at various structural and functional levels. Urban development is a major determinant of ecosystem structure and influences significantly the functioning of natural ecosystems through (a) the conversion of land and transformation of the landscape; (b) the use of natural resources; and (c) the release of emissions and waste. The earth's ecosystems also provide (d) important services to the human population in urban areas. Thus (e) environmental changes occurring at the local, regional, and global scale—such as the contamination of watersheds, loss of biodiversity, and changes in climate—affect human health and well-being. Humans respond to environmental change through (f) management strategies (fig. 1).

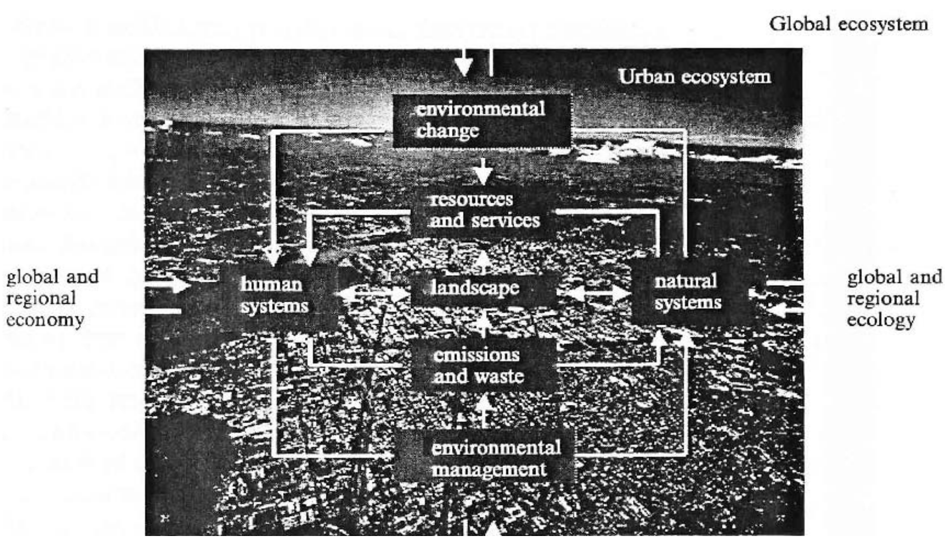

Fig. 1 Human – natural systems interactions

2.1 Human Systems

Human drivers are dominant in urban ecosystem dynamics. Major human driving forces are demographics, socioeconomic organization, political structure, and technology. Human behaviors—the underlying rationales for the actions that give rise to these forces—directly influence the use of land and the demand for and supply of resources (Turner, 1989). In urban areas these forces combine to affect the spatial distribution of activities and ultimately the spatial heterogeneity of natural process and disturbances. It is increasingly clear to both social (Openshaw, 1995) and natural (Pickett et al., 1994) scientists that it is absurd to model the urban ecosystem without explicitly representing humans in them. Would ecologists exclude other species from models of natural ecosystems? However, as Pickett et al. (1997) point out, simply adding humans to ecosystems without representing the way they function is not an adequate alternative. Today, social and natural scientists have the tools to explore the richness of interactions between the social and ecological functions of the human species.

Representing human actors and their institutions in models of urban ecosystems will be an important step towards representing more realistically the human dimension of environmental change. Many of the human impacts on the physical environment are mediated through social, economic, and political institutions that control and order human activities (Kates et al, 1990). Also, humans consciously act to mitigate these impacts and build the institutional settings to promote such actions. They adapt by learning both individually and collectively. How can these dimensions be represented? Lynch (1981, page 115) suggested that "a learning ecology might be more appropriate for human settlement since some of its actors, at least, are conscious, and capable of modifying themselves and thus changing the rules of the game", for example by restructuring materials and switching the path of energy flows. Humans, like other species, respond to environmental change but in a more complex way.

2.2 Natural Systems

Environmental forces—such as climate, topography, hydrology, land cover, and human-induced changes in environmental quality—are important drivers of urban systems. Moreover, natural hazards—such as hurricanes, floods, and landslides—can cause significant perturbations in social systems. Most models of human systems, however, simply ignore these forces. In urban models, biophysical processes are at best included as exogenous variables and treated as constants. This is a severe limitation because human decisions are directly related to environmental conditions and changes. Surprisingly, urban modelers cannot remove the behavior of the job market or degradation of housing stocks from their models but can represent the dynamics of urban systems without considering the degradation of the environment and depletion of natural resources.

As we cannot simply add humans to ecological models, representing biophysical processes in urban models will require going beyond simply adding environmental variables to existing urban models. A number of models currently extend their modules to include changes in environmental variables such as air quality and noise (Wegener, 1995). However, these models may misrepresent complex ecological responses. Before we can model these responses, we need to recognize explicitly the properties of ecosystem organization and behavior that govern them. According to Holling (1978, pages 25–26) four properties of ecological systems determine how they respond to change. First, parts in ecological systems are connected to each other in a selective way that has implications for what should be measured. Second, events are not uniform over space, which has implications for how intense impacts will be and where they will occur. Third, sharp shifts in behavior are natural for many ecosystems. Fourth, variability, not constancy, is a feature of ecological systems that contributes to their self-correcting capacity.

2.3 Integrated Modeling

In modeling the interactions between human and natural systems, we need to consider that many factors work simultaneously at various levels. Simply linking these models in an 'additive' fashion may not adequately represent system behavior because interactions occur at levels that are not represented (Pickett et al., 1994). On the basis of hierarchy theory, Pickett et al. (1994) argue that the consideration of interactions only at the upper level may provide statistical relationship but cannot help explain or predict important feedback for future conditions. This is particularly true in urban ecosystems because urban development controls the ecosystem structure in complex ways. Land-use decisions affect species composition directly through the introduction of species and indirectly through the modification of natural disturbance agents. Production and consumption choices determine the level of resource extraction and generation of emissions and waste. Decisions about investing in infrastructures or adopting control policies may mitigate or exacerbate these effects. Because ecological productivity controls the regional economy, interactions between local decisions and ecological processes at the local scale can result in large-scale environmental change.

We also need to challenge the implicit assumption of most models that decisions are made by one single decisionmaker at one point in time. Urban development is the outcome of dynamic interactions among the choices of many actors, including households, businesses, developers, and governments (Waddell, 1998). These actors make decisions that determine and alter the patterns of human activities and ultimately affect environmental change. Their decisions are interdependent; for example, housing location is affected by employment activity and affects retail activity and infrastructure, which in turn affect housing development.

Human and natural systems, including their equilibrium conditions, change over time. One major problem in describing their relationships is that they operate at very different temporal and spatial scales. The lag times between human decisions and their environmental effects further complicate any attempt to understand these interactions. Moreover, the environmental effects of human actions may also be distanced over space (Holling, 1986). Simulating the behavior of urban-ecological systems requires not only an explicit consideration of the temporal and spatial dynamics of these systems, but also achieving consistency across the different temporal and spatial scales at which various processes operate.

Another source of difficulty in spelling out these interactions is their cumulative and synergistic impacts. In general, environmental impacts become important when their sources are grouped closely enough in space or time to exceed the ability of the natural system to remove or dissipate the disturbances (Clark, 1986). Human stresses in cities may cross thresholds beyond which they may irrevocably damage important ecological functions. In most ecological systems, processes operate in a stepwise rather than a smoothly progressive fashion over time (Holling, 1986). Sharp shifts in behavior are natural. This property of ecosystems requires the consideration of resilience: the amount of disturbance a system can absorb without changing its structure or behavior.

In modeling urban-ecological systems we also need to consider feedback mechanisms between the natural and human systems. These are control elements that can amplify or regulate a given output. At the global level, an example of negative feedback in the biosphere described by ecologists is the homeostatic integration of biotic and physical processes that keeps the amount of CO_2 in the air relatively constant. Feedback loops—both positive and negative—between human and environmental systems are not completely understood. We know that human decisions leading to the burning of fossil fuels and land-use change affect the carbon cycle, and that in turn the associated climate changes will affect human choices, but the nature of these interactions remains controversial. In particular, the feedback of environmental change on human decisions is difficult to represent because environmental change affects all people independently of who has caused the environmental impact in the first place, whereas the impact of each individual decisionmaker on the environment depends on the choices of others (Ostrom, 1991).

Modeling urban-ecological systems will require special attention to uncertainty. Uncertainty can arise from limited understanding of a given phenomenon, systematic and random error, and subjective judgment. Change in natural systems can occur in abrupt and discontinuous ways, and responses can be characterized by thresholds and multiple domains of stability. The knowledge of the environmental systems is always incomplete and surprise is inevitable (Holling, 1995). The explicit characterization and analysis of uncertainty should be a central focus of modeling integration.

3 The Environmental Dimension in Urban Models

Although extensive urban research has focused on the dynamics of urban systems, it has been described only partially through numerical models. Most operational urban models focus on a few subsystems such as housing, employment, land use, and transportation, with a limited set of elements influencing their dynamics. These models predict the spatial distribution of activities based on simple spatial interaction mechanisms and economic axioms. No operational urban models have attempted to describe the interactions between urban and environmental processes in a systematic way. Recently a few modelers have started to address a number of direct impacts of human activities, such as air pollution and noise, on the environment. However, as the idealized urban model proposed by Wegener (1994) depicts quite well, only unidirectional links between urban systems and the environment have been conceived in urban modeling. Today a vast literature synthesizes the theoretical and methodological foundation of urban simulation models (Batty and Hutchinson, 1983; Harris, 1996; Mackett, 1985; 1990; Putman, 1983; 1995; Wegener, 1994; 1995; Wilson et al, 1977; 1981). In this section I draw on this literature to explore how environmental variables are considered and how environmental processes are represented.

Operational urban models can be classified according to the approach they use to predict the generation and spatial allocation of activities or according to the solution proposed to a variety of model design questions (see Table 1, over). It is difficult to classify the vast literature on urban modeling because of the great variability in emphasis that authors place on theory, techniques, and applications. Moreover, the various approaches are interrelated in complex ways. Six major classes of operational models discussed in the literature are relevant here: those relying on gravity, the economic market, optimization, input – output, microsimulation, and cellular automata.

3.1 Gravity, Maximum Entropy, and Discrete-choice Models

The dominant approach in urban modeling can be traced to Lowry's (1964) model, a simple iterative procedure in which nine equations are used to simulate the spatial distribution of population, employment, service, and land use. The model is based on the simple hypothesis that residences gravitate toward employment locations. Two schools of research have provided a statistical basis for the gravity model, guided by Wilson (1967, the entropy-maximizing principle) and McFadden (1973, utility maximization). The results obtained by the two methods were later shown to be equivalent (Anas, 1983). The models most often used by planning agencies in the USA—the disaggregated residential allocation model (DRAM) and the employment allocation model (EMPAL)—are derivatives of Lowry's model using maximum entropy formulation. Developed by Putman (1979), and incrementally improved since the early 1970s, DRAM and EMPAL are currently in use in fourteen US metropolitan areas (Putman, 1996). The integrated transportation land-use package (ITLUP), also developed by Putman (1983), provides a feedback mechanism to integrate DRAM, EMPAL, and various components of the urban transportation planning system (UTPS) models implemented in most metropolitan areas. Although these models substantially improve upon

Table 1 Urban models

Model	Subsystems	Theory or approach	Population or sectors
Clarke	Land use and/or cover	Complex systems Cellular automata Monte Carlo simulation	Aggregated
CUF2	Population Employment Housing Land use	Random utility Multinomial logit	Aggregated
IRPUD	Population Employment Housing Land use	Random utility Network equilibrium Land-use equilibrium Monte Carlo microsimulation	Partially disaggregated
ITLUP	Population Employment Land use Travel	Random utility Maximization Network equilibrium	Partially disaggregated
Kim	Population Employment Transport Travel	Random utility General equilibrium Input – output	Aggregated
MASTER	Population Employment Housing Land use Travel	Random utility Maximization Monte Carlo microsimulation	Disaggregated
MEPLAN	Population Employment Housing Land use Transport Travel	Random utility Maximization Market clearing Input – output	Aggregated
POLIS	Population Employment Housing Land use Travel	Random utility Optimization	Aggregated
TRANUS	Population Employment Housing Land use Transport Travel	Random utility Network equilibrium Land-use equilibrium Input – output	Aggregated
UrbanSim	Population Employment Housing Land use	Random utility Partial equilibrium Multinomial logit	Partially disaggregated

Model	Time	Space	Environmental factors	Source
Clarke	Dynamic	Dynamic Grid cell	Land cover Topography Hydrography	Clarke et al, 1997
CUF2	Static	Static 100 × 100 m grid cell	Percent slope	Landis and Zhang, 1998a; 1998b

Table 1 (continued)

Model	Time	Space	Environmental factors	Source
IRPUD	Quasidynamic	Static Zone	Zone space constraints CO_2 emissions by transport	Wegener, 1995
ITLUP	Static	Static Zone	Zone space constraints Mobile source emissions	Putman, 1983; 1991
Kim	Static	Static Zone	Zone space constraints	Kim, 1989
MASTER	Quasidynamic	Static Zone	Zone space constraints	Mackett, 1990
MEPLAN	Static	Static Zone	Zone space constraints	Echenique et al, 1990
POLIS	Static	Static Zone	Zone space constraints	Prastacos, 1986
TRANUS	Static	Static Zone	Zone space constraints	de la Barra, 1989
UrbanSim	Quasidynamic	Static Parcels	Topography Stream buffers Wetlands 100 years floodplain area	Waddell, 1998

the initial Lowry model, they are based on the same simple assumption. No environmental variables are used in determining the spatial distribution of residence. The allocation of residential and employment activities must of course meet physical constraints and planning restrictions within the available zones. However, other than these constraints no other environmental considerations are included in the equation.

3.2 Economic Market-based Models

A second urban modeling approach is based on the work of Wingo (1961) and Alonso (1964), who introduce the notion of land-rent and land-market clearing. Wingo was the first to describe the urban spatial structure in the framework of equilibrium theory. Given the location of employment centers, a particular transportation technology, and a set of households, his model determines the spatial distribution, value, and extent of residential land requirements under the assumption that landowners and households both maximize their return. Wingo uses demand, whereas Alonso uses bid-rent functions to distribute the land to its users. The aim of both models is to describe the effects of the residential land market on location. Under this approach, households are assumed to maximize their utility and select an optimum residential location by trading off housing prices and transportation costs. The trade-offs are represented in a demand or bid-rent functional form which describes how much each household is willing to pay to live at each location. Anas (1983) introduced discrete-choice behavior into models with economically specified behavior and market clearing. Two models that use this approach are UrbanSim developed by Waddell (1998), and CUF2 developed by Landis and Zhang (1998a; 1998b). Both models are based on random utility theory and make use of logit models to implement key components. However, they differ in a substantial way. UrbanSim models the key decisionmakers—households, businesses, and developers—and simulates their choices that impact urban development. It also simulates the land market as the interaction of demand and supply with prices of land and buildings adjusting to clear the market. UrbanSim simulates urban development as a dynamic process as opposed to a cross-sectional or equilibrium approach. CUF2 models land-use transition probabilities based on a set of site and community characteristics such as population and employment growth, accessibility, and original use in the site and surrounding sites.

As indicated in Table 1, most current operational models are based on an economic market-based approach and rely on random utility or discrete-choice theory. In these models, environmental variables are not part of the equation, except for environmental constraints. The value of the ecological services—such as clean air, clean water, and flood control—that ecosystems provide to households are not reflected in market prices. This is a severe limitation, because changes in environmental quality and other ecological services provided by ecosystems will affect the market behavior of the households (Mäler et al, 1994).

3.3 Mathematical Programming-based Models

A third approach to describing urban activity allocation is based on optimization theory. By using mathematical programming, these models design spatial interaction problems in order to optimize an objective function that includes transportation and activity establishment costs. Herbert and Stevens (1960) used linear programming to simulate the market mechanisms that affect location. Wheaton (1974) developed an optimization model by using nonlinear programming. More recently Boyce et al (1993) and Boyce and Southworth (1979) have explored the options for integrating spatial interactions of residential, employment, and travel choices within a single optimized modeling framework. The projective optimization land-use system (POLIS) developed by Prastacos (1986) is one of the few optimization land-use models used in planning practice. This model, which has been implemented in the San Francisco Bay area, seeks to maximize both the location surplus and the spatial agglomeration benefits of basic employment sectors. As in previous models, only land availability is included as a determinant of employment allocation to zones.

3.4 Input – Output Models

Another important contribution from economic theory to urban modeling is the spatially disaggregated intersectoral input – output (I – O) approach, developed initially by Leontief (1967). The approach provides a framework for disaggregating economic activities by sector and integrating them into urban spatial interaction models. This transforms the basic structure of an I – O table, allowing the modeler to estimate the direct and indirect impacts of exogenous change in the economy on a spatially disaggregated scale. Operational urban models that use such an approach include MEPLAN, TRANUS, and the models developed by Kim (1989). MEPLAN includes three modules: LUS, the land-use model; FRED, which converts production and consumption into flows of goods and services; and TAS, a transportation model which allocates the transport of goods and passengers to travel modes and routes. The land-use component of MEPLAN is based on a spatial disaggregation of production and consumption factors that include goods, services, and labor. Total consumption is estimated by using a modified I – O framework subsequently converted into trips. MEPLAN, TRANUS, and Kim's models use I – O tables to generate interregional flows of goods. MEPLAN uses the results of the I – O framework to evaluate environmental impacts. I – O models have been extended to include environmental variables and incorporate pollution multipliers, but no urban model has attempted to implement this approach for describing economic-ecological interactions. The regional applications of such an approach have encountered various difficulties related to the specification of the ecological interprocess matrix and the assumption of fixed coefficients. A major limitation is that inputs and outputs are measured in values as opposed to physical flows.

3.5 Microsimulation

One major limitation in the way most urban models represent the behavior of households and businesses stems from the fact that they are aggregated and static. Individuals behave in ways that are influenced by their characteristics and the opportunities from which they choose. Without the explicit representation of these individuals it is impossible to predict the trade-offs they make between jobs, residential locations, or travel modes. A distinct approach to model the behavior of individuals is microanalytic simulation that explicitly represents individuals and their progress through a series of processes (Mackett, 1990). Microsimulation is a modeling technique that is particularly suitable for systems where decisions are made at the individual unit level and where the interactions within the system are complex. In such systems, the outcomes produced by altering the system can vary widely for different groups and are often difficult to predict. In microsimulation the relationships between the various outcomes of decision processes and the characteristics of the decisionmaker can be defined by a set of rules or by a Monte Carlo process. Furthermore, the actions of a population can be simulated through time and incorporate the dynamics of demographic change. An example is the microanalytical simulation of transport, employment, and residence (MASTER) model developed by Mackett (1990). The model simulates the choices of a given population through a set of processes. The outcome of each process is a function of the characteristics of the household or business, the set of available choices, and a set of constraints. This approach is applied less extensively in Wegener's (1982) Dortmund model. Although these models do not explicitly use microsimulation for modeling environmental impacts, it is clear that the greater disaggregation of the actors and behaviors has enormous advantages for modeling consumer behavior and environmental impacts.

3.6 Cellular Automata

The use of cellular automata (CA) has been proposed to model spatially explicit dynamic processes not currently represented in urban models (Batty and Xie, 1994; Couclelis, 1985; White et al, 1997; Wu, 1998). Existing operational models are spatially aggregated and, even when they use or produce spatially disaggregated data, they rely on simple spatial geometric processing. A number of modelers have stressed the need to represent more realistically the spatial behavior of urban actors (White and Engelen, 1997). CA consist of cells arranged in a regular grid that change state according to specific transition rules. These rules define the new state of the cells as a function of their original state and local neighborhood. Clarke et al (1997) have developed a CA urban growth model as part of the Human-Induced Land Transformations Project initiated by the US Geological Survey. The model aims to examine the urban transition in the San Francisco Bay area from a historical perspective and to predict regional patterns of urbanization in the next 100 years (Clarke et al, 1997). These predictions are then used as a basis to assess the ecological and climatic impacts of urban change. There are four types of growth: spontaneous, diffusive, organic, and road-influenced. Five factors regulate the rate and nature of growth: a diffusion factor which determines the dispersiveness; a breed coefficient which specifies the likelihood of a settlement to begin its growth cycle; a spread coefficient which regulates growth of existing settlements; a slope resistance factor which influences the likelihood of growth on steeper slopes; and a road gravity factor which attracts new settlements close to roads.

4 The Human Dimension in Environmental Models

Environmental models have been developed for several decades to simulate atmospheric, land, and ecosystem dynamic processes, and to help assess the effects of various natural and human-induced disturbances. However, the use of these models in environmental management has become widespread only in the last three decades (Jørgensen et al, 1995). Since the early 1970s major environmental problems such as eutrophication and the fate of toxic substances have attracted the attention of environmental modelers, and very complex models were developed. More recently, the prospect of major changes in the global environment has presented the scientific community with the challenge of modeling the interactions between human and ecological systems in an integrated way. Over these decades a rich literature on environmental models has developed, but this is well outside the scope of this paper. In this review I focus on the treatment of the human dimension in these models (Table 2, over).

Table 2 Environmental models

Model	Class	Media or subsystems	Scale
NCAR	Ocean – climate general circulation model	Climate – ocean	Global
CMAQ	Atmospheric model	Meteorological emission, Chemistry transport	Local or regional
UAM	Atmospheric model	Photochemical processes	Local or regional
OBM	Biogeochemical model	Terrestrial biosphere	Global
HRBM	Biogeochemical model	Terrestrial biosphere	Regional
DHSVM	Distributed hydrology soil vegetation model	Hydrology	Regional
JABOWA/FORET	Population – community dynamic model	Trees	Local
CENTURY	Biogeochemical model	Nutrient cycles	Local
GEM	Process-oriented ecological model	Ecosystems	Local
PLM	Process-oriented landscape model	Terrestrial landscape	Regional
IMAGE 2	Process-oriented integrated simulation model	Energy – industry Terrestrial Environment Atmosphere – ocean	Global, 13 regions
ICAM-2	Optimization – simulation model	Climate Economy Policy	Global, 7 regions
RAINS	Optimization – simulation model	Emissions Atmospheric transport Soil acidification	Continental, Europe
TARGETS	Integrated simulation model	Population or health Energy or economics Biophysics, land, soils, or water	Global, 6 regions

Model	Time	Space	Human factors	Source
NCAR	Dynamic Minutes 100 years	Dynamic $4.5^\circ \times 7.5^\circ$ $(\text{latitude} \times \text{longitude})$ 9 layers	CO_2 concentration scenarios	Washington and Meehl, 1996
CMAQ	Dynamic 8-hour to 72-hour period	Dynamic Variable 3-D grid	Emissions of atmospheric pollutants	Novak et al, 1995
UAM	Dynamic 8-hour to 72-hour period	Dynamic Variable 3-D grid	Emissions of photochemical pollutants	Morris and Meyers, 1990
OBM	Dynamic 1 year	Dynamic $2.5^\circ \times 2.5^\circ$	Land use CO_2 concentration scenarios	Esser, 1991
HRBM	Dynamic 6 days	Dynamic $0.5^\circ \times 0.5^\circ$	Land use CO_2 concentration scenarios	Esser et al, 1994

Table 2 (continued)

Model	Time	Space	Human factors	Source
DHSVM	Dynamic Hours	Dynamic 30–100 m	Land cover	Wigmosta et al, 1994
JABOWA/FORET	Dynamic Up to 500 years 1 year	Dynamic $10 \times 10\,\mathrm{m}$ grid	Land cover	Botkin, 1984
CENTURY	Dynamic 1 month Thousands of years	Dynamic $1 \times 1\,\mathrm{m}$ grid cell	Land cover CO_2 concentration	Parton et al., 1987
GEM	Dynamic 12 hours	Dynamic 1 km cell	Land cover	Fitz et al, 1999
PLM	Dynamic 1 week	Dynamic 200 m grid 1 km grid	Land cover	Costanza et al, 1995
IMAGE 2	Dynamic 1 day to 5 years	Dynamic Variable from $0.5^\circ \times 0.5^\circ$ grid to region	Land use CO_2 emissions	Alcamo, 1994
ICAM-2	Dynamic 5 years	Static Latitude bands	Explicit treatment of uncertainties	Dowlatabadi and Ball, 1994
RAINS	Dynamic 1 year	Static $150\,\mathrm{km} \times 150\,\mathrm{km}$ in deposition submodel and $0.5^\circ \times 1.0^\circ$ impact submodel	Energy use Sulfur emissions	Alcamo et al, 1990
TARGETS	Dynamic 1 year	Static Regions	Energy use Water use Emissions Land cover	Rotmans et al., 1994

4.1 Climate and Atmospheric Models

Atmospheric models can be classified according to the scale of the atmospheric processes they represent. At the global scale, sophisticated coupled atmospheric – ocean general circulation models (AOGCM) predict climate conditions by considering simultaneously the atmosphere and the ocean (Washington and Meehl, 1996). Using a set of climate parameters (that is, solar constant) and boundary conditions (that is, land cover, topography, and atmospheric composition), these models determine the rate of change in climatic variables such as the temperature, precipitation, surface pressure, and soil moisture associated with alternative scenarios of CO_2 concentrations. These models are currently being used by the Intergovernmental Panel on Climate Change (IPCC) to assess the impact of alternative greenhouse-gases emission scenarios up to the year 2100.

Regional models have been developed primarily to tackle the issue of acid rain. Aggregated emissions of sulfur and nitrogen compounds, estimated on the basis of emissions factors of point, area, and mobile sources, serve as inputs for long-range transport models which predict the emissions and regional distribution of acid compounds. Two regional air-quality models developed by the Environmental Protection Agency (EPA) are the regional oxidant model (ROM) (Young et al, 1989) and the regional acid-deposition model (RADM) (Chang et al, 1990).

Emission factors for criteria pollutants are also used as inputs for urban atmospheric models. The EPA's urban airshed model (UAM) is a three-dimensional photo-chemical grid model designed to simulate the relevant physical and chemical processes affecting the production and transport of tropospheric ozone (Morris and Meyers, 1990). The basis for the UAM is a mass balance equation in which all of the relevant emissions, transport, diffusion, chemical reactions, and removal processes are expressed in mathematical terms (Morris and Meyers, 1990). A more recent model developed by EPA is the community multiscale air-quality (CMAQ) model. CMAQ is a third generation air-quality model that treats multiple pollutants simultaneously up to continental scales and incorporates feedbacks between chemical and meteorological components. The CMAQ modeling system contains three modeling components: a meteorological model, emission models for human-made and natural emissions, and a chemistry-transport model. The target-grid resolutions and domain sizes for CMAQ range spatially and temporally over several orders of magnitude. In these models human decisions are represented by emission factors developed by the EPA (Novak et al, 1995).

4.2 Biogeochemical Models

A number of global models aim to simulate the impacts of human activities on biogeochemical cycles—the continuous cycling of carbon, nitrogen, and sulfur through the biosphere which sustains life. Two examples are the Osnabrück biosphere model (OBM) and the high-resolution biosphere model (HRBM) developed in Germany (Esser, 1991; Esser et al, 1994). These spatially explicit models simulate the dynamics of the carbon cycle through the terrestrial biosphere in response to climate and CO_2 forcing. OBM uses a grid cell of $2.5^\circ \times 2.5^\circ$ $(\mathrm{latitude} \times \mathrm{longitude})$ and an annual time step. HRBM has a greater spatial resolution ($0.5^\circ \times 0.5^\circ$) and a finer (6 days) time step. They compute the storage and transfer of carbon from each cell by using a series of rate constants and coefficients. In these models human impacts are generated through scenarios of land-use change and CO_2 concentrations. However, these models do not represent more complex interactions between human behavior and biogeochemical cycles. For example, land-use decisions affect the carbon cycle not only directly but also through its impact on transportation patterns and related CO_2 emissions in the atmosphere.

4.3 Hydrological Models

Human-induced changes in water and sediment fluxes have been modeled through run-off models. Human activities can cause four major impacts on the hydrological cycle: floods, droughts, changes in surface and groundwater regimes, and water pollution (Rogers, 1994). Primarily, changes in land uses and channelization cause these impacts. The water balance model developed by Vorosmarty and Moore (1991) is an example of a regional model used to predict changes in the water cycle. It uses spatially explicit biophysical data including precipitation, temperature, vegetation, soils, and elevation to predict run-off, evapotranspiration, river discharge, and floodplain inundation at a grid resolution of $0.5^\circ \times 0.5^\circ$. Another example is the distributed hydrology soil vegetation model (DHSVM), a spatially distributed hydrologic model developed by Wigmosta et al (1994) for use in complex terrain. The DHVSM represents dynamically the spatial distribution of land-surface processes (that is, soil moisture, snow cover, evapotranspiration, and run-off) at high resolution (typically 30–100 m). While their aim is to assess the hydrologic effects of land-use decisions, the human dimension in these models is represented by static scenarios of climate and land-use changes.

4.4 Ecosystem Models

Ecosystem dynamics can be simulated through three classes of models: plant physiology, population – community, and ecosystem (Melillo, 1994). Plant physiology models are used to predict plant growth and water balance and are particularly useful in the analysis of plant responses to climate change and CO_2. Population – community models simulate the dynamics of tree growth on small forest patches as influenced by limiting factors, space, and stand structure. Ecosystem models are process-based models that take into account carbon and nutrient fluxes. These models, such as CEN-

TURY and GEM, simulate changes in ecosystem structure and function over a period from decades to centuries. CENTURY computes the flow of carbon, nitrogen, phosphorus, and sulfur through four compartments: soil organic matter, water, grassland, and forest (Parton, 1996). GEM simulates ecosystem dynamics for a variety of habitats by incorporating ecological processes that determine water levels, plant production, and nutrient cycling associated with natural and human-induced disturbances. Inputs in these models are static scenarios of nutrient loads and climate change.

4.5 Earth Systems Models

Important progress in linking biophysical models has been made through the development of earth systems models (Meyer and Turner II, 1994). In 1990 the US Global Change Research Program (USGCRP) set itself the goal of linking general circulation models, land-surface parametrization models, and ecosystem dynamics models to predict energy and water fluxes between land and the atmosphere (US Global Change Research Program Act, 1990). However, coupling atmospheric, terrestrial, and ecosystem dynamics is not a straightforward task owing to the different spatial and temporal resolutions between land-surface and ecosystem processes. Models can be integrated through a nested approach that allows users to calculate parameter values across models of various resolutions. With the help of advances in computer processing these models are rapidly increasing in sophistication. However, earth systems models are still too limited in representing the complex interactions between the earth's subsystems and human systems. Human actions in these models are represented by static scenarios of highly aggregated land uses and pollution loads into the atmosphere, water, and land.

4.6 Integrated Assessment Models

In response to the need to incorporate a more realistic representation of human and ecological processes in existing models, natural and social scientists have built integrated assessment models (IAMs). IAMs incorporate two tasks. They allow users (1) to integrate various knowledge domains to predict environmental changes associated with the behavior of complex socioeconomic and environmental systems, and (2) to assess the likelihood, importance, and implications of predicted environmental changes to inform policy-making. IAMs have gained interest primarily as a new approach to link biophysical and socioeconomic systems in assessing global environmental change (Dowlatabadi, 1995; Parson and Fisher-Vanden, 1995; Rotmans et al., 1995; Weyant et al, 1996). Since 1994 the USGCRP has made IA modeling its central priority.

The integrated model to assess the greenhouse effect (IMAGE 2) developed by the Dutch National Institute of Public Health and Environmental Protection (RIVM) is designed to simulate the dynamics of the global society – biosphere – climate system (Alcamo, 1994). IMAGE 2 is the first IAM to represent environmental phenomena at a fine spatial scale. It performs many calculations on a global grid ($0.5^\circ \times 0.5^\circ$). The time horizon extends to the year 2100 and the time steps of different submodels vary between one day and five years. The model consists of three fully linked subsets of models: the energy – industry system; the terrestrial environment system; and the atmosphere – ocean system.

The energy – industry models compute the emissions of greenhouse gases in thirteen world regions as a function of energy consumption and industrial production. End-use energy consumption is computed from various economic driving forces. It includes four submodels: energy economy, energy emissions, industrial production, and industrial emissions. The terrestrial environment models simulate the changes in global land cover on a grid scale based on climatic and economic factors.

The roles of land cover and other human factors are then taken into account to compute the flux of CO_2 and other greenhouse gases from the biosphere to the atmosphere. This subsystem includes five submodels: agricultural demand, terrestrial vegetation, land cover, terrestrial carbon, and land-use emissions. The atmosphere – ocean models compute the buildup of greenhouse gases in the atmosphere and the resulting zonal-average temperature and precipitation patterns. Four submodels are included: atmospheric composition, zonal atmospheric climate, oceanic climate, and oceanic biosphere and chemistry.

IMAGE 2 makes a major scientific contribution by representing many important feedback mechanisms and linkages between models in the subsystems, and between subsystems. IMAGE 2 links explicitly and geographically the changes in land cover with the flux of CO_2 and other greenhouse gases between the biosphere and atmosphere, and, conversely, takes into account the effects of climate in the changing productivity of the terrestrial and oceanic biosphere. It also dynamically couples natural and human-induced emissions with chemical and physical processes in the atmosphere and ocean and then feeds climate change back to the biosphere.

Another example of IAM is ICAM-2, an optimization model developed at Carnegie Mellon University to assess the effectiveness of climate change policies (Dowlatabadi and Ball, 1994). Although most IAMs focus on climate changes, more recent efforts in integrated modeling have attempted to address a broader set of policy questions. TARGETS (tool to assess regional and global environmental and health targets for sustainability) is an example of a model designed to inform the policy debate on a broader set of global change issues related to economic development and sustainability. TARGETS is currently being developed by RIVM as part of its research program on global dynamics and sustainable development (Rotmans et al., 1994). This model aims to assess simultaneously several human stresses on various global and regional issues such as climate change, tropospheric ozone, deforestation, and the dispersion of chemicals. None of the current integrated modeling efforts, however, has addressed urban ecosystems.

5 A Conceptual Framework for Modeling the Urban Ecosystem

In this section I present a framework to integrate urban and ecological modeling. This framework is part of a strategy that Alan Borning, Paul Waddell, and I have developed at the University of Washington to build an urban ecological model as part of PRISM, which is a multidisciplinary initiative aimed at developing a dynamic and integrated understanding of the environmental and human systems in the Puget Sound. Our aim is to integrate the various components of the Puget Sound into a metamodel. The urban-ecological model addresses the societal dynamics of environmental change.

5.1 Model Objectives

One major aim of the PRISM human dimension is to describe how human actions generate environmental stresses and to predict the impacts of changes in human actions on the biophysical system. Human decisions in PRISM will be treated explicitly through the development of UEM which will represent the principal actors and behaviors affecting environmental change. This model will predict the environmental stresses associated with urban development and related changes in land-use and human activities under alternative demographic, economic, environmental, and policy scenarios. We start with the assumption that urban development is the outcome of the interactions between the choices of households, businesses, developers, and governments. These actors make decisions that alter the patterns of land-use and human activities. UEM will be designed to model these processes in a dynamic and spatially explicit framework that links these decisions to changes in the biophysical structure of the Puget Sound. This is a first step, we believe, toward coupling human and biophysical processes in the urban ecosystem.

5.2 Conceptual Framework

The urban ecosystem will be represented by a number of human and biophysical variables. Figures 2(a) and 2(b) are schematic diagrams of the major subsystems considered in the model and their interactions. Human systems are represented by four components: actors, resources, markets, and institutions. Major resource-stock variables are population, economic activities (jobs), land, buildings (residential and nonresidential), infrastructure (transportation, energy, water supply, wastewater), and natural resources (water, forests, and ecosystems). The *actors*—households, businesses, developers, and governments—make decisions about *production and consumption*

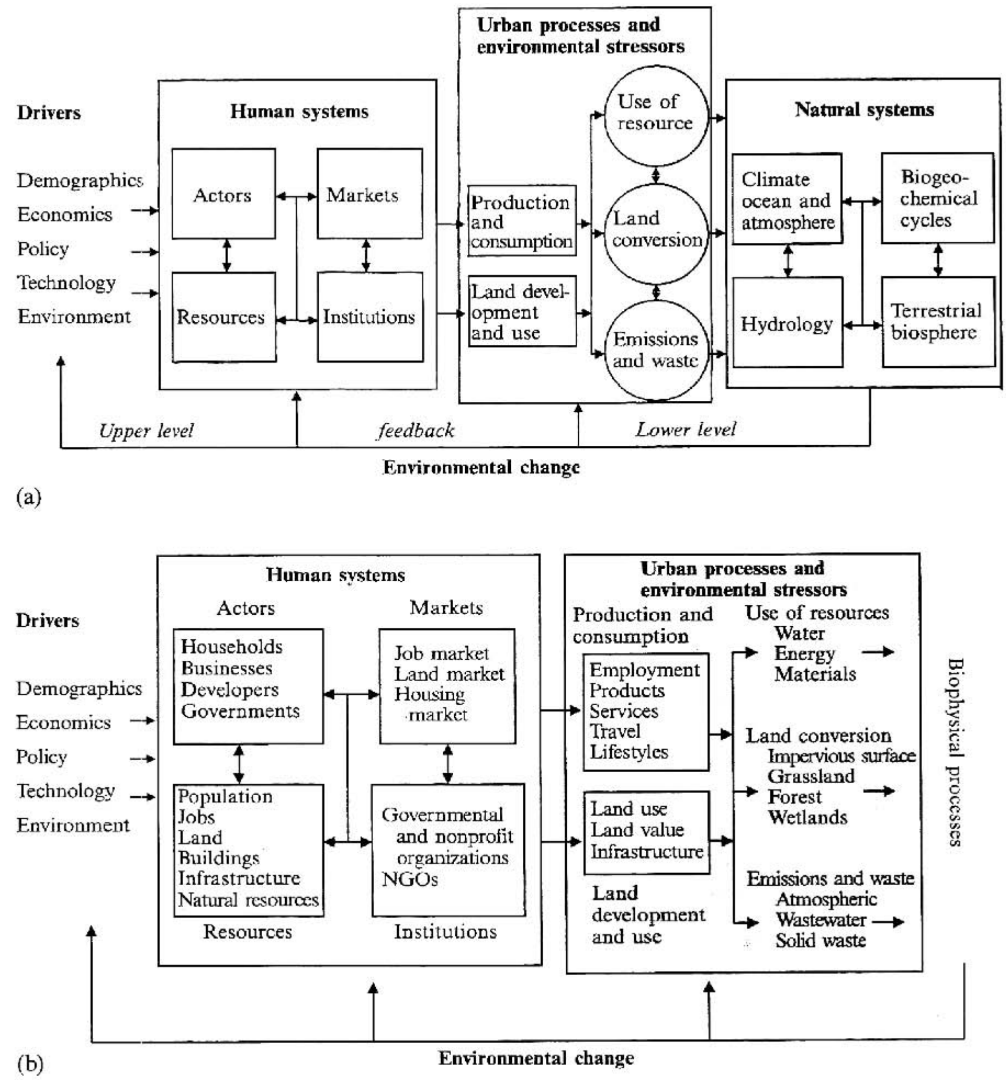

Fig. 2 Urban ecological frameworks

activities and their location, leading to changes in *land use*. These decisions affect, directly and indirectly, the biophysical system through *land conversion*, the *use of resources*, and the generation of *emissions and waste*. Businesses make choices about production, location, and management practices. Households make choices about employment, location, housing type, travel mode, and other lifestyle factors leading to consumption. Developers make decisions about investing in development and redevelopment. Governments make decisions about investing in infrastructures and services and adopting policies and regulations. Decisions are made at the individual and community levels through the economic and social institutions. The actors interact in three submarkets: the job market, the land market, and the housing market. These actors also interact in nonmarket institutions including governmental and other nonprofit and nongovernmental organizations. Decisions are influenced by demographic, socioeconomic, political, and technological factors represented in the model by exogenous scenarios, and are affected by environmental conditions and changes predicted by the biophysical modules. The types of activity and the context in which the activity takes place both determine the level of pressure and the patterns of disturbances.

The output of the urban ecosystem model will serve as the input to several biophysical models such as the climate and atmospheric model, the hydrology model, and the aquatic and terrestrial ecosystem models. The urban-ecological framework is designed to take into account the interactions between the ecological impacts and urban processes at various levels of the hierarchy of processes. These include feedback from the ecological changes on the choices of households and business locations, market prices, availability of land and resources, and regulation. Feedback is also included at the levels of the processes of production and consumption, and land development.

5.3 Model Structure

Using the framework described above we plan to develop an object-oriented model that links urban and ecological processes. We build on UrbanSim, an existing urban simulation model developed by Waddell (1998), to predict three types of human-induced environmental stressors: land conversion, use of resources, and emissions. Figure 3 represents the urban-ecological dynamics that the integrated model will address (Waddell and Alberti, 1998). Our initial focus will be on modeling changes in land use and land cover. Instead of linking the urban and ecological components sequentially, we propose to integrate them at a functional level. Our current strategy is to extend the object properties and methods now implemented in the UrbanSim model. UrbanSim predicts the location behaviors of households, businesses, and developers, and consequent changes in land uses and physical development. These are among the required inputs to predict the changes in land cover and ecological impacts. We propose to add the production and consumption behaviors of households and businesses, and link them through a spatially explicit representation of land and infrastructure to ecological processes.

The UrbanSim data structure is currently being revised from the current aggregate approach to one based on microsimulation and from a zone description of space to a high-resolution grid structure (Waddell and Alberti, 1998). We use a combination of the aggregated economic I – O methodology and a microsimulation approach to model the production and consumption behavior of individual businesses and their location. A microsimulation approach is also being implemented to model households' choices of jobs, location, and lifestyles. A highly disaggregated representation of households (that is, individuals) and businesses (that is, the standard industrial classification) will allow us to represent explicitly detailed production, consumption, and location behaviors of various actors and to link these behaviors to ecological impacts.

The urban ecosystem model simulates three types of human-induced environmental stressors: land conversion, use of resources, and emissions. Changes in land use – cover will be modeled

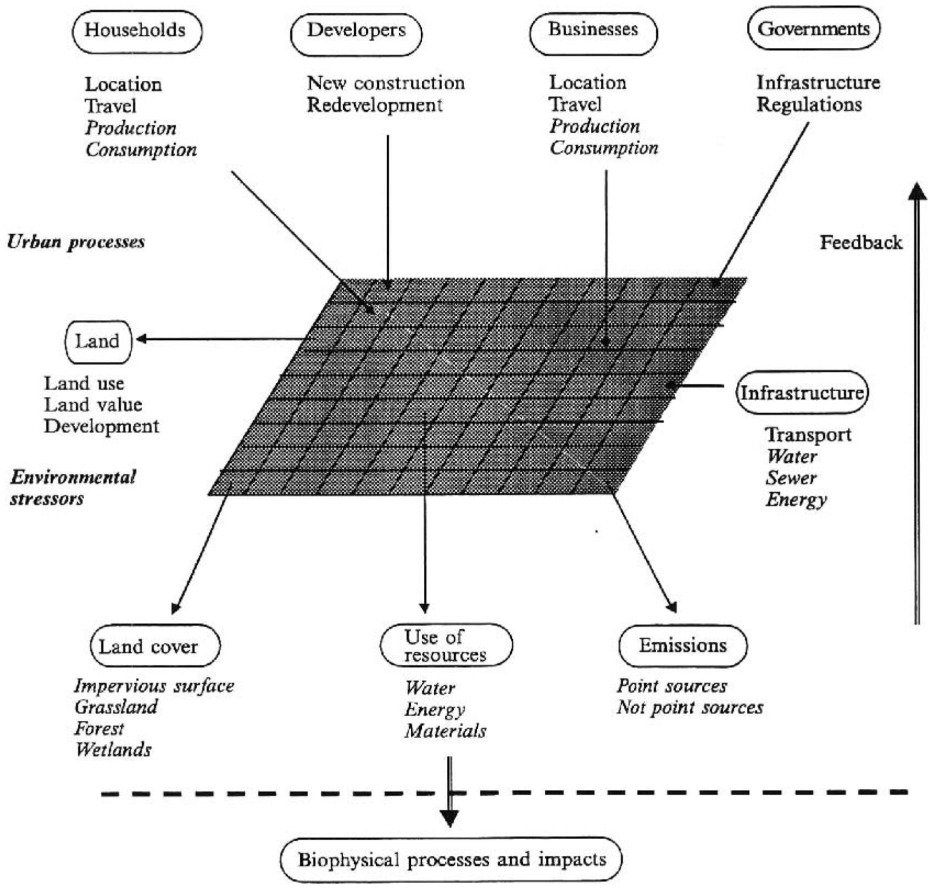

Fig. 3 UEM structure. Note that processes in italics are new components not presently modeled in UrbanSim (source: Waddell and Alberti, 1998)

based on a set of land use – cover determinants, including original use, accessibility, environmental conditions, cost of conversion, and policy constraints. Land conversion will be predicted based on the changes in housing and commercial buildings, household and business characteristics occupying these buildings, and the biophysical characteristics of the land parcels. The resource models will be represented by various modules, each predicting the use of water, energy, and materials, which will be linked to the UEM on the basis of consumption and infrastructure capacity. The emission modules are mass-balance models that will simulate pollution loads into the atmosphere, water, and soil, and relative contributions from the various media.

5.4 Spatial and Temporal Dynamics

Efforts to represent more realistically the spatial and temporal dynamic behavior of urban ecosystems are particularly relevant if urban and ecological processes need to be integrated. An explicit treatment of space will be realized in two steps. First, we will develop and implement a grid of variable resolution to georeference all spatially located objects in the model. This gives us the flexibility to vary spatial resolutions for different processes and to test spatial scale effects on model predictions. The second step will involve the explicit treatment of spatial processes across the area.

The grid provides a foundation for linking urban spatial processes to processes that occur in the natural environment.

We will also improve the treatment of time. Most urban models assume a static cross-sectional approach. The current UrbanSim model uses annual time steps to simulate household choices, real-estate development and redevelopment, and market-clearing and price-adjustment processes within the market (Waddell, 1998). Travel accessibility is updated by a travel model simulation run every ten years, or when significant changes in the transportation system have occurred. Improvements can be achieved by implementing different time steps for different behaviors represented in the model, such as location choices and real-estate development.

5.5 Feedback Mechanisms

The model framework accounts for the interactions between the ecological impacts and urban processes. Ecological changes will feed back on the location choices both of households and businesses, and on the availability of land and resources. For example, the amount and distribution of vegetation-cover canopy in urban areas and its health have both social and ecological functions. Among the most obvious social functions are the attractiveness of the area and its consequent economic value. Important ecological functions include removal of air pollutants, mitigation of microclimate, and consequent savings in building energy, and reduction of storm water run-off. All these factors improve urban environmental quality and provide ecological services to urban residents. These benefits are not currently reflected in land values but eventually will affect long-term urban development. We will define a set of environmental quality indices (for example air quality, water quality, etc) and risk indices (for example floods, landslides, etc) that influence location choices and profitability of development.

6 Implications for Future Research

In this paper I have argued that integration between urban and environmental models needs to be achieved at a functional level to answer critical planning questions. The question is how to represent human processes explicitly in ecological models, and ecological processes explicitly in human models. Although there is no universally valid protocol for designing an integrated UEM, a few considerations emerge from recent experience in the two lines of modeling research. Based on the discussion offered in this paper I indicate a set of attributes that need to be considered to integrate urban-ecological modeling. Above all it is critical that the modeling effort is transparent to users (Smith, 1998). It is important to emphasize that a model is not an end in itself, but rather a tool that will provide its developers and users with a new perspective on the problem being analyzed.

Problem definition. One major concern of Lee's (1973) famous "Requiem for largescale models" was that modeling efforts had failed to reproduce realistically the problems that planning agencies face and thus to provide useful tools for decisionmaking. The first critical step in defining rules for modeling is to be clear about the questions that need to be answered (Wilson, 1974). The current impasse in coupling ecological and human systems models would suggest that even more important is the way we formulate such questions. The integration of urban and ecological models, I have said, cannot be achieved simply by inserting humans into ecosystems or biophysical elements into human systems because the interactions occur at various levels. This implies that the traditional questions, such as how humans affect natural systems and how natural systems affect humans, need to be reformulated to reflect an integrated approach. For example, we will ask how natural and human-generated landscape patterns and energy flows affect natural disturbance regimes and how land-use choices and practices are controlled by human-induced environmental change.

Multiple actors. Urban decisionmakers are a broad and very diversified group of people who make a series of relevant decisions over time. In order to model urban-ecological interactions we must represent explicitly the location, production, and consumption behavior of these multiple actors. This requires a highly disaggregated representation of households and businesses. Disaggregation of economic sectors could be achieved by using a revised version of the I – O model methodology. Microsimulation could help address the difficult trade-offs that households and businesses make between location, production, and consumption preferences (Wegener and Spiekermann, 1996). Morgan and Dowlatabadi (1996) suggest that new methods must be developed to incorporate separate multiattribute utility functions by different social actors in integrated models.

Time. Time needs to be treated explicitly if relevant temporal dynamics of urban and environmental change are to be represented. Time can be represented as a discrete or continuous variable. Though treating time continuously is certainly a daunting task, introducing time steps or using multiple time steps for different processes modeled can provide an important improvement over current models. Today, most operational urban models are based on a cross-sectional, aggregate, equilibrium approach. Improvement over these models could be achieved by representing time explicitly as a discrete variable. A more ambitious continuous approach to the treatment of time will need to be explored.

Space. We need to represent more realistically the spatial behavior of urban processes, both human and ecological. Existing operational urban models are spatially aggregated and, even when they use or produce spatially disaggregated data, they rely on simple spatial geometric processing. Several scholars have developed various functionalities in modeling spatial processes that can be implemented in UEMs. Additional research is required to explore the potential for using CA to model spatially explicit urban dynamic processes efficiently. A more flexible conception of space could help reconcile the spatial resolution needed to model different urban processes. This will require implementing a data structure that will accommodate process resolution ranging from cells to various land units as well as various levels of aggregation across several hierarchical scales.

Scale. The appropriate scales for modeling depend not only on the problem being tackled, but also on considerations of consistency across spatial and temporal aggregations (Lonergan and Prudham, 1994). Modeling urban ecosystem dynamics requires crossing spatial and hierarchical scales. According to hierarchy theory, landscape processes and constraints change across scales (O'Neill et al, 1986). Because landscapes are spatially heterogeneous areas, the outcome of changes in driving forces can be relevant only at certain scales. Yet our current understanding of spatial scale links is still limited. Two scale issues must be addressed in modeling land-use and land-cover change (Turner II et al., 1995). First, each scale has its specific units and variables. Second, the relationships between variables and units change with scale. To tackle these issues, a hierarchical approach needs to be developed.

Feedback. Representing feedback mechanisms in UEMs could improve substantially the ability to predict human behaviors and their ecological impacts. This could be achieved by developing a set of environmental quality indices derived from the biophysical models. We could evaluate not only the relative magnitude of human and natural systems controls in urbanized regions, but also how this will vary in relation to alternative urban structure and management strategies. The explicit representation of feedback loops is also crucial in analyzing the interaction between multiple resource uses and in assessing their overall ecological impact.

Uncertainty. The behavior of ecological and human systems is highly unpredictable owing to their inherent complexity. Modeling these systems is subject to uncertainty. The treatment of uncertainty goes beyond the scope of this paper. It is clear that the use of advanced techniques, such as the Latin hypercube sampling techniques to map the relevance of uncertainty in input data and model parameters needs to be explored.

The integrated knowledge of the processes and mechanisms that govern urban ecosystem dynamics is crucial to advance both ecological and social research and to inform planners and policymakers. It is also crucial to planning education to provide a new framework and more sophisticated tools for the next generation of urban planners and practitioners.

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Scientific, Institutional, and Individual Constraints on Restoring Puget Sound Rivers

Clare M. Ryan and Sara M. Jensen

Abstract The tasks of restoring rivers, watersheds, and other critical habitats are complex and represent some of the most difficult challenges faced by natural resource scientists and managers today. Blame for society's inability to adequately deal with restoration challenges is placed on both scientists and policy makers. Some people argue that the necessary information and levels of certainty fall far short of scientific standards for decision making; others argue that science is not the issue, and indecisiveness merely reflects a lack of political leadership and will. Regardless, the discussion ultimately focuses on the science-policy interface as a root cause of the inability to address such complex management issues. Both science and policy hold unique cultural positions, values, and norms. When the two spheres try to communicate, these differences can interfere with developing, selecting, and implementing management alternatives. This chapter addresses the topic of science and policy communications, offers a conceptual framework to assist in understanding how scientists and policy makers might forge a new dialogue, and discusses additional institutional and individual constraints to restoration efforts. Examples of approaches toward overcoming these constraints provide hope for addressing and ultimately realizing the restoration challenges that lie ahead.

Keywords: scientific method · policy · restoration · urban rivers · science-policy interface

Introduction

> "We must manage through sound science, not politics."
> "We don't have enough data to make a decision."
> "We don't have jurisdiction over this issue."
> "It's my property, and I'll do as I damn well please on it."

Each of these statements illustrates the vexing nature of some of the major constraints to the restoration of Puget Sound rivers. In addition to a number of well-recognized institutional and individual constraints, the vast gap between scientific and policy-making cultures and processes present a formidable challenge. This chapter discusses several of the more common scientific, institutional, and individual constraints and presents examples of efforts that are making strides towards overcoming these barriers. These examples illustrate approaches that emphasize the strategies of

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incentives, innovation, vision, and leadership, each of which is a necessary component to realizing the ultimate goal of restoring Puget Sound rivers. Because much of the blame for lack of progress in current restoration efforts points to the two way failures in science-policy interactions and communication, the chapter begins with an examination of the differences between science and policy cultures and processes and presents a conceptual framework for a more comprehensive understanding of science-policy communications.

Understanding Scientific and Policy-Making Cultures and Processes

Both science and policy are human constructs, developed over millennia and shaped by our learned and innate behavior. In the United States, we gather knowledge and information through science and make public decisions through policy. Science can come from many sources (universities, government agencies, think tanks) and policy can be considered at many levels (elected officials versus bureaucrats; city councils versus Congress). However, science and policy as a whole hold unique cultural positions, values, and norms. When the two try to communicate, these cultural differences can interfere.

The need for effective communication is important in general for the field of restoration ecology, and in particular for Puget Sound river restoration efforts. Many Puget Sound rivers flow through urban centers, which are located in ecologically important and sensitive areas. As human population grows and urbanizes, its citizens must increasingly understand the role a healthy ecosystem plays and weigh the benefits against other urban needs and desires. "Urban restoration ecology" may be a new term to some scientists, many of whom have traditionally studied more pristine and controlled habitats. It may also be new to urban governments, who may not grasp the relevance of ecology to the urban core. For these reasons, there is a unique set of variables to consider in urban river restoration ecology. However, the underlying relationship between science and policy in this arena remains the same. To understand and address the differences between science and policy-making processes, it is useful to step back and understand how science and policy differ from and fail each other and recognize the limits to the current practice of both science and policy in communicating and solving environmental and ecological problems.

Science and Policy Cultures

To begin with, scientists and policy-makers work in distinct cultures. Science holds an honored but insulated place in American society. As an ideal, science operates in a sphere apart from political, religious, or other concerns, allowing scientists to pursue analytical truth.[1] This separation also largely frees scientists from time pressures (science can wait decades or even centuries for the "truth" to emerge). The system is dispersed and relatively non-hierarchical so one view cannot gain a monopoly of resources. The scientific community controls quality through peer review, by encouraging disputes on analytical grounds, and by limiting entry to those that can set aside years for intensive, specialized training. As a result, science is primarily produced and published for other scientists (Taylor 1984).

[1] This is generally true. However, for many citizens, science is not neutral territory. For every romantic view of science as progress and modern magic, there is a Dr. Frankenstein or Dr. Strangelove (Antypas and Meidinger 1996). The questions of whether scientists and society as a whole can be trusted with their discoveries have been raised with such issues as nuclear power, atomic weapons, and more recently, genetic engineering.

In contrast, policy-makers do not undergo a uniform training to enter the occupation. Few come from scientific backgrounds—many are lawyers. The policy community is hierarchical and varied, with participants ranging from elected officials to career bureaucrats, from city council members to state governors. Furthermore, policy-makers face great external pressures from a wide range of actors. They are accountable to voters, special interests, campaign contributors, and superiors. In sum, "policy is not science—it is broader and must accommodate societal concerns within a political arena, not a scientific one" (Hanley 1994, p. 530). This is a culture of compromise and favors, where timing can make or break a policy.

The two cultures therefore have inherent communication problems. For science, the very culture that produces impressive data hampers its communication with other fields. The insulation that protects scientists can, in some cases, blind them to trends and events in the outside world. The system offers little incentive to publish findings for public consumption. Scientists are more comfortable with both uncertainty and long research time frames and are therefore often unable to translate results without sounding vague to policy-makers (who often equate uncertainty and vagueness). The fragmentation of efforts that protects against monopoly of resources or truth often leads to a narrow research focus, and the related narrow scope of most scientific journals discourages multi-disciplinary research efforts.

The policy culture also creates a number of problems for understanding and incorporating science. The general lack of basic scientific knowledge, lack of awareness of inherent scientific uncertainties, and the relatively short decision-making time frames make it difficult for many policy makers to include science as an input. Even when science enters the policy process, the extreme external pressures on policy, common instances of short-term over long-term rationality, and fragmentation of responsibilities, authorities, interests, and values all dilute and hamper its use (Yaffee 1997). In short, there are many reasons that science and policy have difficulty communicating. The gap between cultures is wide, and few on either side have the inclination, time, and skills to understand the other.

Science and Policy Communication

Despite the formidable barriers, scientists and policy makers do communicate. There are four primary models representing this dialogue (Fig. 1).

The One-Way Information Flow Model

In the first model (Fig. 1A), science and policy operate in separate spheres and may never directly interact. For example, a scientist researching the effects of global warming on stream temperatures might submit the results to a peer-reviewed journal and consider the project completed. A policy-maker may never see those results and is unlikely to be able to incorporate them into a policy decision affecting Puget Sound rivers. In the same way, a policy maker may look to the scientific realm for information or data on an issue without directly interacting with the scientists involved (e.g., by conducting a literature search, which would not be a direct interaction with a scientist but only an interaction with published studies).

The Trans-Science Model

This model (Fig. 1B), developed by Alvin Weinberg in 1972, considers that there are some policy issues that depend on scientific answers, but which science cannot answer. The trans-science sphere in this model represents a hybrid realm of policy-relevant scientific questions, whose resolution requires non-scientific methods (Jager 1998). An example might be the question of what

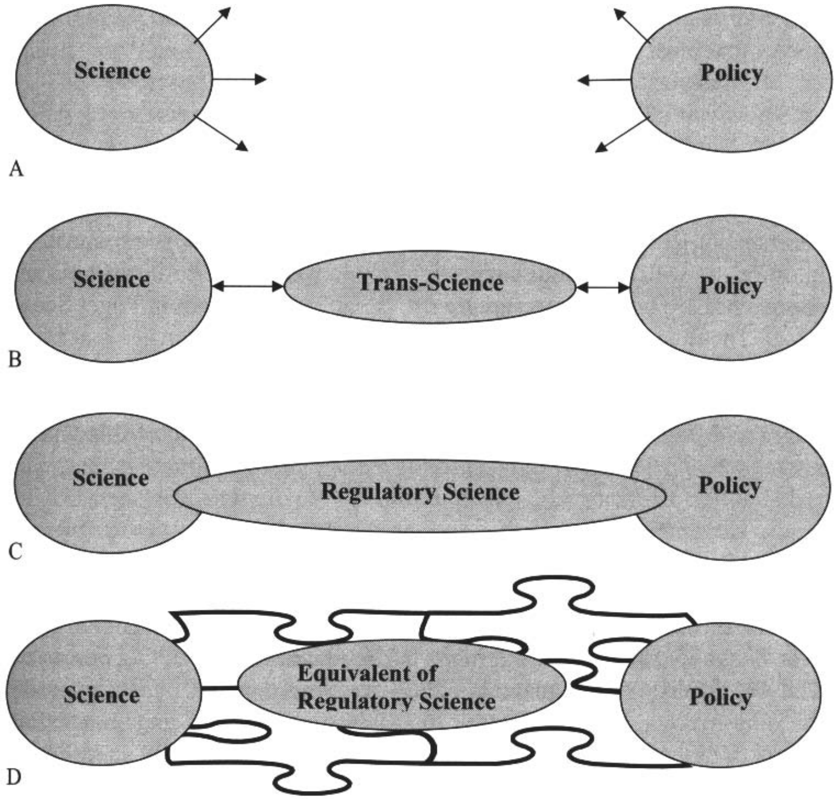

Fig. 1 Models of Science-Policy Communications: (A) one-way information flow model, (B) trans-science model, (C) Regulatory Science model, and (D) Mutual construction model

the potential impacts are of removing one of the dams on the Columbia River to provide access to habitat for endangered salmon. Research in this area is likely to involve models and other estimates of impacts, which yield policy recommendations, but cannot provide a certain answer or prediction. This question would have to be resolved in a political arena, taking into account risks, costs, and other value trade-offs.

The Regulatory-Science Model

The regulatory science sphere in this model (Fig. 1C) represents research in which the content and context are driven by specific policy or legally driven questions (Jager 1998). An example is research that supports the quantification and determination of Total Maximum Daily Loads (TMDLs) in water bodies that must comply with the Clean Water Act.

The Mutual Construction Model

In this model (Fig. 1D), science and policy actively influence each other in constructing research questions and options. For example, there are many questions as well as various suggestions about which monitoring criteria and approaches are best able to capture the variables of concern in Puget Sound rivers. These suggestions are mutually influenced by the agencies that frame the question and fund such research, and by scientists who offer and respond with monitoring approaches that they hope will be incorporated into policy. Ralph and Poole (Chapter 9) provide an additional example of this type of interaction.

Towards a Conceptual Framework of the Science-Policy Dialogue

Science thus can and does influence environmental policy decisions. However, it is only *one* input among many that feed into policy-making processes. The weight that science plays in policy decisions depends on the issue, the decision-makers, and the stage in the policy process (i.e., agenda setting versus final legislation), among other things. In turn, policy has an influence on science. For example, federal, state, or local funding decisions may have a direct influence on the nature and content of scientific research.

What are some of the possibilities for future dialogue between science and policy? The post-normal science approach is often presented as an emerging field, one that has a chance to democratize science by extending the “peer” community to everyone with a stake in the issue. According to Hellstrom (1996), post-normal science differs from normal science by confronting underlying views, extending problems, including uncertainties, intersecting scientific and policy considerations, incorporating external pressure, and stressing trans-disciplinary research. The approach applies when difficult or complex policy choices must be made on the basis of uncertain scientific inputs (Hellstrom 1996), or when system uncertainties and decision stakes are high (Funtowicz and Ravetz 1993). One element of the approach is that the task of defining what is good and acceptable research moves from the scientific community to the policy and economic communities of interests. Although many argue that there is no difference between post-normal science and other applied scientific efforts, this is a potentially important approach for urban and agricultural ecosystems where the key variables (uncertainties and stakes) are high.

The conceptual framework presented in Fig. 2 places scientific and policy processes along two gradients: external pressure and a science-policy continuum. This framework builds on the current literature and adds several additional elements. It purposely does not include the “trans-science” model discussed earlier because, short of a scientific dictatorship, no environmental problem can be solved solely with science. Therefore, *all* policy decisions are trans-scientific.

The Academic Science Sphere

Academic science is the realm of empirical inquiry, theory, and the pursuit of knowledge. Members of this sphere publish results under established systems of peer reviewers and referees to control quality. The scientific cultural characteristics discussed previously are present in this sphere. Within this general science culture, the conceptual framework recognizes three types of scientific endeavor. There is a core science realm made up of knowledge-based researchers pursuing basic science. This group receives the least external pressure on its work. Members of this scientific arena hold authority and status with policy makers. However, a diversity of expert opinion and high levels of uncertainty can diminish their influence on policy.

There is also an issue-based research realm. This group might include scientists researching the safety of new products at a chemical company, EPA toxicologists studying pesticides, and scientist-activists. These scientists subscribe to the importance of peer-review and other attributes of core science, but they differ in that they are working towards a pre-determined agenda (perhaps stated, perhaps not). These scientists may hold authority and legitimacy in some policy arenas but are regarded suspiciously in others. This type of science may be more relevant to policy makers and as a result is exposed to slightly greater external pressure.

The third type of science in this sphere is interdisciplinary research. A large consortium of scientists addresses a particular research question (e.g., social scientists, economists, and ecologists),

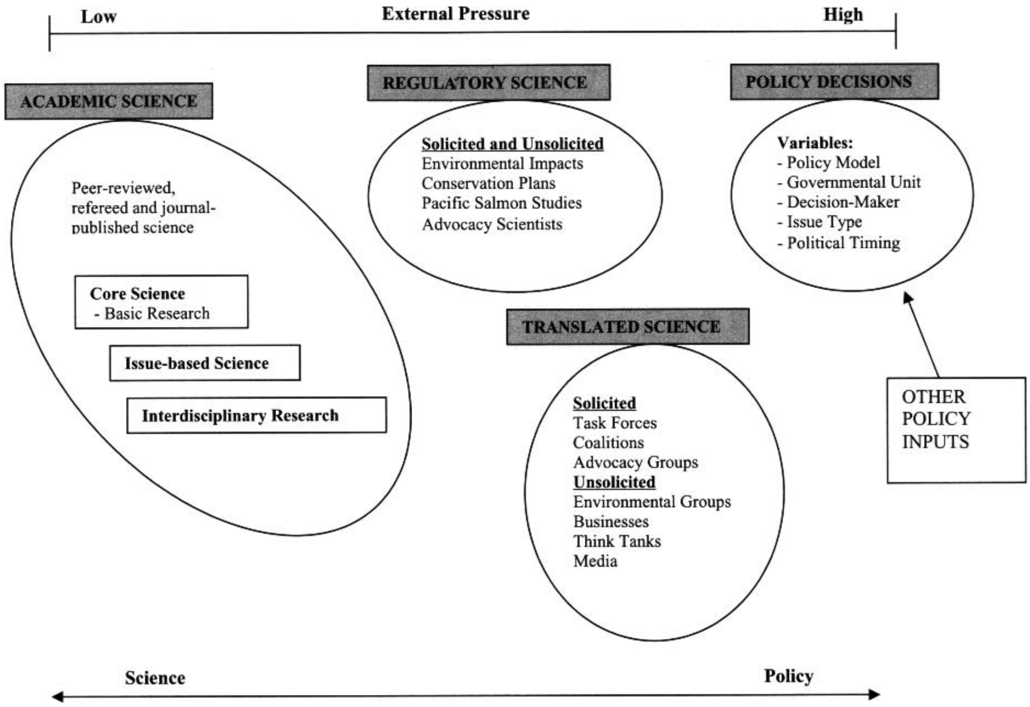

Fig. 2 Science and Policy Conceptual Framework. There are two gradients included in the framework: external pressure and a science-policy continuum. The four process ovals are situated in the framework along these two gradients

which has a broader potential audience. Of the three types, this research has the largest scope and most relevance to policy and therefore experiences the most external pressure.

Scientists can and do transfer between these three types of science, and they may fall in two or all three categories at once depending on current research projects. Importantly, information flows not only from these science realms into the policy realm but also from the policy realm into the science realm. Policy can influence all three research types, although basic research tends to be more insulated from outside policy influences. Influences can stem from grants, fellowships, tax codes, legislation on human and animal subjects, and definitions of long-term needs by agencies.

The Regulatory Science Sphere

The concept of a regulatory science sphere is adapted from current policy modeling literature. This sphere is one step away from science and one step closer to policy along the continuum; thus it is subject to greater external pressures than the academic science sphere. Original research prevails, although results may not be peer-reviewed, refereed, or published in scientific journals. Research or information gathering is generally conducted in direct response to a policy need or law, such as environmental impact statements (EIS) and habitat conservation plans. Research in this sphere can be solicited (usually the case for an EIS), or unsolicited (for example, a Sierra Club study of urban sprawl). Data are often directly relevant to policy-makers, but there is a potential for bias and lack of scientific rigor. Information flows in a two-way relationship between this sphere and the other spheres in the model.

The Translated Science Sphere

No original research is conducted in this sphere; rather existing science is translated to policy-makers. As shown by its location on the continuum, science and policy directly influence one another here, as various groups interpret scientific data to compose policy recommendations. This sphere includes task forces and scientific coalitions charged with making a coherent policy recommendation (solicited science). It also includes business and environmental groups that selectively use scientific information to support an agenda (unsolicited science). Members of this sphere essentially interpret science for policy makers. They thus have some of the authority of science as well as the ability to distill a message for policy. This power is diminished, however, by the strong likelihood or suspicion of bias.

The Policy Decisions Sphere

Policy decisions are made in complex ways in a variety of arenas and are subject to the greatest external pressure. The conceptual framework focuses on some of the scientific inputs to policy decisions rather than the mechanisms of those decisions. However, the framework does incorporate several of the important variables that affect these decisions. These variables are listed in the policy sphere and include assumptions about how policy is formulated and implemented (the policy model or theoretical frameworks), the governmental unit, the background and sentiments of the decision maker, the issue type, and political timing. These last two variables are discussed in more detail below.

Issue Type: The particular issue under consideration determines the weight that science will have in the decision and the scientific arena to which the policy makers will turn. For example, Congress could conceivably trust EPA scientists on an issue of toxicology but turn to the core science arena to learn about acid rain. Similarly, scientific considerations might predominate in setting drinking water standards, while politics and business interests drive superfund legislation. Thus the flows in the policy model change with every issue under consideration.

Political Timing: The point along a decision-making process can greatly affect the influence of science on policy and vice-versa. For example, science may strongly influence problem identification and agenda setting, but it may get diluted at later stages of the policy process. Adams and Hairston (1996, p. 28) noted that "By the time a policymaker says, 'Let's call in the experts,' an issue typically has become widely visible and controversial. At that point, resolution of the issue may rely as much or more on politics as on science." On the flip side, before an issue is on the political agenda, scientific processes may be able to proceed relatively free from direct policy influence.

The framework illustrated in Fig. 2 attempts to capture information and influence flows rather than physical processes, patterns, or changes. Policy decisions (or non-decisions) are drivers in urban systems. Policy can directly affect land-use patterns, hydrology, climate and atmosphere, and the viability of wildlife. Land-use patterns and hydrologic systems act as secondary drivers for other physical processes. These physical processes in turn provide subject matter for scientists to study, which may ultimately lead to changes in policy. Importantly, policy and science are not simply drivers but are themselves processes, driven by human behavior.

The framework also illustrates that the science sphere is difficult for policy-makers to access, and its answers are usually perceived as vague. What fills the gap between science and policy (translated and regulatory science) can help, but there will likely be concerns about bias, hidden agendas, and scientific rigor. The post-normal science approach discussed previously is promising, but without a set of rules and norms is open to the same problems as the translated and regulatory science spheres. Policy makers are left without clear avenues for solid scientific information.

To have the greatest potential to affect policy, ecologists and other scientists should consider current policy questions when designing their research. Working in interdisciplinary groups may allow for a more comprehensive understanding of the problems and provide unique opportunities for integration of results. If these first two steps are followed, adding humans to the framework should not be a difficult stretch because many ecologists and other scientists are already considering economic and social values as part of their work. Szaro et al. (1998) recommend that future natural resource management include human motivation and responses as part of the system to be studied and managed.

Finally, there is often a lack of avenues for scientists to participate in policy-making processes, with a clear bias from both sides against scientist-activists. Two recent articles discuss this phenomenon within ecology and find similar attitudes, even though they also find a large segment of scientists who feel restricted by the boundaries of the academic science sphere (Brown 2000; Kaiser 2000). Scientists have the same concerns about translated and regulatory science as policy makers, with the additional problems that participating in these spheres can be time consuming, contribute little to or even damage a scientific or academic career, and be potentially frustrating for holders of a minority view.

The conceptual framework raises a number of questions regarding how we as a society ask scientific questions and what might be done to better inform policy-making processes. In the area of restoration, for example, do we have a clear idea of what we are trying to restore? Can we do a better job of framing or anticipating scientific questions in a policy-relevant way?

Reframing research questions and approaches may get us further towards bridging the gap between science and policy. Carefully crafted, applied research can be relevant to policy and still remain in the science sphere. Many issues fall in the realm *between* science and policy, however. For these issues, it is important to create rules, norms, and quality controls for the region between science and policy. Finally, policy makers can help by better educating themselves in the realm of science. If policy makers can grasp what science can and cannot answer, and if government regulatory agencies make hiring scientists a priority, we will also progress along the path towards bridging this gap. Technically trained policy makers can serve as their own scientific translators and thus avoid the potential bias in the inter-science spheres.

While the gaps in science-policy communication seem quite wide, there is hope that they can be bridged. Awareness of these gaps between science and policy cultures and processes provides an important context for examining and understanding the numerous on-the-ground restoration efforts currently underway in the Puget Sound region. Let us now turn to a brief discussion of some of the institutional and individual constraints to restoration of Puget Sound rivers.

Institutional Constraints on Restoration

A number of institutional behaviors constrain our ability to conduct restoration efforts (Yaffee 1997). The following institutional behavioral biases lead to policy impasses and poor choices. Restoration activity in the Puget Sound region exhibits many of the constraints that Yaffee identifies: short-term rationality, competitive behavior, and fragmentation.

Short-Term Rationality

Institutions have a tendency to make decisions that are rational and effective in the short term, yet counterproductive and ineffective over the long term. These usually end up being poor choices and reflect the “tragedy of the commons” idea so prevalent in natural resource management today.

Competitive, Not Cooperative, Behavior

Institutions exhibit a tendency to promote competitive behavior at the expense of cooperative actions, yet often cooperation is needed to identify and implement good solutions. Our institutions operate within a "win-lose" model of decision making. This is changing slowly, but we need to think about incentives for cooperation and integration. It is clear that institutions will soon be forced to begin thinking about performing the "unnatural" acts of cooperating and sharing information and resources to a much greater extent than they have in the past.

Fragmentation of Interests and Values

Interest group politics have become dominant, and restoration involves many groups who do not agree on goals, values, or data. There is a proclivity to split the different elements of society, avoiding the integration of interests and values necessary to craft and sustain effective courses of action on contentious issues.

Fragmentation of Responsibility and Authority

In the Puget Sound region, there are an inordinately large number of institutions that have responsibility and authority for restoration, but very few (perhaps no) incentives to cooperate. Again, we see the tendency to separate the institutions responsible for resource management, there by diminishing accountability and resulting in management strategies that are piecemeal solutions.

Fragmented Information and Knowledge

There is a tendency to fragment what is known about a situation and its context, so that decision makers are hampered and make poor choices because they are operating with inadequate information. This problem was addressed to some extent in our earlier discussion of science-policy dialogues.

To Yaffee's (1997) list, two additional constraints should be added: *Leadership and risk-taking*. Perhaps the lack of leadership and reluctance to take risks are merely logical results of the preceding behavioral biases, but many of the more successful efforts boil down to the simple (yet incredibly difficult-to-implement) ideas of leadership and risk-taking. Certainly in the Puget Sound area we have seen a leadership vacuum reflected in many key restoration questions, such as: Who is in charge? Where are we going? What is the "vision" of restoration that we want and how will we get there (Currens et al. 2000)?

These institutional biases operate at all levels of organizations, and they are extremely difficult to confront and solve. Institutions have a tendency to break things apart and hold them apart, when integration is what is needed. Several of the examples discussed later in the chapter illustrate approaches to changing or intervening in these tendencies.

Individual Constraints on Restoration

Because individuals are an integral component of our institutions, many of the individual constraints faced in restoration efforts are directly related to, and in fact are likely causes of, the institutional constraints. Although there are numerous other societal issues that contribute to our inability to conduct restoration efforts, this section briefly discusses only three individual constraints: private property, litigation, and individual behavior.

Private Property Ethic

The society of the United States exhibits and encourages a strong individual private property ethic, which often does not coincide with the goals and requirements of restoration efforts. Although a system of regulation and enforcement is in place that limits some behaviors and actions on private property (e.g., dredging and filling requirements, building standards), many of the regulations are not directly related to restoration goals. In addition, legal challenges arise in association with attempts to compel individuals to conduct activities on their property that will aid in restoration when such actions may preclude their use of the property for other uses (the "takings" issue). If in fact private landowners are conducting activities on their property that provide or contribute to a larger "public good" (such as restoration), management institutions and policy makers would be wise to explore compensation and incentive schemes such that individual landowners are not bearing the sole burden for contributing to this larger societal benefit. In short, restoration is expensive and society should be prepared to pay for it.

Litigation

The United States is a highly litigious society. In spite of energetic, costly, and lengthy courtroom efforts, litigation rarely resolves the real issues underlying the conflict. We must seek and use alternatives to litigation that will meet the many interests and goals in our restoration efforts. Again, leadership and creative risk-taking in developing solutions has the potential to get us further towards restoration than contentious legal battles.

Individual Behavior

Government institutions cannot do all that is necessary for restoration, and all levels of society will have to participate in some way if we are to achieve this goal. A difficult challenge is the question of how to educate and motivate individual citizens to change their behavior in ways that will benefit restoration efforts. For example, more comprehensive information is needed regarding what land and resource-use practices are appropriate to aid restoration, and how to encourage individual landowners to conduct certain activities on their land or as part of their daily lives. Much work is needed in the area of individual behavior change and on the incentives that can be used to influence individual behaviors.

Realizing Restoration: Incentives, Innovation, Vision, and Leadership

Although it is important to be aware of the constraints, it is also useful to provide examples where many of these barriers are beginning to be addressed (Chapter 7). Below are highlighted just a few examples of the literally hundreds of restoration and enhancement projects going on today in the Puget Sound area. Although it is difficult to identify clear "success stories," the following examples embrace the use of key strategies for realizing restoration—incentives, innovation, vision, and leadership. Several of the examples illustrate promising approaches to overcoming some of the institutional and individual barriers discussed earlier in this chapter.

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Incentives in Bear Creek

Bear Creek, in the Lake Washington watershed, is one of the most productive streams for its size in Puget Sound, supporting chinook (*Oncorhynchus tshawytscha*), coho (*O. kisutch*), sockeye (*O. nerka*), and other fish species. Bear Creek continues to support salmon due to 15 years of effort by citizen activists, along with staff from King and Snohomish Counties, the city of Redmond, and involved landowners. All parties have cooperated to protect near-pristine stream banks, extensive wetlands, and major forested upland areas. Much of the stream system is protected because city and county staff worked directly with landowners, offering a variety of tax incentives, money for easements, and regulations. In addition, the Salmon Recovery Funding Board (SRFB) provided $250,000 to match more than $4 million in public and private funds to acquire land and timber rights for one of the largest wetlands in the headwaters of Bear Creek (Ruckleshaus 2000).

Innovation in Chimacum Creek

Chimacum Creek is located on the Olympic Peninsula south of Port Townsend, Washington. Hood Canal summer chum salmon (*O. keta*) have been listed as an endangered species and have been missing from Chimacum Creek for several years. Through a creative and innovative approach of community involvement, landowners in the Chimacum Valley and other Jefferson County citizens have been working to restore the health of the creek and reintroduce chum salmon. The local community has an annual salmon celebration and, some years back, started a treasure hunt to inform people about the area's natural and cultural history (Ruckleshaus 2000). Farmers, artists, tribal members, volunteers, and government officials have been fencing areas to protect the stream, creating small wetlands and other features once part of this natural system. In fall 2000, chum salmon were seen in Chimacum Creek for the first time in more than 10 years (Ruckleshaus 2000).

A Vision in Oregon—River Renaissance

The Willamette River, which runs through downtown Portland, Oregon, was recently designated as an American Heritage River. However, the health of the Willamette watershed has been severely compromised by urban growth and development. The listing of steelhead trout (*O. mykiss*) and chinook salmon, a likely listing of the harbor as a Superfund site, and numerous combined sewer overflows are only a few of its challenges. The city of Portland's Bureau of Planning has as its goal the creation of a community-wide vision and effort to revitalize the Willamette River, its banks, and its tributaries throughout Portland. At the core is a vision that attempts to integrate the natural, economic, urban, and recreational roles that make the Willamette important to the region. The plan attempts to encompass new initiatives and efforts already underway as a way of aligning city work plans and generating opportunities to coordinate efforts and benefits.

Leadership in Puget Sound—The "Shared Strategy"

Puget Sound is currently struggling with institutional and political gridlock and paralysis, which will likely prove to be untenable for the long term. Recently, leaders have emerged in the Puget Sound region in an attempt to overcome the political and jurisdictional barriers associated with restoring Puget Sound rivers, and ultimately, Puget Sound salmon. Specifically, Daniel J. Evans (former Washington State Governor and United States Senator) and William D. Ruckleshaus (currently Chair of the Washington Salmon Recovery Funding Board) gathered together many of the key players involved in restoration to discuss, develop, and initiate a solution to the restoration challenges faced by the region. Attendees included Governor Gary Locke and other state and local elected officials, representatives of federal, state, tribal, and local governments, as well as businesses, environmental and conservation organizations, and watershed groups. Convened in early 2001, the group has the goals of developing a "Shared Strategy" and implementing a proposed regional recovery plan for salmon in Puget Sound. Implementation of the strategy is yet to be seen, but the results from this meeting, which all key parties supported, attempted to create a new institution to assist in reaching the goal of restoration. State, federal, and tribal leaders with significant statutory authority for salmon recovery made a commitment to implementing the strategy, and there appears to be significant political and agency support. However, many people are skeptical of this approach, which may result in the creation of yet another institution to add to the list of numerous existing institutions responsible for the management and rehabilitation of Puget Sound rivers.

Realizing Restoration—Can We Get There From Here?

While rehabilitation and improvement of Puget Sound rivers is certainly possible, these efforts will require overcoming significant scientific, institutional, and individual constraints. In addition, to accomplish our restoration goals, we must have a clear vision of what those goals are along with an understanding and acceptance of the costs and trade-offs required to achieve them.

Scientists, institutions, and other individuals will play critical roles in the effort to achieve the goal of restoration. A major, but not insurmountable, barrier is the difficulty with science and policy communication. The current situation of political and institutional gridlock is not sustainable, and political leadership, innovation, and risk-taking are needed to break the impasse. Together with scientists and other citizens, we must think about going beyond the common view of restoration as restoring only the natural ecosystem and think about how we can "restore" the political and social systems that are necessary to support a comprehensive restoration effort. Finally, political and governmental institutions are not solely responsible for achieving restoration goals. Individual members of society, through their behavior and policy choices, will also be key players in restoration efforts.

In addition, we must accept the premise that restoration is expensive, and we should prepare to pay for it. Re-framing our current thinking about ecosystems, marketing, and education will help us make this leap. If we can begin to think of ecosystems as infrastructure and invest in ecosystems the same way that we do highways, for example, we may be able to make significant progress towards our restoration goals.

Finally, for many ecological restoration questions and policy choices, there are no clear answers. What restoration goals are appropriate and feasible? How much should we spend and who should pay? Should we prioritize drinking water, salmon, or energy? Because various segments of society often value competing priorities differently, in the end decision-makers are faced with extremely difficult policy choices, not scientific decisions, about where to put our limited resources (e.g., transportation, homelessness, restoration, education). These choices require an understanding of what values are traded off in order to achieve a particular policy goal, such as restoration. Science can only take us so far towards realizing the goal of restoration. Our political and social systems, fueled by imagination, ingenuity, and a willingness to take risks must carry us the rest of the way.

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Toward Ecosystem Management: Shifts in the Core and the Context of Urban Forest Ecology

Rowan A. Rowntree

Abstract The core of urban forest ecology is the body of scientific knowledge found in the literature. The formation of this core began in the 1950s and 1960s and took shape in the 1970s and 1980s with formal studies of structure and function. During the following ten years, the idea of the urban forest ecosystem was introduced, and is now the basis for further development of the scientific core. The context for this core is provided by statements of public policy and perceptions of land management needs. An important shift is occurring in context as land management organizations, ranging from urban-based alliances to state and federal agencies, embrace the ecosystem concept as an approach to understanding and governing complex mixtures of biophysical and human phenomena using a hierarchy of time and space scales. This rapid shift in context places a burden on the scientific core to articulate and test models of urban forest ecosystems. To accomplish this, an approach to research is needed that will help us understand how urban, periurban, and exurban lands interact functionally with other components of the larger landscape. Part of this approach requires scientists and managers to develop a common vocabulary and set of realistic expectations to confront problems of systems complexity and uncertainty.

Keywords: urban forestry · ecosystem management · policy · land management · deforestation · afforestation · complexity · uncertainty

Both the core and the context of urban forest ecology are changing. The core is evolving slowly from a body of studies focused on structure and function to one including the ecosystem concept. The public and political context is shifting more rapidly to embrace the ecosystem approach to land management in both urban and wildland regions. The ecosystem approach is explicitly "science based." The question is whether resource science in general and urban forest ecology in particular are equipped to provide the support necessary to implement ecosystem management. The fact that ecosystem science and the antecedents of ecosystem management have historically dealt with nonurban areas exacerbates the problem for urban forest ecology.

A preliminary definition of terms will serve as a point of departure. The *urban forest* is all the vegetation in an urbanized area. The *periurban* area immediately surrounds the urban forest, and the *exurban* area is the larger hinterland into which people are migrating from urban and periurban zones and from which resources for the city are taken. Rather than defining three separate areas, it is more realistic in many regions to think in terms of a gradient of "urbanness" (e.g., population or building/road density) from a high in the city to a low in the exurban hinterland. The *core* of urban forest ecology is the body of scientific literature that develops concepts and methods, explicates

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principles, and interprets empirical results. It focuses on relations among vegetation, soils, wildlife, water, energy, the atmosphere—and, of course, humans.

The *urban forest ecosystem* is a concept that enlarges the scope of the urban forest to include humans. People are not part of the urban forest but they are part of the urban forest ecosystem. The ecosystem concept also enlarges our temporal and spatial scales of concern. As such, it is an accounting system that requires us to examine how our actions produce costs and benefits not only within our ecosystem but in ecosystems linked to ours across large units of space and time. The urban forest ecosystem is a concept that requires us to understand ecosystem processes that produce changes over time. These changes occur gradually (the aging of trees) and episodically (a large urban fire in the Oakland-Berkeley hills of California).

The *context* of urban forest ecology comprises the public policy and implementation strategies that inform scientists about the needs of land managers to have useful concepts, methods, and management approaches. This context also provides scientists with data about how forest ecosystems are operating and the constraints and opportunities of management. Thus core and context must interact functionally: science informs policy, and the implementation of policy informs science. The scientific core does not operate in isolation of public perceptions about such concerns as the value of science, what is good in nature, and the complexity and uncertainty in this world.

In the sections following, I first restate the problem of urbanization and forests, then describe how urban forest ecology has addressed that problem. Questions and conceptual dilemmas emerge from the gap between the problem and where the core of science stands today. The gap can be closed, and the science can become more capable of answering fundamental questions, if a more systems-oriented approach is taken in the research. This approach can better nurture the development of both the scientific core and the public policy context of urban forest ecology. The theme of this book is integration, and this chapter examines the potential for integrating core and context so that the combined human effort invested in science and management achieves a higher degree of efficiency.

Phenomenon and Problem: Deforestation and Afforestation During the Course of Urbanization

As people migrate, settle, and establish economic and social organizations, these activities change the soil and vegetation with consequences that we do not fully understand, even though we have been asking questions about the human impact for more than a century. These consequences distribute themselves at various scales of time and space, among different types of people as benefits and costs expressed in both monetary and nonmonetary terms. We are changing landscapes daily without knowing the magnitude and distribution of these costs and benefits. It is not just *deforestation* for roads, cities, and industry that is of interest to us, but also the processes of *afforestation* of urban lands, and of derelict and discarded lands. By a combination of both removing and planting trees, our society is changing the functioning of whole ecosystems and the course of global ecological evolution.

During the last fifty years, scientists have addressed the dynamics of vegetation and soil changes in natural ecosystems. But now it is difficult to find an ecosystem that has not been influenced by humans. This knowledge about natural ecosystems is slowly being augmented with research on how human land uses related to urbanization change vegetation and soil attributes that in turn modify ecosystem structure, function, and future trajectory. The problem can be stated in a more prescriptive way: How do we mitigate the costly environmental effects of human settlement, land use, and urban development? Much of the demand for research information comes from individuals and organizations already acting to ameliorate these negative effects. Ecological restoration, urban forestry, and greenspace management are but a few of the international efforts begging not just for more technical knowledge but for a comprehensive and integrated approach that fully accounts for the spatial and temporal distribution of benefits and costs of different actions. The rationale for developing an ecological science and management policy for urban forestry is simple. Nature and society operate as systems. Any action will affect the operation of the whole system; and various system states, or modes of operation, generate various spatial-temporal distributions of benefits and costs.

During the last two decades, urban forestry took its first steps in employing ecological concepts and methods, thus putting the discipline on sound, but preliminary, scientific footing. Scientists, teachers, and managers in urban forestry are being asked to take the next step and formulate their future in terms of ecosystem management by incorporating expanded time and space horizons, accounting for externalities, and monitoring fluxes of energy, water, and matter within and between adjacent and proximate ecosystems. Is urban forestry ready for ecosystem management? To answer this we need to examine how the core and context are informing each other. From this examination, we can determine if the scientific core is evolving in conjunction and synchrony with its management context.

The Core of Urban Forest Ecology

As a new field of study and practice emerges, concepts and methods are either borrowed or invented. Those who have participated in the construction of the core of knowledge in urban forest ecology wisely have avoided making up new concepts that would create a narrow and somewhat exclusive vocabulary. The task has been to import and test concepts from forest ecology to see if they are useful in establishing a scientific foundation for urban forest ecology. The fundamental concepts of structure and function are traditionally used in examining any system, from cell to landscape, and have served forest ecology well. Thus there was little difficulty in importing and successfully using them in urban forest ecology. More difficult is the search for means to describe how the parts make up the whole.

Structure and Function

Structure is the array of static attributes of the urban forest, the concept that asks the question "What is where?" *Function* is the dynamic operation of the forest: how the vegetation interacts with other components of the urban forest ecosystem, including humans, and how internal and external forces change urban forest structure over time. Examples of structure are the spatial distribution of species, biomass, size, age, and condition classes—the attributes one would record in an inventory of all the trees in a city. A slightly expanded definition of structure—one that would move our thinking toward the concept of the urban forest ecosystem—would include other attributes determining the condition of trees, plants, and soils. For example, just as the size and spatial distribution of rock outcrops are part of what determines the structure and function of mountain forests, the spatial patterns of buildings and other artificial surfaces define the geography of growing space for urban trees and plants and determine many of the conditions under which they will live.

Examples of function are the physiological operations sustaining life in the vegetation and soil and how these operations affect other components of the urban forest ecosystem, including people. Trees transpire and create a moist microenvironment for insects while cooling the air for humans. Roots break sidewalks in their search for nutrients, water, and gas exchange. Shade allows people to turn down their air-conditioners. When a tree affects something else in the system, or vice versa, that is ecological function. The functions of disease and aging change forest composition over time. The functional interactions with weather and humans provide for episodic changes like fires and ice storms. Inherent in the ecological approach, and the foundation for ecosystem management, is the capacity to understand structure and function at different spatial and temporal scales and to incorporate both gradual and episodic (often termed "catastrophic") changes.

At the smallest scale, the ecological functioning of the urban forest begins with the interactions between individual trees and with other components of the ecosystem. When thinking about structure and function at different scales, one asks how activities at one scale influence those at another. How, for example, does the operation of a single tree fit into the functioning of the forest ecosystem? Answering this question incorporates the study of arboriculture into urban forest ecology. Therefore, sound ecological thinking integrates the operations of individual trees with the functioning of groups of trees, or stands, at the scale of the yard, block, neighborhood, planning district, census tract, city forest, urbanized area forest (including built-up areas outside the city limits), and the region. It's axiomatic that activities at each spatial and temporal scale influence the scales above and below them. Fitting the urban forest ecosystem into this scheme requires, first, a brief discussion of entities and boundaries.

Boundaries, Gradients, and Linkages

We use boundaries to define entities. An attribute of urban forest structure is its spatial extent. Thus a basic question is "Where is the urban forest's boundary with nonurban vegetation?" This raises allied questions about integrating our understanding of the structure and function of urban, periurban, and exurban forests. Urban forest boundaries can be defined in several ways, such as the political boundaries of the city, the "urbanized area," and the Standard Metropolitan Statistical Area (SMSA). These political, jurisdictional, or morphological criteria can be part of a definition of the urban forest if we augment them with biological or ecological considerations. For example, one attribute of structure distinguishing the urban forest from the wildland forest is the absence of a fully articulated understory. A functional difference is found in the truncated nutrient cycles of the urban forest where people import fertilizers and export fallen leaves.

Undeveloped areas within the city limits or the urbanized areas or the SMSAs may appear, in both structure and function, to be nonurban forests or stands. These woods may have fully articulated vertical stratification, unbroken biochemical cycles, and other structural and functional attributes similar to nonurban forest stands. However, one perspective is that while there is little visible evidence of human influence in these stands, their current and future status is governed entirely by invisible but powerful human processes of land speculation, regulation, taxation, and development. Therefore, their existence is wholly determined by socioeconomic processes based in the general and local urban culture.

Following this line of reasoning raises the question "Aren't all forests whose structure and / or function are predominantly governed by urban-based processes—visible or invisible—to be considered urban forests?" Forests whose current and future structure and function are determined principally by urban forces are certainly different from forests evolving under nonurban conditions. Further development of this line of reasoning leads to (1) a semantic discussion that could easily become preoccupied with what is and what is not urban; (2) the notion that forest science and policy of the future, at least in urbanizing states, will draw more and more upon the accumulated knowledge and wisdom of what is now called urban forestry; or (3) the need for an overarching concept that does not rely on distinctions between urban and nonurban for its efficacy. I will pursue the third alternative in the next section.

As preparation for an elaboration of that concept, visualize the urban forest as part of a mosaic of functionally connected vegetation systems laid out across the landscape like a patchwork quilt. The borders of quilt patches are discrete. On nonurban landscapes, the edges of the pieces in the mosaic are often "fuzzy," more like gradients than discrete boundaries. Urban activities commonly impose discrete boundaries, but there are increasingly cases where the edge of the urban forest is hard to distinguish from that of the surrounding wildland landscape. An example is found along a transect from a city to the dispersed settlement of exurban areas. The gradient of urbanization may occur sharply in some places and more gradually in others. The form, or morphology, of the gradient determines the behavior of functional processes linking urban, periurban, and exurban zones along the gradient. For example, Zembal (1993) found that the endangered lightfooted clapper rail, a bird found in periurban coastal marshes, decreased in numbers when certain patterns of land use intervened to prevent coyotes from preying on introduced foxes and cats, predators of the rail. In addition to wildlife connections, we are beginning to understand energy, water, and nutrient fluxes among urban, periurban, and exurban segments of the landscape gradient.

The innovative work of Pouyat, McDonnell, and Pickett (in press) and others at the Institute for Ecosystem Studies (Millbrook, New York) has helped scientists free themselves of the constraints of viewing the urban forest in jurisdictional or Census Bureau terms by suggesting that we use an "urban-to-rural gradient" of land use intensity to explain the continuum of vegetation change from city to country. Bradley has recently updated a model for understanding the sequence of land uses along this gradient in a way that illuminates the relationship between the hierarchy of urban-influenced uses and vegetation structures that will occur along the gradient (Bradley 1984; Bradley and Bare 1993).

Needed: A Systems Approach Embracing Multiple Scales of Space and Time

There is a need for a systems-oriented approach to guide the core and context of urban forest ecology into the future. This need is nurtured by modern changes in the urbanization process and resulting settlement patterns. It is nurtured also by changes in the kinds of questions being asked of scientists and managers. The ease of telecommunicating with modems and faxes encourages more dispersed settlement in high-amenity exurban wildlands. A portion of the western slope of the Sierra Nevada lies just to the east of Sacramento, California. This area is being populated by people with urban values, urban-generated equity, and urban histories. They have socioeconomic links with the Sacramento urban area. There are biophysical links as well. The structure of the Sacramento urban forest determines how much automobile-emitted air pollution will migrate on the easterly flow of air to the forests surrounding these homes on the west slope of the Sierra. This air pollution can make a critical difference to Sierra forest health. Conversely, the health of the Sierra forests directly affects the people in the Sacramento urban forest. A recent forest fire in the Sierra forests disrupted the water supply and damaged electric-generating capabilities. All of this has an impact on what is done in the Sacramento urban forest. Reduced hydroelectric-generating capacity in the Sierra increases the need for planting energy-saving trees in Sacramento.

Another example points up the need to have a systems approach that can link biophysical and socioeconomic relationships across long distances in a meaningful way. McPherson (1991) calculated that 17 percent of the water requirements of a yard tree planted to reduce a householder's air-conditioning energy use was saved by reducing the power plant's cooling water use. If we can account for changes in the flux of energy, pollution, and water across ecosystem boundaries, as in these examples, we will have a truer accounting of the spatial distribution of benefits and costs resulting from changes in urban forest structures and functions.

Because of the way urban forests are linked by a large number of biophysical and human processes to periurban and exurban forests, we need a concept that can take urban forestry forward in both science and management. The ecosystem concept allows the urban forester to see structural and functional characteristics inside the urban forest in relation to characteristics in adjacent vegetation systems. This helps, for example, in understanding the ecological consequences of a city's expansion into undeveloped wildlands, or of urban exotics escaping into native forest stands. We now look at how the ecosystem concept is beginning to dominate the policy-management context for urban forest ecology, today, and what attributes of the concept may govern the future evolution of both core and context of urban forest ecology.

Renaissance for the Ecosystem Concept

The year 1995 will mark the sixtieth anniversary of the publication of A.G. Tansley's classic paper advancing the notion that "it is the [eco]systems so formed which . . . are the basic units of nature on the face of the earth" (Tansley 1935:299). (Readers interested in the development of the ecosystem concept are referred primarily to Golley 1993, with examples of important papers available in Real and Brown 1991.) It took more than thirty years before a full articulation of the ecosystem concept in natural resource management was published by leading ecologists and resource scientists (Van Dyne 1969). Another quarter century had to pass before federal and state land management agencies adopted ecosystem management as policy. This was as bold and challenging a step as the introduction of Pinchot's "multiple use" concept of the early 1900s. (For a concise review of the early history of forest ecosystem policy, see Caldwell 1970.) The new philosophy requires that the public and private sectors join to plan and manage ecosystems that cross jurisdictional boundaries and comprise multiple ownerships, thus it is particularly important as context for urban forestry. The policy emphasizes that the ecological behavior and condition of these lands will be determined by the coordinated effort of private and public land planners and managers.

State land agencies together with other federal agencies also are adopting ecosystem management as their guiding policy. This is a major shift for land planning and urban forestry over a relatively short time. Because of the rapidity of contextual change, the new approach has its detractors, especially concerning the potential for constraints on private property. Nevertheless, federal and state agencies are adopting the policy, and professional organizations like the Society of American Foresters and the Ecological Society of America are forging their own interpretations of what ecosystem management means. In several cases, urban-based organizations are one to two years into an examination of how they can use and implement this policy for urban forestry. Because the U.S. Forest Service provides most of the funds for urban forestry research and application, we shall examine more closely this organization's articulation of ecosystem management inasmuch as it has become the context in which we do our science and think about urban forest planning and management.

Ecosystem Management as Context for Urban Forest Science and Practice

In February 1994, the Chief of the U.S. Forest Service described the main orientation of ecosystem management as a land policy (USDA Forest Service 1994a): "Ecosystem management is a holistic approach to natural resource management, moving beyond a compartmentalized approach focusing on individual parts of the forest. It's an approach that steps back from the forest stand and focuses on the forest landscape and its position in the larger environment in order to integrate the human, biological, and physical dimensions of natural resource management. The purpose is to achieve sustainability of all resources." Applied to urban forest ecology, this would suggest that we stop viewing urban, periurban, and exurban forests as separate compartments and focus on what connects these systems and how actions in one system affect the operation of the systems linked to it.

In most statements about ecosystem management, Forest Service policy makers have stressed that it is a science-based approach to land management. Thus it is pertinent to our discussion to read how the research branch of this agency has responded to the new policy. This research policy statement sharpens the focus of the evolution and development of urban forest ecology's core and context. The Forest Service Research (FSR) Strategic Plan for the 1990s (USDA Forest Service 1992) defines three high priority research problems that are closely related to the work urban forest scientists do. The headings are taken from the FSR Strategic Plan. I have added comments relating the Plan to urban forest ecology's core and context.

1. *Understanding Ecosystems.* The FSR plan seeks to understand the basic structure and function of ecosystems. Urban forest ecology examines the human-induced attributes of ecosystems, specifically the results of human land use changes, especially those occurring when land is developed and used for residential, commercial, and industrial purposes.
2. *Understanding People and Natural Resource Relationships*. If we are to grasp the ecological changes resulting from shifts in land use, and inform land managers how to anticipate and mitigate them, our research should understand those forces motivating spatial and temporal migrations of land uses. This type of demographic, cultural, and sociological information is required if we are to predict where future uses will occur and what they will do to the land. In order to assign values (benefits or costs) to alternative ecosystem and/or landscape vegetation structures, we will have to understand what drives those values and how they are best expressed, quantitatively and qualitatively, for different groups of people.
3. *Understanding and Expanding Resource Options*. The ecosystem management policy implies very strongly that resource options should be preserved for future generations. To do that requires scientists and managers to employ an ecological accounting system that describes who will benefit and who will pay, when and where, for a given resource decision. Ecosystem management is an accounting system that links resource systems in space and in time.

How Does Urban Forest Science Respond?

For the scientific core and policy context to be efficiently integrated, they must inform one another. Articulation of the general ecosystem management policy followed by the specific ecosystem management research policy is context informing core. How can science respond in order to inform land management? First, it has to identify the central question that will drive the research and advice to management.

That central question can be stated as "How do, and how should, vegetation-soil complexes (and associated biophysical attributes of the ecosystem) change as people settle and urbanize the land?" Or, "How do various land uses, manifested in various spatial-temporal patterns, change forest vegetation and soils at different scales of inquiry?" And "How do these patterns translate into benefits and costs?" Part of the problem is we do not fully understand how to develop information about these altered ecosystems, or parts of them, that can be utilized up and down the interscalar ladder. For example, we can examine how effective a tree's shade is in reducing the need for air-conditioning in a house. Up the spatial scale, we can model a neighborhood or town tree-planting program to increase the magnitude of these savings. Further up the spatial scale, we can design a tree-planting plan for an electric utility's service area that comprises hundreds of such towns. But, this proposed increase in tree density will have unknown effects on micro, meso, and macro climates, as it will on regional water, carbon, and hydrocarbon budgets and on regional air quality. There will be some good effects, some bad effects. So, just as we inquired up the ladder of spatial scales we must inquire up the ladder of temporal scales to see who will bear the costs and who will reap the benefits over time. Perhaps, in this example, the current generation of householders will bear the cost of planting, a second generation will reap the benefits of lower energy bills, and their children's generation will bear the cost of removing a large population of aging trees.

Research must begin by designing studies along three dimensions: (1) from small to large spatial scales, (2) from small to large temporal scales, and (3) from low to high levels of ecosystem disturbance from land uses. The third can be described by experimental sites or domains along a gradient from low to high modification of presettlement ecosystem structure. This can be called the "urban-to-rural land use gradient," though it does not always occur in space as a smooth continuum from city to country. The experimental domains are defined by their land use attributes, such as dense commercial, sparse residential, or transportation corridor. It is at this point that the types of land use have to be limited to focus the research. For example, urban forest ecology should not include wildland recreational use of a nonresidential character (e.g., hiking, camping). Yet the study of how a second-home residential, commercial, and recreational community set in a mountain forest ecosystem is changing the functional role of vegetation and soils takes advantage of the core skills of urban forest ecologists.

If research is conducted at different spatial and temporal scales, it will illuminate the linkages between knowledge at one scale and knowledge at another. This will also reveal the links between the various experimental domains along the gradient of ecosystem modification. For example, learning that increasing the density of tree cover in an urban center loads ozone precursors (volatile hydrocarbons) on downwind forests (near the rural or unmodified end of the gradient) helps us understand the elusive relations that impart a benefit to one domain (in this case the urban center) and a cost to another.

This approach can result in a nested set of studies from smaller to larger spatial and temporal scales. "Nested" means that the studies are designed, often concurrently, so that results generated at one scale can be evaluated for use at smaller and larger scales. This interscalar approach is also helpful in building decision support models that will address the scale of, for example, the homeowner who wants to steward his or her trees through a season of drought (a small spatial-temporal scale) to an interagency council wanting to know what the cumulative effects of private land development in eleven counties will be twenty-five years from now. In evaluating how interscalar information is used, scientists will pay particular attention to two inherent problems: (1) the expansion of error as small-scale information is "blown up" to larger scales; (2) different variables becoming important at different scales, making it difficult to assume that processes operating at one scale operate similarly at another.

What follows is an example of how the questions discussed above can be restated so as to organize studies into two groups. In practice, however, a single study can address both of the following questions:

1. *How has presettlement forest structure and function changed as a result of different settlement patterns*? This work can be conducted at three spatial scales of inquiry—the community, the county, and the multicounty region. There are various temporal scales, but the intent is to speak to the problem of long-term, cumulative effects of settlement, tree removal, soil disturbance, and revegetation, including tree planting. Presettlement, and preurban, forest structure is a baseline condition against which changes can be measured and value judgments made. The scales are described below in terms of political units, but the ecosystem approach precludes drawing discrete boundaries around political or jurisdictional areas. In the measurement of both structure and function, the researcher can include adjacent and surrounding areas by looking one level up the scale.

   Community Scale. Presettlement forest structure can be documented from historical sources for communities in different forest types (McBride and Jacobs 1975, 1986). Contemporary forest structure and function are specified and compared to presettlement structure to learn how community land uses have changed the ecosystem. For example, research in the upper montane Sierra forest type at the community of Bear Valley, California, uses an undeveloped forest nearby as the presettlement "control" forest. The road network and water supply reservoir in Bear Valley have modified the natural distribution of water for meadow and tree growth. A prohibition against tree removal works together with these changes in water distribution to change the trajectory of forest succession from that occurring on the control plot (McBride and Rowntree, in preparation). This study is developing a benefit-cost array that will support a forest management plan for Bear Valley that utilizes knowledge about these changes. (The community wishes to arrest succession and manage for early- to middle-seral plant associations.) It is being determined how representative Bear Valley is of all upper montane Sierra communities and to what degree these results can be extrapolated throughout that forest type. In other regions of the country, community scale studies can, for example, examine how exotic tree species (such as Norway maple) compete with, and replace, natives (such as sugar maple), and how imported natives might change the genetic architecture of a local native population of trees.

County Scale. County general plans specify where residential and commercial land uses can occur and at what densities. An example of research at this scale is to take a county plan and determine what changes will occur to forest and ecosystem structure and function as the general plan is implemented. This determination considers both tree removals and tree plantings that, among other things, bear on natural regeneration, or lack of it, and the mixing of native and exotic species and genetic material. Future projections are augmented by an analysis documenting historically the cumulative effects of land use change to the present. Work at this scale feeds immediately into regional scale research (Zipperer 1993). Once these structural scenarios are complete, studies under question 2, below, can examine changes to function, such as modified countywide water, carbon, and pollutant fluxes.

Regional Scale. Here, information from community and county scales is aggregated upward in spatial scale to a region of about three to eleven counties attempting to discern large-scale patterns in land use induced changes. Often, the region under study contains both developed and undeveloped land, and there is a range of land use/vegetation mixtures. At the regional spatial scale, results are often expressed at large temporal scales. For example, a seven-county study of future impacts of residential and commercial land use change employs county general plans as the data base for constructing a twenty-five year "build-out scenario" that is superimposed on the existing vegetation map for the seven-county area. This describes what vegetation changes would occur if building proceeds according to the counties' general plans (Rowntree et al. 1993). The results form the basis for calculating loss of wildlife habitat, changes in visual and recreational quality, and (see question 2 below) changes in regional water, energy, and pollutant patterns.

2. *How have fluxes or flows of energy, water, and pollutants changed with land use induced changes in forest structure and function*? These studies also should be conducted at several spatial scales and seek to understand modified fluxes into, through, and out of the ecosystem or landscape when land use modifies vegetation and soil structure.

Site Scale. Research in urban forest ecology has, for a number of years, measured changes in energy flux resulting from changes in vegetation, particularly as these relate to human benefits and costs, such as studies measuring energy savings to a homeowner from the reconfiguration of trees and landscape plants around the residence to form windbreaks and shade trees (Heisler 1986, 1990). Associated changes in water utilization can be calculated for any changes in vegetation configuration that may save energy, and the two are combined to estimate a net savings or cost. Basic research at the site scale seeks to understand the flux of incoming solar radiation as it bears on winter solar heating potential (e.g., amounts of winter sun transmission through the crowns of different species), human thermal comfort or stress, and human exposure to the ultraviolet (UV) portion of the light spectrum (Yang et al., in press).

Other flux studies at the site scale link the interaction of energy and water, such as ambient air cooling potential of trees and ground cover in various configurations. Together with the shading potential of trees, evapotranspiration (ET) cooling has the potential for reducing air-conditioning energy use. However, to engage in ET a tree must have access to soil water, and in urban areas soils are often dry due to rainfall runoff from impervious surfaces or too compacted to hold and deliver sufficient water for effective ET cooling. Thus our research must understand the interaction of these factors (Simpson 1993). In addition, there is potentially a wide range of "pumping rates" among the species used for residential and commercial planting. Rates at which different natives and cultivars use water, intercept and transmit solar radiation, produce volatile hydrocarbons, absorb noxious gases, and collect airborne particulates should be examined at the site scale to establish basic flux relations, then extrapolated to larger spatial scales.

Parking lots are important site-scale research locales. Without trees they become urban heat islands, produce high amounts of polluted runoff, and are places where people bear high heat and UV loads. Trees modify the energy and water fluxes so that there is less heat and UV stress on people, less heat is advected (horizontally) to adjacent sites, and less energy and gas (and less air pollution) are used to cool automobile interiors. Research at the site scale can refine these facts, establish relationships, quantify benefits and costs, and form the basis for aggregation to larger spatial scales.

Community Scale. Because towns and cities are political jurisdictions, this scale is useful in providing certain types of planning and management information dealing with energy, water, and pollutant flux. Other kinds of information are better passed to managers at the county or regional scale. Some scientific questions are more effectively addressed by adding a scale between site and community, such as "neighborhood." For example, Simpson (1993) seeks to answer the question "What is the minimum area of trees, at high urban densities, required to achieve measurable ET cooling?" This requires testing at several scales ranging from site to community. Similarly, Nowak (1994a, b) employs measurements of urban forest leaf area at different scales to estimate the quantity of pollutants removed from the atmosphere.

Scientists can develop a typology of experimental sites along the urban-to-rural gradient, such as high density urban commercial areas, parking lots, quarter acre single-family residential communities, and freeway interchanges. For each type, the range of fluxes for water, energy, and pollutants can be established from empirical measurements and simulation studies that rely on inherent site attributes as well as on the way the site is linked to adjacent sites.

County and Regional Scales. Models of water, energy, and pollutant flux can be constructed at the county and regional scales based on relations established at the site and community scales. At the regional scale, we can begin to see interactions between large urbanizing areas and adjacent forested areas. For example, three of the major urbanizing regions in the West—the Colorado Front, the Salt Lake Valley, and the Sacramento-San Joaquin Valley—are adjacent to major forested mountain ranges, and the urban air pollution affects vegetation, soils, and runoff quality in the mountains. Because these cities rely on mountain runoff for water, air pollutants can theoretically be returned to the cities in the water. This is an example of how accounting for fluxes between two ecosystems can illuminate the role of the urban forest. Research can now begin to model the fraction of gaseous and particulate air pollution removed from the airstream by various densities and configurations of urban vegetation in both present and future urbanized areas. This will estimate reductions in future air pollution loads on adjacent mountain forest lands. The model can also estimate the production of ozone precursors (volatile hydrocarbons) by the urban vegetation, water use and the effects of runoff, energy use, and carbon sequestering and storage.

Difficulties with the Ecosystem Concept

Whether it is used in core scientific studies or in the policy and management context, the ecosystem concept is not without its problems. For natural systems, some of these difficulties are minimized. For modified systems where humans are rearranging structure and function, some of these difficulties are exacerbated. The following discussion includes, but is not limited to, problems that confront urban forest ecologists.

Where Do Humans Fit In?

There are few ecosystems today that haven't been modified, directly or indirectly, by humans. However, a question that comes up early in any discussion about applying the ecosystem concept to human-modified landscapes is "How does one accommodate the activity of humans in a model of structure, function, and flux?" A corollary is "Are humans internal or external to the ecosystem?" (USDA Forest Service 1994b). They can be viewed usefully as both internal and external components. That is, humans are tool-using "megafauna" operating within an ecosystem, albeit with more consequence than other fauna. They rearrange the flux of energy, water, and matter. (In smaller amounts, so does a hummingbird.) In the Sierra Nevada Ecosystem Project (SNEP), analysis proceeds on the assumption that ecosystems are being modified by humans internal to Sierran ecosystems, but also as forces producing fluxes into those systems from outside (SNEP 1994).

The point is that ecosystem theory and ecosystem science easily accommodate human activity. In fact, the usefulness of the ecosystem concept to human society may be largely in the area of understanding and guiding interactions between humans and nature. Ecosystem theory incorporates feedback loops, and these can be used to clarify human-ecosystem interaction. For example, humans perceive a given ecosystem state. They evaluate it in relation to their needs and usually make changes. They watch how these changes affect system properties and processes and evaluate the new system state. Further changes are made, and so on. The feedback of human evaluation and modifications into the sequence of ecosystem states either amplifies or dampens the degree to which an ecosystem's trajectory will vary from what would have occurred naturally.

Complexity

Reality is complex, and the ecosystem concept is a mental construct that attempts to model the real world. When it does fairly well at that, it approaches a complexity that may frustrate its use in science and / or management. The ecosystem concept requires a high level of scientific participation if it is to reach its potential. Can the core of urban forest ecology participate at the required level? As a science, urban forest ecology is just beginning to deal with complex systems.

For example, scientists and practitioners have long believed that adding trees will make a city cooler. Tree-planting programs and demonstration projects have been based on this idea. Evapotranspiration (ET) cooling is one of the oldest hypotheses in urban cumatology, yet the scientific information is inadequate to indicate how many degrees reduction in average air temperature will occur with an addition of a number of trees in any given pattern (Simpson 1993). The physics of evapotranspiration and heat transfer suggest that the relationship is based on sound theory. If so, why don't we have a more precise understanding of the relationship?

First, the relationship, like so many aspects of urban forest ecology, is more complex than it seems. There is great variation in the rate that different species transpire water. Many urban trees have too little water and too much radiation, and consequently close up their stomata and don't transpire for much of the day. An excessive density of trees restricts airflow, and heat and moisture build up in and below the canopy. Years ago, it was sufficient to assume there was roughly a linear relationship: more trees equals a cooler city. This assumption adequately supported tree-planting programs. Today, however, cities and electric utilities demand a more precise relationship. How many trees in what configuration will bring down temperatures by how much over how large an area? The more precise relationship is required for benefit-cost analyses, yet it will be years before scientists can produce these numbers for planners and managers.

A similar problem may develop in the context of ecosystem management. At first, the idea is attractive, but we don't appreciate the information and knowledge requirements of implementing it. As time passes, urban forest management becomes committed to it, but scientists cannot participate at the level required to make ecosystem analysis and management work. At the outset of this chapter, it was stated that we have to understand how the core and context of urban forest ecology inform and support one another. For ecosystem management to work—given its inherent complexity—there has to be (1) improved communication between scientists and managers (i.e., core and context need to efficiently inform one another), and (2) a realistic ratio between program and science funding.

Program funds nurture the activities of urban forestry which in turn create the demand level for scientific information. Over the last fifteen years, the ratio between Forest Service program funds (administered to the states by the State and Private Forestry branch of the agency) and funds dedicated to urban forest research has been in a range between 10:1 and 20:1 (program to research). As urban forest ecology shifts to a higher plane of scientific expectation in the context of ecosystem management, the ratio will have to change in order to reduce the disparity between demand for knowledge and the scientific core's ability to provide it.

Uncertainty

Uncertainty in ecosystem management might be described as the disparity between what we know and what we believe we should know about how these systems work. Because the ecosystem concept is a more complete representation of reality than previous mental constructs, one feels closer to the truth. But, because it is difficult to meet the information demands of this more complex view of the world, there will be more uncertainty. Thus scientists and managers move from one kind of uncertainty—where our models were imperfect representations of reality—to another, where our models are more complete, but we haven't the information power to document and run them with confidence.

According to Frank Golley, ecologist and historian of the ecosystem concept, some scientists have charged that the concept is too deterministic, giving the false impression that we can control these systems (Golley 1993:190). These critics say that deterministic cause-and-effect models do not take into consideration the inherent chaos in nature, particularly in disturbed systems. This charge is important to our discussion because urban forest ecosystems are disturbed ecosystems. Golley agrees with these critics that disturbed ecosystems tend to be more chaotic, but he makes the error of lumping natural and human disturbances together (p. 197). There is an important distinction that is particularly relevant to urban forest ecology.

Human-disturbed ecosystems, including urban forest ecosystems, may be less chaotic than many natural-disturbed systems because of the relatively predictable behavior of human institutions compared to natural disturbances. Of course, natural systems into which humans have just begun to

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intervene can become quite unpredictable and chaotic because of the many yet unknown interactions between human and natural processes. In established urban forest ecosystems, however, the human hand is much more dominant, and thus control over fluctuations is greater. Internally, chaos and uncertainty are less of an issue than in natural systems. It is externally, where urban forest ecosystems are linked with natural ecosystems (particularly those functioning under one or more disturbance regimes), that there is a high potential for chaos and uncertainty.

Multiple Ownerships

Because ecosystems include more than one landowner or manager, a challenge for ecosystem management is having all landowners understand and accept the concept. This issue is of particular interest to urban forestry professionals. Lynton Caldwell, a senior scholar of land and forest policy, advocated an ecosystem approach to land management twenty-five years ago, stating: “the natural processes of physical and biological systems that comprise the land do not necessarily accommodate themselves to the artificial boundaries and restrictions that law and political economy impose upon them” (Caldwell 1970:203). There seems to be no disagreement that parts of a system must be coordinated in order for that system to run efficiently and accomplish its objective.

Federal and state plans to implement ecosystem management respect private property rights. Yet the trend over the last century has been to gradually curtail private property rights as society has learned how the environment works and about the importance of property owners’ cooperating for the common good. We are still on the steep part of the learning curve regarding how our individual activities affect the ecosystem in which we live and the ecosystems to which ours is coupled. Thus the challenge is in education rather than regulation. This places even more responsibility on scientists to explain what ecosystems are, how they work, and how landownership and land management affect their structure, function, and long-term trajectory.

Members of the urban forestry community will be interested in how this effort proceeds, because they have been involved for years in educating homeowners and commercial property owners toward a better understanding of how their individual properties contribute to the urban forest ecosystem. Without a doubt, this is another critical topic on which the core and context of urban forest ecology need to inform one another.

Conclusion: Science and Context

The core and context of urban forest ecology can take the first steps toward ecosystem management by boiling the concept down to a fundamental principle on which scientists and managers can focus. It will not be new, for it has been part of conventional wisdom in land management, indeed in our view of the world, for years. It is that everything is related, and nothing changes without having consequences throughout the system and adjoining systems. The task is to understand and account for these changes. Theoretically, ecosystem management must account for all changes, with each change given a human value—a magnitude of benefit or cost expressed quantitatively or qualitatively. While this may be difficult if not impossible for a while, ecosystem management makes explicit the responsibility for scientist and manager to make as full an accounting as possible. Thus the ecosystem concept infuses into both the core and context of our field not only a better representation of reality but a higher level of responsibility.

During this paradigm shift, I am optimistic about the core and context of urban forest ecology advancing in a mutually beneficial manner. The basis for this optimism is exemplified by a course developed in 1993 by a group of urban forest ecologists—planners, managers, and scientists. Entitled “An Ecosystem Approach to Urban and Community Forestry” (USDA Forest Service 1993), this week-long workshop was tested in several cities in the Midwest and East, where the students ranged across the spectrum of urban forestry professionals. The Urban Forestry Center of the University of Pennsylvania's Morris Arboretum conducted an evaluation of the course and concluded that it successfully conveyed ecosystem principles and management strategies for urban forestry (R.L. Neville, USDA Forest Service, Syracuse, New York, pers. comm., 1994). The course is being fine-tuned for a second round of offerings in 1994–95.

Kai Lee begins the preface of his recent book, *Compass and Gyroscope*, with the observation that "civilized life cannot continue in its present form" (Lee 1993). The sheer number of people, combined with our powerful technologies, guarantees that we will alter the planet on which we must continue to evolve. *Homo sapiens* is trying to adjust quickly to changing ecosystems we don't understand. The peril lies in our fear of complexity and our desire to have science tell us what to do. Kai argues that the response to that fear is in an approach called "adaptive management" where the best science, albeit incomplete, is brought to bear on an ecosystem, management is implemented under rigorously monitored conditions, and adaptations in management are made as the feedback from monitoring teaches us more about the way the ecosystem behaves. Adaptive management is being tested in rural ecosystems, such as the Hayfork Adaptive Management Area in northern California, which is part of the implementation of the Forest Ecosystem Management Assessment Team's study of the northern spotted owl region.

Adaptive management areas for urban forest ecosystems must be established soon, for it is this approach that will test the ability of scientists and managers to cooperate in apprehending these complex systems. (See McPherson 1993 for a discussion of urban forest ecosystem monitoring.) This cooperation can be enhanced if there is a conjunction of meaning and purpose founded on common vocabulary and concepts. And this conjunction will occur if the two domains of urban forest ecology—scientific core and policy context—can continue to inform one another as they shift and evolve.

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What Is the Form of a City, and How Is It Made?

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Three branches of theory endeavor to explain the city as a spatial phenomenon. One, called "planning theory," asserts how complex public decisions about city development are or should be made. Since these understandings apply to all complex political and economic enterprises, the domain of this theory extends far beyond the realm of city planning, and it has been well developed in those other fields. So it has a more general name: "decision theory."

The second branch, which I call "functional theory," is more particularly focussed on cities, since it attempts to explain why they take the form they do and how that form functions. This is a reasonably thick theoretical limb—if not as robust as decision theory—and engages renewed interest today. I have summarized its leading ideas in appendix A, and there, from a safe distance, point to some of the more common blemishes on this limb.

The third branch, spindly and starved for light, but on which so many actions are hung, is what I would call "normative theory." It deals with the generalizable connections between human values and settlement form, or how to know a good city when you see one. This is our concern.

As on any healthy tree, the three branches should spring securely from a common trunk. Unlike the branches of trees we know, they should not diverge. They should interconnect and support each other at many points. A comprehensive theory of cities would be a mat of vegetation, and some day the branches will no longer exist in separate form. While working perilously far out on the weakest branch, we must be aware of the other two and look for favorable places to insert a graft.

So this chapter scans planning theory and functional theory, the two companion branches to our own. It also sets forth what I mean by the "form" of the city. Otherwise, what are we talking about?

Functional Theories

Almost all recent theories about the spatial form of urban settlements have been theories of urban function. They ask: "How did the city get to be the way it is?" and that closely related question, "How does it work?" One cannot ask, "What is a good city?" without some convictions about answers to those previous questions. Theories of function, in their turn, cannot be constructed without some sense of "goodness," which allows one to focus on the essential elements. All functional theories contain value assumptions—most often hidden ones—just as all normative theories contain

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assumptions about structure and function. Theoretical developments in one arena impose themselves on the other. A developed theory of cities will be simultaneously normative and explanatory.

As yet, there is no single theory of city genesis and function that brings together all the significant aspects of city life. These theories look at the city from quite different points of view, and some particular viewpoints are much more fully developed than others. Appendix A is a brief review of those reigning theories, grouped by the dominant metaphors by which they conceive of the city. These metaphors control the elements to be abstracted and shape the model of function.

The city may be looked on as a story, a pattern of relations between human groups, a production and distribution space, a field of physical force, a set of linked decisions, or an arena of conflict. Values are embedded in these metaphors: historic continuity, stable equilibrium, productive efficiency, capable decision and management, maximum interaction, or the progress of political struggle. Certain actors become the decisive elements of transformation in each view: political leaders, families and ethnic groups, major investors, the technicians of transport, the decision elite, the revolutionary classes.

Common Deficiencies

From the standpoint of normative theory, these functional theories have some common deficiencies. Perhaps it is these very deficiencies which allow me (or is it the pervading dullness which motivates me?) to compress this extensive literature into a single appendix. If we had a compelling functional theory, no book on city values could be written without it. As it is, these theories depend on values which are unexamined and incomplete. Second, most of them are essentially static in nature, dealing with small shifts, balancings, or external changes which will be damped out, or lead to final explosions, or, at most, cause radical jumps that reach some new and endless plateau. None deals successfully with continuous change, with incremental actions that lead in some progressive direction. Third, none of these formulations (except the historical, or "antitheoretical," view) deals with environmental quality, that is, with the rich texture of city form and meaning. Space is abstracted in a way that impoverishes it, reducing it to a neutral container, a costly distance, or a way of recording a distribution which is the residue of some other, nonspatial, process. Most of what we feel to be the real experience of the city has simply vanished. Fourth, few of the theories consider that the city is the result of the purposeful behavior of individuals and small groups, and that human beings can learn. The city is the manifestation of some iron law or other, rather than the result of changing human aspirations.

It surprises no one to hear that it is impossible to explain how a city should be, without understanding how it is. Perhaps it *is* surprising to encounter the reverse: that an understanding of how a city is depends on a valuing of what it should be. But values and explanations seem to me inextricable. In the absence of valid theory in either branch, concepts elaborated in the one must employ provisional assumptions from the other, while making that dependence explicit and maintaining as much independence as is possible.

In distinction to functional and normative theory, planning theory deals with the nature of the environmental decision process—how it is and should be conducted. This is a subject treated at length in many other sources. Since normative theory is intended to be useful in creating better cities, clearly it must be aware of the situations in which it is likely to be used.

The Process of City Building

Cities are built and maintained by a host of agents: families, industrial firms, city bureaus, developers, investors, regulatory and subsidizing agencies, utility companies, and the like. Each has its own interests, and the process of decision is fragmented, plural, and marked by bargaining. Some of these agents are dominant, leading; others will follow those leaders. In this country, the leading agents tend to be the great financial institutions, which establish the conditions for investment; the major corporations, whose decisions as to the location and nature of productive investment set the rate and quality of city growth; and the large developers, who create extensive pieces of the city itself. On the public side, we must add the major federal agencies, whose policies of taxation, subsidy, and regulation merge with the actions of private finance to set the investment conditions, and the large, single-purpose, state or regional agencies which are charged with creating highways, ports, water and disposal systems, large reservations, and similar major chunks of city infrastructure. The basic patterns set by these form givers (to appropriate an egotistical term from architecture) are filled in by the actions of many others, in particular the location decisions of individual families and of firms of modest size, the preparatory activities of real estate speculators, small developers, and builders, and the regulatory and supporting functions of local government. The latter agencies, although unable to control the main currents, do much to set the quality of a settlement, through their fire, building, and zoning codes, by the way they service development with schools and roads and open space, and by the quality of those services: education, policing, and sanitation.

Role of Theory in City Building

This process has certain marked characteristics. The leading agents, who have such a tremendous influence, do not control city development in any directed, central fashion. Typically, they are single-purpose actors, whose aim is to increase their profit margin, complete a sewer system, support the real estate market, or maintain a taxation system which generates sufficient revenue (and yet provides sufficient loopholes). These purposes are usually remote from the city form that they shape. No one takes anything like a comprehensive view of the evolving spatial structure, except perhaps the local planning agency, which is one of the weaker actors. When this is added to the great number of agencies who have *some* role to play in the game, and whose acts, however passively responsive, have great cumulative power, then we have a city-building process which is complex and plural, marked by conflict, cross-purpose, and bargaining, and whose outcome, while often inequitable or even unwanted, seems as uncontrollable as a glacier.

Yet it is controlled, if not with conscious purpose, by the leading actors we have named, and it can also be modified consciously by public effort, although with only partial (and sometimes with surprising) effect. Most purposeful public actions, beyond the single-minded decisions of public works agencies, are reactions to pressing difficulties, which are carried out with haste, poor information, and no theory, and which are designed to return the system to some previous condition.

Comprehensive theory might seem of remote value in such restrictive situations, and yet it is just here that a coherent theory is so badly needed. It is needed to make restricted actions effective, as well as to enlighten the inevitable political bargaining, or even to point to needed changes in the decision process itself. Thus structural theory guides the quasi-intuitional actions of a trained engineer in some emergency, and military theory illuminates the confused art of war. But theory must be of a certain kind, if it is to be useful. It must speak to purposes, and not about inevitable forces. It must not be esoteric, but be clear enough to be useful to all sorts of actors. It must be usable in rapid, partial decisions and in the constant "steering" of policy as the complex settlement changes. Indeed, as we shall see, various normative theories of the city have been used in just that way, however misguided they might have been.

Creating cities can be quite different in other societies. The power to decide may be highly decentralized but also egalitarian, instead of decentralized but unequal, in the United States. More often, it is more highly centralized. The motives of power may differ. The basic values of the society may not only be different from our own, but also more homogeneous and stable. Decisions may be made according to tradition, without explicit rational analysis. The level of material resources, of skill and technology, can be substantially lower, which changes the constraints and shifts the priorities. The rate of change may be faster or slower. All these variations in the dimensions of the decision process make varying demands on any normative theory. A general theory must be able to respond to those differences. At the same time, there do seem to be certain regularities in the contemporary decision process, at least within the large urban settlements which dominate our landscape today. We find plurality, complexity, and rapid change everywhere.

What is the Problem?

Whenever any significant actor, public or private, engages to make an important decision in this complex environment, that effort to decide has typical features. The first question is: "What is the problem?" The consciousness of a problem is always an integrated perception, however vague, that is simultaneously an image of the situation and its constraints, of the goals to be achieved, of who the clients are, and what kinds of resources and solutions are available. Problems do not exist without some inkling of all of these features, and the decision process is no more than a progressive clarification of this set, until a firm basis for action is found—one in which solution, aims, clients, resources to be used, and perceived situation all seem to match one another. To achieve this mutual fit may require modifying any or all of these separate features. But the initial concept of the problem is crucial. Often enough, it is wrong to begin with—the situation so poorly understood, the clients so restricted, the aims or the solution envisaged so inappropriate, that nothing can be done except to make things worse.

Submission, Revolution, Therapy, Reform

Some of the preconceptions that accompany initial problem definition are fundamental. One is the view of the basic type of response that is appropriate For example: seeing a difficulty, one may not try to remove it, but simply seek to understand it and to predict its future course, so that one can adapt, survive, and prosper if possible. Grass bends to the wind, but the "street-wise" person does more: he takes advantage of the wind's momentary course and power.

At the other end of the scale, one may be convinced that a fundamental change in the rules of the game is essential. Society must make a radical shift. An environmental problem is the occasion for motivating others to that radical change. Nothing less than this great leap will do, and so a housing shortage is best converted into a confrontation and a revolutionary lesson. Or, following another alternative, one makes a persuasive model of a habitat or society which is radically better than the present one, but which can be realized gradually.

Between passive response and great leaps lies the strategy of making repeated changes in selected factors, in order to improve the whole piecemeal. One such gradualist approach is to change persons so that they can function better in an existing context. People's lives are enriched by learning to observe and understand their own city neighborhoods, and they begin to come to grips with their own life situations. Teaching children or the handicapped how to get about the city, or homeowners how to make a garden or repair a house, are other examples of this mode of intervention.

Alternatively, one may focus on modifying the environment, the better to fit the intentions of the person, which is the typical planning approach. The normative theory we have in mind is designed for use in this environment-modifying, piecemeal, and gradualist mode. However, it can also supply educative information, or the fuel for a more radical change. Changing minds, changing society, or even changing nothing at all, may in many situations be a more appropriate response than changing the environment. Most people are convinced of the eternal rightness of their own favorite mode. On the contrary, a well-formulated problem always entails prior consideration of the proper scope and mode of intervention.

Who should Decide for Whom?

It is also crucial to decide who the clients are. Who should make the decisions? In whose interests should those decisions be made? Are deciders and decided-for the same? The clients identified at the beginning of a decision effort usually exclude certain vital interests. Bringing in a new client, in the course of the decision, is delicate work, sure to be resisted by those already at the table and likely to impede any decisive action.

A highly decentralized decision process, in which the immediate users of a place make the decisions about its form, is a powerful ideal. It reinforces their sense of competence, and seems more likely to result in a well-fitted environment, than if they are excluded. The basic view is philosophical anarchism. But there are users whom we judge incompetent to decide: too young, too ill, or under coercion. There are indivisible goods, like clean air, that affect millions of users simultaneously. There are places used by numerous transient clients, such as a subway. There are conflicting interests, users who succeed each other, and distant persons whose interests are partially affected by some local use. There are unknown clients, people who are not there yet, or who have not yet been born. There are clients who are unaware of their own requirements, or of what they might value if they had the opportunity. All these difficulties, plus the political troubles inherent in any effort to shift control to new clients as a problem develops, give planning decisions their characteristic tone of ambiguity, conflict, and fluidity.

Other professionals hold a contrary view: all crucial decisions are inevitably, or even preferably, made by a powerful few. Since dominant interests cannot be suppressed, and since some professionals are uniquely endowed by their marvelous training and ability to solve environmental problems, those gifted ones should stand beside the seats of power. Problems are complex, values subtle, and solutions specialized and delicate. Find an expert who can grasp the situation, and give him room to work. Some of our more remarkable environments arose from heroic leadership of that kind, but few are well fitted to the purpose of their users. This model performs best when values are clear and common, and problems largely technical.

Planners and Informers

Professional planners take on many different roles in this complicated decision landscape. Most of them, perhaps, are project planners, working for some definite client, such as a corporation or government agency, and preparing a solution to some limited, well-defined problem, according to an explicit set of purposes. Here they are sheltered from most of the debates about the client or the mode of intervention. Those crucial decisions have been made for them.

Other planners consider themselves to be working in the public interest. Since they must work near some center of power in order to be effective, they are beset by the issues I have sketched out above: who is the client? who should make the decisions? how should goals be determined? are there in fact any common interests? how can I know them? how can power be effective without overriding those common interests with its own aims? At times, planners in this public interest role may try to avoid some of these dilemmas by attending primarily to the decision *process*—keeping it as open and equitable as possible—without attempting to set goals or to recommend solutions.

Retreating still further from decision, and in despair of discovering the public interest, many planners take on the primary role of informers (not spies!). They create accurate and timely information for public use: descriptions of the present state and how it is changing, predictions about coming events, and analyses of the results to be expected from this or that line of action. Actual plans and decisions are left to others, but presumably they will be better decisions because better informed. If these planners have strong beliefs about the decision process, they may shape their information especially for the use of certain groups: for decentralized users, for radical reformers, or for central decision makers. Alternatively, as I have mentioned above, they may think of themselves primarily as teachers, involved in educating, and so in changing, the public.

Lastly, some professionals are primarily advocates. They may be the advocates of some idea—such as new towns or bicycle paths or houseboats—in which case they must organize their own client base. These are pattern makers, who hope to be effective through the persuasiveness of their ideas. If sufficiently radical, they create utopias, patent models for a new society.

Advocates

More frequently, they will be advocates of some interest group—a social class, a corporation, a neighborhood—and press that interest vigorously, in competition with other contenders. Many professionals, of course, are advocates without being aware of it, while others take a more conscious position. They look on society as highly connected but irredeemably plural and contradictory. All decisions are made by struggle and compromise; few values are held in common. Inevitably, any professional works for one group or another. Some will add: but this system is unjust, since some groups have little power and no hired advocate. Therefore, a professional of conscience works for those poorly represented groups, advocating their interests as forcefully and as narrowly as a planner hired by a real estate developer.

Advocates, informers, project designers, and public planners—these are perhaps the predominant professional roles today. Their theories and models, usually implicit and unexamined, play an important part in environmental decision, amid all the customary confusion of that process. Unmanageable problems are made manageable by restricting the clients to be attended to, by taking a model of change and thus a type of solution for granted, by assuming a narrow set of operative values, and by controlling the supply of information. Information of a fairly broad range is often gathered while initiating planning studies. In the press of decision, only a small portion of that information is used, and that is the portion which accords with the models already in the decider's head. Developing a theory that is sufficiently concise and flexible to be used under pressure is one way of directing the attention of decision makers to one set of issues rather than another.

The process of decision (and of design, which is a subset of decision) is one of managing the progressive development and definition of a problem, to the point where situation, client, aim, and solution are sufficiently well-fitted to take action. This process, when applied to large environments at least, has difficulties which seem to be common throughout the world. It likewise poses some common issues: such as those about the nature of the client, the model of change and its management, and the nature of the professional role. It has consequences for the ethics of planning, as well.

Planning, to my mind, has its own special interest in any public debate. I would characterize that special interest as one which is prejudiced in favor of five things (besides its focus on spatial form and form-associated institutions): the long-term effects, the interests of an absent client, the construction of new possibilities, the explicit use of values, and the ways of informing and opening up the decision process. These are professional counterweights to the de-emphasis of those considerations by other actors.

Descriptions of the City Form

But what is this city, that we dare to call good or bad? How can we describe it in ways that different observers will confirm, and which can be related to values and performance? This simple step conceals unseemly difficulties.

Settlement form, usually referred to by the term "physical environment," is normally taken to be the spatial pattern of the large, inert, permanent physical objects in a city: buildings, streets, utilities, hills, rivers, perhaps the trees. To these objects are attached a miscellany of modifying terms, referring to their typical use, or their quality, or who owns them: single-family residence, public housing project, cornfield, rocky hill, ten-inch sewer, busy street, abandoned church, and so on. The spatial distribution of these things is shown on two-dimensional maps: topographic maps, land use maps, street maps with notations, utility networks, maps of housing condition. These maps are accompanied by population counts (divided into classes of age, sex, income, race, and occupation), and usually by maps showing the spatial distribution of population (by which is meant where people sleep). Then there are descriptions of the quantity of traffic on the various main arteries, and statistics on the principal economic activities (that is, only those human activities which are part of the system of monetary exchange), and data on the location, capacity, and condition of particular public or semi-public buildings or areas, such as schools, churches, parks, and the like. These descriptions are familiar, and they are infected with difficulties, which are also familiar to anyone who has handled them. Lay citizens are baffled by these maps, graphs and tables. This might be taken as a sign of the scientific sophistication of the field, except that professionals have the same troubles.

Definition of Settlement Form

The fundamental problem is to decide what the form of a human settlement consists of: solely the inert physical things? or the living organisms too? the actions people engage in? the social structure? the economic system? the ecological system? the control of the space and its meaning? the way it presents itself to the senses? its daily and seasonal rhythms? its secular changes? Like any important phenomenon, the city extends out into every other phenomenon, and the choice of where to make the cut is not an easy one.

I will take the view that settlement form is the spatial arrangement of persons doing things, the resulting spatial flows of persons, goods, and information, and the physical features which modify space in some way significant to those actions, including enclosures, surfaces, channels, ambiences, and objects. Further, the description must include the cyclical and secular changes in those spatial distributions, the control of space, and the perception of it. The last two, of course, are raids into the domains of social institutions and of mental life.

The cut is not trivial, however, since most social institutional patterns are excluded, as well as the larger part of the realms of biology and psychology, the chemical and physical structure of matter, etc. The chosen ground is the spatiotemporal distribution of human actions and the physical things which are the context of those actions, plus just so much about social institutions and mental attitudes as is most directly linked to that spatiotemporal distribution, and which is significant at the scale of whole settlements. This choice is more fully discussed, and compared with conventional descriptions, in appendix B.

Social and Spatial Structure

No one would claim that to describe these things is to grasp a human settlement in its fullness. We must see any place as a social, biological, and physical whole, if we mean to understand it completely. But an important preliminary (or at least a necessary accompaniment) to seeing things whole is to define and understand their parts. Moreover, social and spatial structure are only partially related to each other—loose coupled, as it were—since both affect the other only through an intervening variable (the human actor), and both are complex things of great inertia. For me, the acts and thoughts of human beings are the final ground for judging quality. These apparently ephemeral phenomena become repetitive and significant in at least three situations: in the persistent structure of ideas which is a culture, in the enduring relationships between people which are social institutions, and in the standing relations of people with place. I deal with the last. While the social, or economic, or political aspects of settlements are rather well-defined—and often too narrowly defined—the physical aspect is put so uncertainly that it is difficult to see whether it plays any role at all.

The cut I suggest seems to be the closest one that can be taken, that still permits us to comment on the contribution of spatial pattern to human aims. Moreover, it is a coherent view, since its common core is the spatial distribution, at a given scale, of tangible, physical persons, objects, and actions. It has the advantage of growing out of the commonsense view of the environment, while regularizing and expanding it.

Building a full theory will be a long-range effort, if it is to be a theory which deals with form and process, and which is an understanding, an evaluation, a prediction, and a prescription, all in one. It will hinge on purposeful human behavior and the images and feelings that accompany it. This is the joint at which all three branches of theory should grow together. Our particular subject, which is normative theory, must be considered with that possibility in mind. Such normative theory as exists today is disconnected from the other theoretical realms, but carries hidden assumptions about function and process.

Requirements for Normative Theory

There are certain requirements, then, for any useful normative theory of city form:

1. It should start from purposeful behavior and the images and feelings which accompany it.
2. It should deal directly with settlement form and its qualities, and not be an eclectic application of concepts from other fields.
3. It should connect values of very general and long-range importance to that form, and to immediate, practical actions about it.
4. It should be able to deal with plural and conflicting interests and to speak for absent and future clients.
5. It should be appropriate to diverse cultures and to variations in the decision situation (variations in the centralization of power, the stability and homogeneity of values, the level of resources, and the rate of change).

6. It should be sufficiently simple, flexible, and divisible that it can be used in rapid, partial decisions, with imperfect information, by lay persons who are the direct users of the places in question.
7. It should be able to evaluate the quality of state and process together, as it varies over a moderate span of time.
8. While at root a way of evaluating settlement form, the concepts should suggest new possibilities of form. In general, it should be a possible theory: not an iron law of development, but one that emphasizes the active purposes of participants and their capacity for learning.

Where shall we look for the material for such a theory?

Dimensions of Performance

Performance characteristics will be more general, and the easier to use, to the degree that performance can be measured solely by reference to the spatial form of the city. But we know that the quality of a place is due to the joint effect of the place and the society which occupies it. I can imagine three tactics for avoiding the necessity of taking the entire universe into account in this attempt to measure city performance. First, we can elaborate those linkages between form and purpose which exist because of certain species-wide or human settlement-wide regularities: the climatic tolerances of human beings, for example, or the importance of the small social group, or the very general function of any city as a network of access. Second, we can add to the description of the spatial form of a place those particular social institutions and mental attitudes which are directly linked to that form and repeatedly critical to its quality, as I have already done at the end of this chapter. Both of these tactics will be employed below.

Third and last, however, we must realize that it would be foolish to set performance *standards* for cities, if we mean to generalize. To assert that the ideal density is twelve families to the acre, or the ideal daytime temperature is 68° F., or that all good cities are organized into residential neighborhoods of 3000 persons each, are statements too easily discredited. Situations and values differ. What we might hope to generalize about are performance *dimensions*, that is, certain identifiable characteristics of the performance of cities which are due primarily to their spatial qualities and which are measurable scales, along which different groups will prefer to achieve different positions. It should then be possible to analyze any city form or proposal, and to indicate its location on the dimension, whether by a number or just by “more or less.” To be general, the dimensions should be important qualities for most, if not all, persons and cultures. Ideally, the dimensions should also include all the qualities which any people value in a physical place. (Of course, this last is an unbearably severe criterion.)

“Durability”

For example, we might consider *durability* as a performance dimension.[1] Durability is the degree to which the physical elements of a city resist wear and decay and retain their ability to function over long periods. In choosing this dimension, we assume that everyone has important preferences about the durability of his city, although some want it evanescent and others would like it to last forever. Furthermore, we know how to measure the general durability of a settlement, or at least how to measure a few significant aspects of durability. A tent camp can be compared to a troglodyte settlement, and, given the values of a particular set of inhabitants, we can tell you which one of

[1] But we won’t. This is a red herring.

them is better, or people can make that evaluation for themselves. They can also decide how much durability they are willing to give up in return for other values. Perhaps we can show that very low or very high durabilities are bad for everyone, and so we identify an optimum range. Although the linkage of durability to basic human aims is only a chain of assumptions, we believe that the assumptions are reasonable. Correlations of durability with preference exist, and people are content to use this idea as a workable intermediate goal. Meanwhile, its connection to city form—to such concrete physical characteristics as building material, density, and roof construction—can be explicitly demonstrated.

Criteria for Performance Dimensions

To be a useful guide to policy, a set of performance dimensions should have the following characteristics:

1. They should be characteristics which refer primarily to the spatial form of the city, as broadly defined above, given certain very general statements about the nature of human beings and their cultures. To the extent that the value set on those characteristics varies with variations in culture, that dependence should be explicit. The dimension itself and its method of analysis should remain unchanged.
2. The characteristics should be as general as possible, while retaining their explicit connection to particular features of form.
3. It should be possible to connect these characteristics to the important goals and values of any culture, at least through a chain of reasonable assumptions.
4. The set should cover all the features of settlement form which are relevant, in some important way to those basic values.
5. These characteristics should be in the form of dimensions of performance, along which various groups in various situations will be free to choose optimum points or "satisficing" thresholds. In other words, the dimensions will be usable where values differ or are evolving.
6. Locations along these dimensions should be identifiable and measurable, at least in the sense of "more or less," using available data. They may be complex dimensions, however, so that locations on them need not be single points. Moreover, the data, while conceivably available, may for the present escape us.
7. The characteristics should be at the same level of generality.
8. If possible, they should be independent of one another. That is, setting a level of attainment along one dimension should not imply a particular setting on some other dimension. If we are unable to produce uncontaminated dimensions of this kind, we can settle for less, if the cross-connections are explicit. Testing for independence will require detailed analysis.
9. Ideally, measurements on these dimensions should be able to deal with qualities which change over time, forming an extended pattern which can be valued in the present. More likely, however, the measurements will deal with present conditions, but may include the drift of events toward the future.

Previous Attempts

There have been many previous attempts to outline a set of criteria for a "good city." The dimensions I propose below are not original inventions. Appendix C indicates some of my sources. Previous sets have always broken at least some of the rules above. They have at times been so general as to go far beyond settlement form and to require a complex (and usually impossible) calculation which involves culture, political economy, and many other nonformal features. Or they refer to some particular physical solution that is appropriate only in a particular situation. They may mix spatial and nonspatial features, or mix levels of generality, or mix the scale of application. Frequently, they are bound to a single culture. They do not include all the features of city form which are important to human values. They are often given as absolute standards, or they call for minimizing or maximizing, instead of being dimensions. The qualities are sometimes not measurable, or even identifiable, in any clear way. They frequently overlap each other.

The list that follows is an attempt to rework and reorder the material in a way that escapes those difficulties. The presumed generality of this list lies in certain regularities: the physical nature of the universe, the constants of human biology and culture, and some features which commonly appear in contemporary large-scale settlements, including the processes by which they are maintained and changed.

Concept of Ecology

But some view of the nature of human settlements, however unclear or general, is necessarily assumed in making any list. Unfortunately, it is much easier to say what a city is not: not a crystal, not an organism, not a complex machine, not even an intricate network of communications—like a computer or a nervous system—which can learn by reorganizing its own patterns of response, but whose primitive elements are forever the same. True, somewhat like the latter, the city is interconnected to an important degree by signals, rather than by place-order or mechanical linkages or organic cohesion. It is indeed something changing and developing, rather than an eternal form, or a mechanical repetition which in time wears out, or even a permanent recurrent cycling which feeds on the degradation of energy, which is the concept of ecology.

Yet the idea of ecology seems close to an explanation, since an ecosystem is a set of organisms in a habitat, where each organism is in some relation to others of its own kind, as well as to other species and the inorganic setting. This system of relations can be considered as a whole, and has certain characteristic features of fluctuation and development, of species diversity, of intercommunication, of the cycling of nutrients, and the passthrough of energy. The concept deals with very complex systems, with change, with organic and inorganic elements together, and with a profusion of actors and of forms.

Moreover, an ecosystem seems to be close to what a settlement is. Complicated things must in the end be understood in their own terms. An image will fail to stick if it is only a borrowing from some other area, although metaphorical borrowings are essential first steps in understanding.

Apt as it is, the concept of ecology has its drawbacks, for our purpose. Ecological systems are made up of "unthinking" organisms, not conscious of their fatal involvement in the system and its consequences, unable to modify it in any fundamental way. The ecosystem, if undisturbed, moves to its stable climax of maturity, where the diversity of species and the efficiency of the use of energy passing through are both at the maximum, given the fixed limits of the inorganic setting. Nutrients recycle but may gradually be lost to sinks, while energy inevitably escapes the system or becomes unavailable. Nothing is learned; no progressive developments ensue. The inner experiences of the organisms—their purposes and images—are irrelevant; only their outward behavior matters.

A "Learning Ecology"

An evolving "learning ecology" might be a more appropriate concept for the human settlement, some of whose actors, at least, are conscious, and capable of modifying themselves and thus of changing the rules of the game. The dominant animal consciously restructures materials and switches the paths of energy flow. To the familiar ecosystem characteristics of diversity, interdependence, context, history, feedback, dynamic stability, and cyclic processing, we must add such features as values, culture, consciousness, progressive (or regressive) change, invention, the ability to learn, and the connection of inner experience and outer action. Images, values, and the creation and flow of information play an important role. Leaps, revolutions, and catastrophes can happen, new paths can be taken. Human learning and culture have destabilized the system, and perhaps, some day, other species will join the uncertainty game. The system does not inevitably move toward some fixed climax state, nor toward maximum entropy. A settlement is a valued arrangement, consciously changed and stabilized. Its elements are connected through an immense and intricate network, which can be understood only as a series of overlapping local systems, never rigidly or instantaneously linked, and yet part of a fabric without edges. Each part has a history and a context, and that history and context shift as we move from part to part. In a peculiar way, each part contains information about its local context, and thus, by extension, about the whole.

Values are implicit in that viewpoint, of course. The good city is one in which the continuity of this complex ecology is maintained while progressive change is permitted. The fundamental good is the continuous development of the individual or the small group and their culture: a process of becoming more complex, more richly connected, more competent, acquiring and realizing new powers—intellectual, emotional, social, and physical. If human life is a continued state of becoming, then its continuity is founded on growth and development (and its development on continuity: the statement is circular). If development is a process of becoming more competent and more richly connected, then an increasing sense of connection to one's environment in space and in time is one aspect of growth. So that settlement is good which enhances the continuity of a culture and the survival of its people, increases a sense of connection in time and space, and permits or spurs individual growth: development, within continuity, via openness and connection.*

Values Implied

These values could, of course, be applied to judging a culture as well as a place. In either case, there is an inherent tension as well as a circularity between continuity and development—between the stabilities and connections needed for coherence and the ability to change and grow. Those cultures whose organizing ideas and institutions deal successfully with that tension and circularity are presumably more desirable, in this view. Similarly, a good settlement is also an *open* one: accessible, decentralized, diverse, adaptable, and tolerant to experiment. This emphasis on dynamic openness is distinct from the insistence of environmentalists (and most utopians) on recurrence and stability. The blue ribbon goes to development, as long as it keeps within the constraints of continuity in time and space. Since an unstable ecology risks disaster as well as enrichment, flexibility is important, and also the ability to learn and adapt rapidly. Conflict, stress, and uncertainty are not excluded, nor are those very human emotions of hate and fear, which accompany stress. But love and caring would certainly be there.

Any new model of the city must integrate statements of value with statements of objective relationships. The model I have sketched is neither a developed nor an explicit one, and I retreat to my more narrow concern with normative theory. But the surviving reader will see that these general preferences—for continuity, connection, and openness—underlie all the succeeding pages, even while the theory makes an effort to see that it is applicable to any context.

* The bias of the teacher is now unmasked.

Five Performance Dimensions

Given that general view and the task of constructing a limited set of performance dimensions for the spatial form of cities, I suggest the following ones.† None of them are single dimensions; all refer to a cluster of qualities. Yet each cluster has a common basis and may be measured in some common way. I simply name the dimensions at this point. Subsequent chapters will discuss each dimension in detail.

There are five basic dimensions:

1. *Vitality*: the degree to which the form of the settlement supports the vital functions, the biological requirements and capabilities of human beings—above all, how it protects the survival of the species. This is an anthropocentric criterion, although we may some day consider the way in which the environment supports the life of other species, even where that does not contribute to our own survival.
2. *Sense*: the degree to which the settlement can be clearly perceived and mentally differentiated and structured in time and space by its residents and the degree to which that mental structure connects with their values and concepts—the match between environment, our sensory and mental capabilities, and our cultural constructs.
3. *Fit*: the degree to which the form and capacity of spaces, channels, and equipment in a settlement match the pattern and quantity of actions that people customarily engage in, or want to engage in—that is, the adequacy of the behavior settings, including their adaptability to future action.
4. *Access*: the ability to reach other persons, activities, resources, services, information, or places, including the quantity and diversity of the elements which can be reached.
5. *Control*: the degree to which the use and access to spaces and activities, and their creation, repair, modification, and management are controlled by those who use, work, or reside in them.

   If these five dimensions comprise all the principal dimensions of settlement quality, I must of course add two meta-criteria, which are always appended to any list of good things:
6. *Efficiency*: the cost, in terms of other valued things, of creating and maintaining the settlement, for any given level of attainment of the environmental dimensions listed above.
7. *Justice*: the way in which environmental benefits and costs are distributed among persons, according to some particular principle such as equity, need intrinsic worth, ability to pay, effort expended, potential contribution, or power. Justice is the criterion which balances the gains among persons, while efficiency balances the gains among different values.

Meta-Criteria

These meta-criteria are distinct from the five criteria that precede them. First, they are meaningless until costs and benefits have been defined by specifying the prior basic values. Second, the two meta-criteria are involved in each one of the basic dimensions, and thus they are by no means independent of them. They are repetitive subdimensions of each of the five. In each case, one asks: (1) What is the cost (in terms of anything else we choose to value) of achieving this degree of vitality, sense, fit, access, or control? and (2) Who is getting how much of it?

I propose that these five dimensions and two meta-criteria are the inclusive measures of settlement quality. Groups and persons will value different aspects of them and assign different priorities to

† At the end of appendix C, the curious reader will find some of the excess baggage which I discarded while developing these magic five.

them. But, having measured them, a particular group in a real situation would be able to judge the relative goodness of their place, and would have the clues necessary to improve or maintain that goodness. All five can be defined, identified, and applied to some degree, and this application can be improved.

Questions

Now, is this really so? Do the dimensions really meet all the criteria which were given at the beginning of this section? Do they in fact illuminate the "goodness" of a city, or are they only a verbal checklist? Can locations on these dimensions be identified and measured in a concrete way? Are they useful guidelines for research? Do they apply to varied cultures and in varied situations? Can general propositions be made about how optima vary according to variations in resource, power, or values? Can degrees of achievement on these dimensions be related to particular spatial patterns, so that the benefits of proposed solutions can be predicted? Do our preferences about places indeed vary significantly as performance changes? All that remains to be seen.

First, it is necessary to elaborate on each dimension, in order to expand its various subdimensions and to explain its probable connections to particular forms and more general values. In doing so, we can review what evidence there is and indicate some gaps in our knowledge. However, it will shortly be obvious how much of this evidence is speculative.

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What Should an Ideal City Look Like from an Ecological View? – Ecological Demands on the Future City

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Abstract The "ecological" city is a goal which will never be completely reached. Nevertheless politics and planning are obliged to approach it as closely as scientific knowledge will allow. Science now has sufficient ecological knowledge to make recommendations both positive and negative. However, there is an urgent need for long term research including not only the planning of ecological measurements but also examining the efficiency of the results.

Keywords: urban planning · ecology

Introduction

On November 18/19, 1994 at the invitation by Prof. Dr. Lothar Finke, the second session of the "City Ecology Working Party" of the Ecological Society took place amongst the Regional Planning Faculty of the University Dortmund. On the agenda for discussion was the question "What should an ideal city look like from an ecological viewpoint?" During that discussion, "Ecological demands on the future city" were especially intensively discussed. The participants agreed that the results of the discussion should be made available to a broader public and entrusted the present authors with that task. This paper presents, in an informal way, those discussions and the conclusions from them.

Problems Associated with the Term "Ideal City"

Cities "consume" natural resources, which puts loads upon their environment (e.g. by drinking water withdrawal, extraction of raw materials, food production, deposition of wastes, emission of air pollutants, wastewater production). Nearly none of the principles characteristic of natural ecosystems are present in cities: these missing characteristics include ecological stability or elasticity (self-regulation ability), internal substance cycles, and independence from other ecosystems (i.e., local primary producers as basis of the food or energy pyramid) (Haber 1992). That is why we concluded that the question was wrongly stated from the very beginning, since an "ecologically ideal city" cannot exist. What is needed to begin the discussion is a definition of "ideal": the usual meaning

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includes a highly desirable “perfect” condition, often to be reached only with difficulties and possibly only partly achievable. Considered in this way it is worth while thinking about the ”ecologically ideal city”, especially if one stays aware that the result is a TARGET, and though striven for is unlikely to be reached completely – at best, only approximated.

Are Specific Proposals Possible?

Each city is an individual, due to differences e.g. in the historical development, its position in the natural space, position relative to other cities (isolated: position in the agglomeration area), total area, number of inhabitants, city structure, economy, industrial structure, traffic). That is why it is difficult to develop generalizable, specific proposals about “what might constitute an ecologically ideal city”. However, ecology as science seems required to make a statement. Perhaps a good starting point might be urban development, seeking to indicate how cities might be developed in “ecologically compatible” ways. It is also important to discuss the important concept of the “sustainability” of urban-industrial development (“sustainable development”: Brundtland report 1987; Alleux 1993; Haber 1994).

Recommendations (examples, principles) for ecologically compatible or sustainable urban development have been given by various authors (e.g. Deutscher Rat für Landespflege 1992; Haber 1992; Finke 1993; Schmidt 1993; Sukopp & Wittig 1993; Boecker 1994). Some attempts are being made to gradually introduce such ideas into political and planning approaches so as to improve urban environment and the quality of life in cities (e.g. Federal Minister for Regional Planning, Construction and Urban Construction 1994). By studying the efforts made in representative cities, one might develop some recommendations as to specific requirements and how best to approach them.

Demands on Urban Development from an Ecological View

The city is now the essential habitat of mankind. Already today about half of the world population lives in cities, and in 2025 it will be more than two thirds (Sadik 1992). At present, over 83% of the population GROWTH is occurring in cities (United Nations 1990). The “ecological city” has to be adjusted to the necessities of life of its main inhabitants, i.e. people. People can survive only in a sane (that is, not dysfunctional) environment. In addition, it must be the aim of any ”ecologically more compatible” urban development to handle the natural resources (soil, water, air, flora and fauna) in cities as well as in their environment as carefully as possible in the sense of a ”sustained” development.

As cities grow in population, the also expand greatly in area. Satellite photography show us that urban conglomerations go right across the entire European continent, e. g. along the Rhine-river and its tributaries, beginning at Rotterdam and reaching up to Basel and Stuttgart: for this feature, the term “Rhineland megalopolis” is used (Birkenhauer 1989). Today urban uses and the infrastructural networks between cities (traffic, supply, waste disposal) might be regarded as essential elements which still “colonize” landscape areas outside the city itself. Cities must be planned and built to minimize both hygienic-toxicological problems and economic risks that result from cities’ growth out into the surrounding landscapes. The ecological city must be designed to:

- not harm the physical and psychic health of man but to promote it;
- not impact its environment or destroy it;
- allow maintenance of “nature” in typical city locations.

Guidelines for Implementing Ecological Knowledge in Urban Planning and Development

A gradual approach to the ecologically ideal city may be possible if one were to follow, in the framework of urban planning, the following principles [see Sukopp & Wittig 1993]:

- reduction of energy input;
- avoiding or cyclization of flows of various substances;
- protection of all media of life;
- preservation and promotion of nature;
- providing lost of small-scale structure and rich differentiation in the environment.

One might also add as another principle the preservation and promotion of historical city elements. These principles supplement each other and overlap in such a way that, as a rule, a few of them do get addressed in planning targets. But that is mostly haphazard, since for each principle to be implemented (individually) requires a whole package of specific measures.

Reduction of Energy Input

Numerous environmental problems in a city, and problems caused by cities in their larger environment, come from the huge growth in energy use (mostly from non-renewable carbon sources), and production of waste gas, soot, ash and waste heat resulting from it. That is why the reduction of energy use will essentially contribute to reducing the environmental impacts and thus to improving the ecological conditions of cities and their environment. The following measures can contribute to reducing the energy use:

- rational energy conversion in the general development planning (e.g. Roth et al. 1984);
- decreasing use and demand for energy in the fields of production, room heating, air conditioning, and traffic;
- reducing the percentage of energy consumed for heating and energy transport;
- avoiding any unnecessary energy use (e.g. preferring local public transport to individual vehicles other than bicycles);
- extension of bicycle and foot paths (preferably separated from motor traffic);
- shift of the industrial transport to rails, avoidance of empty running;
- avoiding motor vehicle traffic congestion by decentralizing and mixing of uses (all important utilities should exist in each section of a city);
- shortening the distances between production and consumers.

Avoidance of Cyclical Substance Flows

In natural ecosystems, substance flows for the most part occur largely WITHIN the system and tend to be strongly cyclical – that is, most materials are actually recycled more or less in-situ within a system. In cities, however, systems depend (presently and for the immediate future) almost completely on the continual flow-through, or input+output, of critical substances, with little in-situ recycling (Mueller 1992). This results in high stresses inside a city and also in its environment. It contradicts the principle of reducing energy input, since all that transportation of materials consumes energy. This is why cities must avoid "throughput" processes and, wherever possible, seek cyclical substance flows whenever possible (ultimately all: e.g., recycling of building material: Andrae et al. 1994; Rentz et al. 1994). Again, short distances should be established between production and consumption, so as to minimize energy use in transporting materials. As measures to avoid substance "throughput" flows one can imagine such approaches as:

- using less packaging material;
- preferring of regional products by retailers and buyers;
- replacement of company-specific returnable packs by standardized, product-specific packs without company labelling;
- saving of energy (less transport of fuel);
- replacement of fuel-dependent energy by solar energy (this should also have positive effects on urban climate);
- development and use of durable, reusable building and packaging materials degradable after use without producing toxic and/or space requiring residues (additional effect: reduction of the demand for dumping grounds);
- decentralized composting of organic wastes;
- development of a comprehensive water management (rain water as industrial water, closed cooling and industrial water cycles in production enterprises and washing facilities for clothing and vehicles etc., improving groundwater recharge);
- limiting new substance and energy flows in local urban functions.

Principle of Protecting all Media of Life

The protection of all media of life (air, soil, surface water, groundwater) serves to improve the living conditions of the inhabitants of cities while also providing, at the same time, the prerequisites to more nature in a city. In the narrower sense, protection involves:

- supervising measures (installation or extension of monitoring networks);
- preventive measures (e.g. extension of the sewage system, separating systems, avoiding output of toxic substances);
- rehabilitation measures (implementation of principles 1&2).

In the wider sense protection of the media of life involves also the possibility of their regeneration, e.g. promotion of groundwater recharge and keeping open or installing fresh air pathways (by selective construction) that will help ventilate cities.

Preservation and Promotion of Nature and Urban Spaces

The preservation and promotion of nature in cities involves the following measures (see Auhagen & Sukopp 1983):

- creation of priority areas for environmental and nature protection;
- setting of nature and landscape conservation priorities differentiated by zones;
- promotion of the development of spontaneous nature in the inner city;
- consideration of historical continuity;
- preservation of large connected spaces, especially in the inner city;
- cross linkage of spaces (Kleyer 1994);
- preservation of location differences;
- preservation of differentiated intensities of use;
- preservation of the variety of typical elements of an urban landscape;
- prevention of all avoidable interventions in nature and landscape;
- functional integration of structures into ecosystems;
- maintenance of green spaces.

All these measures presuppose that actual biotope stocks and condition are known. Biotope mapping and its updating are therefore indispensable (see Working group "Methods of biotope mapping in the settled areas" 1993; Schulte 1993).

Principle of Small-Scale Structuring and Rich Differentiation

Having lots of small-scale structure and rich differentiation (e.g., lots of different microhabitats, lots of relatively fine-scale spatial fragmentation of human activities) serves to preserve and promote species richness within "in-city nature" . Another benefit is that small-scale structuring and differentiation of HUMAN activities helps reduce energy use – for instance, if each city section contains most of the functions needed by the population, then people won't have to travel so much to get what they need, and will thus use less energy – most necessities can be reached by foot (Breuste 1994). Eventually the individual and unmistakable design of various city sectors is involved (i.e. preservation of built structures). The more individuality that individual "quarters" or section have, the easier is the identification of their inhabitants with "their" quarter (development of the love of one's local area). Importantly, that increased identification increases peoples' sense of responsibility for their sector, thus strengthening their readiness to behave in an ecologically conscious way. Moreover, if one cannot generate in the population a desire to behave "ecologically soundly", then even the best urban planning will not succeed.

Needs for Research and Testing

Urban Ecology is a comparative young scientific discipline. That is why there is a high demand for research and testing not only with regard to the ecological conditions existing in cities but also with regard to the practicability of using ecological knowledge in urban planning (plus studies of how well that use actually works in practice). That is why it is urgent to implement model projects in various types of cities and regions – to a certain extent, as an "initial idea" of an urban planning oriented much more strongly to ecological principles.

Some specific types of projects are needed, for example:

Investigations about the required data bases – "What do we need to know?" – Although in most of cities there are extensive data bases of various sorts, most are not of importance to ecologically oriented urban planning, and most cannot be used effectively in the framework of such a planning. Therefore, one clear need is for model projects to establish a framework stating which data are indispensable, and in what forms it should be available.

Cross-linked planning – To date, most urban planning has been done only at the level of "sections" or sectors of a city. When developing programs for the future of cities, we must deal with the solution of problems in an interdisciplinary way, with a long-term view, and with an overview of the city as a whole, not piecemeal. This means, first of all, to carry out an integrated (cross-linked) planning with all ecologically oriented disciplines participating. To implement ecological targets for urban development it will also be indispensable to consider well the expectations and demands of both inhabitants and planners (Boecker 1994) . This kind of interdisciplinary planning must be TAUGHT as a field of study – it will not happen accidentally – and its practicability must be shown through carefully designed model projects.

Illustration of ecological-economic target conflicts – We need, in ecological projects about cities, interdisciplinary studies that will help clear up conflicts between ecological and economic goals. They are required so that we can understand why, repeatedly, urban ecological problems are over-ridden by "practical economic necessities". Perhaps we could develop approaches that would show how economic instruments could help and speed up dealing with ecological issues (example: and “environmental effects” tax on impervious surfaces).

Implementation of model projects – There is no point to integrated planning if we cannot implement the planning results. In general, experience so far shows that ecologically oriented large-area urban planning may be implemented only in parts, if at all. That is why it is of special importance that the “feasibility” of ecological urban planning concepts be proven by means of model projects: successful demonstration projects that are based on or incorporate ecological principles would greatly encourage and simplify next-generation projects based on the demonstrated results.

Support by science in the implementation phase – To achieve optimum implementation of ecological targets in city planning, the scientists participating in drafting the plan MUST continue on to support its implementation. Unfortunately, there is a great lack of implementing models into actual practice – scientists tend to work in an academic vacuum removed from real-world applications. That is a critical problem that needs to be fixed.

Measuring results of planning efforts – We need to objectively measure the success, failure, and efficiency of planning efforts (questions to be answered include: ”Were the above-mentioned demands and principles fulfilled; which ecological improvements were specifically reached; what is the relation between success or effects of a measure and the volume of the means, e.g. funds, used”). In the event of failures, we need to identify causes and be able to suggest improvements. Only by such an analysis of actual results, involving goals, techniques and results, can we be sure that the guidelines for ecologically-oriented urban development can actually be implemented in an optimum way.

The above needs can be satisfied only through long-term projects. Depending on the pre-existing available data bases, we should think of a preparatory phase of 2–3 years: it will involve sorting of data, and perhaps careful collection of selected new data; then a planning phase of 1–2 more years, then an implementation phase of 2–5 years, depending on the size of the planning area and the scope of the project. After further 5–10 years scientific investigations should be made of the results.

Summary

The “ecological” city must neither endanger human health nor destroy or negatively influence its surroundings. It must also allow the development and/or maintenance of “nature” even in its centre. Although these requirements cannot be perfectly realized, politicians and planners must be made to TRY to approach this as a goal. Our level of scientific knowledge is clearly good enough, at both theory and practical levels, to provide specific recommendations (both positive and negative) for “ecologically-minded” urban planning efforts, and there are successful examples. However, there is a still an urgent need for long-term research and test projects that use project orientated data sampling, analysis of problems, identifying conflicts and offering solutions, and team-orientated planning. We then need a well-designed application phase, done under scientific observation so that we can DETECT and quickly CORRECT errors.

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Land Use Planning and Wildlife Maintenance

Guidelines for Conserving Wildlife in an Urban Landscape

Michael E. Soulé

Abstract The study of plants and animals on islands, both natural and artificial, has produced a body of generalizations immediately useful to land use planners concerned with minimizing the impacts of habitat destruction on the environment. A case study of 37 isolated chaparral fragments in San Diego, California, demonstrates the consequences of habitat fragmentation, including rapid and predictable extinctions of native birds in isolated canyons. This study and others can be used to generate planning guidelines for the prevention of such disappearances. Among the most important measures that can be taken are consolidation of open space set-asides and the provision of corridors linking habitat patches. Corridors can mitigate some of the negative effects of development on wildlife, especially where they facilitate the movement of large predators.

Keywords: San Diego · bird community · conservation biology · coyote · urban planning · ecology · island biogeography

The public concern about environmental issues will continue to increase as the planetary environment deteriorates under the weight of a rapidly growing human population and accelerating discharges of toxic chemicals, solid and organic wastes, greenhouse gases, and other by-products of human activities. Since the publication of *Design with Nature* (McHarg 1971), an environmental perspective has gained prominence in land use planning. This interest is exemplified by the attention given to physical factors, such as soil hydrology (Dearden 1980; Dunne and Leopold 1978), geologic hazards (Griggs and Gilchrist 1983), and visual amenities (Elsner and Smadon 1979), and by the integration of planning and landscape architecture (McBride 1977).

Currently, many environmentalists and the public at large are asking that planners give more attention to the impact of development on native animal species (wildlife values). For example, there is growing concern among environmentalists that laws such as the Endangered Species Act, though they provide for the short-term needs of certain critically threatened, “flagship” species, do not address the fundamental issues of the deterioration of entire ecosystems or regions. The worrisome if slow decline of songbirds and amphibians, and the steady disappearance of wetlands in the United States (Terborgh 1989; McKibben 1989) exemplify this gradual environmental deterioration. Surveys (Kellert 1980) have shown that most city people appreciate natural amenities, including native wildlife, and that citizens are willing to pay for a more authentic environment.

Just as the 1970s was the decade when land use planning and landscape architecture were integrated, the 1990s might be the decade when planners recognize the relevance of conservation biology, landscape ecology, and restoration ecology. An integration of principles and guidelines from these

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modern biological disciplines would provide planners with additional tools to deal with the effects of development on biological diversity in general, and the viability of native species in particular.

The principles of modern island biogeography, one of the core disciplines of conservation biology and landscape ecology, can provide useful guidelines for planners wishing to assist communities in maintaining a rich environmental mosaic that complements other components of human welfare. To demonstrate this point, this article opens with an overview of conservation biology, followed by a case study from San Diego showing how the results of such research are relevant to the issue of cumulative impacts of development[1] on environmental quality. Guidelines that might promote the maintenance of wildlife in the suburban situation are then suggested, and the limits of extrapolation from the San Diego system to those in other regions are explored.

Island Biogeography and Conservation Biology

We live in a world in which natural habitat is increasingly confined to isolated patches. For some time it has been observed that isolation increases the risk of extinction, and in the last quarter-century the rules governing species extinction in isolated patches of habitat have been clarified by practitioners of the scientific discipline known as island biogeography (MacArthur and Wilson 1967). Though some controversies linger, there is sufficient agreement (Soulé and Simberloff 1986) among these practitioners to warrant a system of guidelines for land use planners. Similar guidelines have been discussed for over two decades in the literatures of applied island biogeography and conservation biology (Diamond 1975).

Island biogeography is one of the cornerstones of conservation biology (Soulé and Wilcox 1980; Soulé 1986; the journals *Conservation Biology, Biological Conservation*, and *Biogeography*), a field dedicated to the application of science to the protection of genetic resources, species diversity (the prevention of extinction), and ecological diversity (the maintenance of ecosystem processes and habitat diversity). Island biogeography overlaps considerably with landscape ecology (Turner 1989); both areas are concerned, in part, with the loss of species from habitat fragments, and with the disappearance of wildlife in the vicinity of human settlements.

One of the established principles of island biogeography is that the rate of species extinction in an isolated patch of habitat is inversely related to its size (MacArthur and Wilson 1967); this is one aspect of a more general phenomenon known as the *area effect*, a term referring to the deleterious effects on biotic systems of decreasing patch size, per se. Even quite large habitat islands have observable rates of extinction. For instance, it is now recognized that most national parks in the western United States are too small to prevent the extinction of many medium-sized and large mammals (Newmark 1987). On a local scale, isolated patches of habitat the size of most open space "set-asides" are often much too small to prevent catastrophic rates of habitat disturbance and the loss of many species of animals (as described below). Unfortunately, by the time the disappearance of wildlife is noticed by the human residents in a new subdivision, it is too late to do anything about it.

Edge effects are also associated with habitat fragmentation. Because the ratio of edge habitat to interior habitat increases as fragment size decreases (Fig. 1), it is important to understand how edges affect wildlife. Edges (or ecotones, as habitat interfaces are called in wildlife biology) occur where a habitat, such as a forest, meets a road, a clear-cut, or some other element, natural or artificial. Edges benefit certain species, such as deer. But most conservationists believe that edges, overall, are detrimental to the maintenance of species diversity (see Conservation Biology 1988). Among some of the

[1] The term "development" usually describes a two-step process: (1) the destruction of natural systems or habitat; and (2) the replacement of natural systems by artificial ones that increase the welfare or wealth of some humans. It is more appropriate to refer to the first step as "denaturation" (Soulé 1990). Denaturation, if sufficiently extensive, not only reduces the amount of natural habitat, but also causes the fragmentation of the habitat that remains.

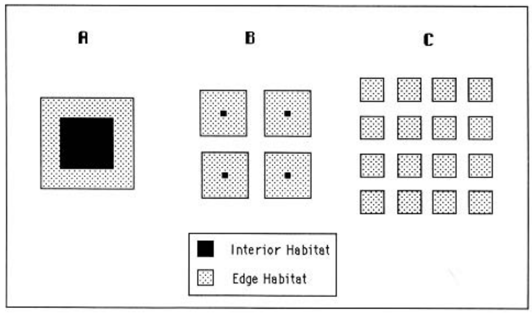

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Fig. 1 Diagram of the relationship between edge effects and the amount of interior (unaffected) habitat as a function of the area of a habitat fragment. Note that the edge effects penetrate a constant distance, regardless of the size of the fragment. A represents a large fragment; B, four fragments that together equal the area of the A fragment; and C, 16 small fragments that together equal the area of the A fragment

major categories of deleterious edge effects are (1) higher frequency and increased severity of fire, (2) higher rates of hunting and poaching, (3) higher intensities of predation, (4) higher probability of nest parasitism on bird nests by brown-headed cowbirds, and (5) higher intensities of browsing and other forms of disturbance that favor weedy species.

As habitat destruction spreads and the distance between remnant patches increases, animals find it more difficult to disperse between patches. The relation between isolation and movement frequency is inverse, and is known as the *distance effect*. A corollary of this principle is that endangered populations in isolated patches are more likely to be "rescued" by dispersing individuals from other patches if the patches are close together (Brown and Kodric-Brown 1977). Dispersal of individuals between patches can help protect against demographic "accidents," such as an episode of unusually high mortality. Immigrants can also "rescue" a population that is in jeopardy because of inbreeding or an unbalanced sex ratio. Generally, therefore, compact archipelagos comprising islands that are close together have more species per island than do archipelagoes comprised of remote islands. This is because proximity facilitates both the rescue of endangered populations and the recolonization of habitat islands where local extinctions have occurred.

Another relevant generalization—from the discipline of community ecology rather than island biogeography—is that large predators help to maintain the diversity of species within an ecosystem because they suppress the numbers of destructive smaller predators, the depredations of which can be disastrous for species such as ground-nesting birds. A common myth is that large predators (such as wolves, coyotes, and cougars) are bad for wildlife. But this is true only if one uses a very restricted definition of wildlife, and if one means by "bad" that there are, say, fewer deer where predators are abundant. In most parts of the United States, it is deer, not predators, that damage natural and artificial ecosystems.

None of the above problems occurs instantaneously; as fragmentation increases, the area of individual patches gradually decreases, the distance between patches increases, and edge effects creep inward. It is expected, therefore, that extinctions of species within isolates will be cumulative. In those rare situations where the ages of the isolates are known, one might expect to detect such an *age effect*; namely, the older the isolated patch, the more altered it should be, and the fewer species it should contain.

A San Diego Case Study: The Fate of Birds in Chaparral Fragments

The consequences of fragmentation have been studied in deciduous forests in the eastern United States (see Wilcove et al. 1986 for a review), in the tropics (Terborgh and Winter 1980; Lovejoy et al.

1986), and elsewhere, but there have been few systematic analyses of fragmentation in the western United States (Newmark 1987). The case described below is an analysis of fragmentation in sage scrub and chaparral habitats in coastal Southern California. This example focuses on a particular group of bird species living in remnant habitat islands left after denaturation and development in San Diego County. In general, the results of this study are typical of those in forest habitats, except that the relative immobility of many of the birds in chaparral may lead to a higher rate of extinction than would be expected on the basis of results from temperate forests. Chaparral[2] is a form of dense scrub vegetation. Among botanists, chaparral is celebrated for its extraordinary diversity of plant species (Raven and Axlerod 1978). Among fire fighters and planners, it is often vilified for its flammability, especially during the rainless summer and fall typical of Mediterranean climates. Even though the dominant shrubs in coastal chaparral are rarely more than three meters high, and often less than one or two, this habitat supports a very rich fauna, including mountain lions, bobcats, coyote, deer, diverse birds, reptiles, and insects.

Only a fraction of coastal scrub vegetation remains in Southern California (Westman 1987; Jensen 1990), and most of the remnants of chaparral habitat in the coastal section of San Diego County are limited to steepsided canyons that dissect the coastal mesas. Until recently, these interconnecting canyons constituted a network of natural open space. They also served as neighborhood boundaries. Historically, people, especially children, have used the canyons for the same purposes that people everywhere use open space, namely visual relief, exercise, walking dogs, and other forms of spontaneous recreation and play. Recently, the coastal canyons have been serving another function—shelter for the homeless. Other socioeconomic conditions and technological innovations, including escalating land values, the perceived need for a dense system of freeways, and the availability of efficient earth-moving machinery, have led to the denaturation of most canyon habitat and thus to the physical isolation of the remaining fragments of chaparral. This case study, therefore, addresses a common dilemma in land use—the conflicts arising from pressures for short-term economic gain, on the one hand, and for long-term environmental quality, on the other.

A Summary of Methods and Results

The San Diego study (Soulé et al. 1988; Bolger et al. 1991) focused on species of birds that require natural scrub habitat for breeding and shelter. These were the black-tailed gnatcatcher, roadrunner, California quail, California thrasher, rufous-sided towhee, Bewick's wren, and wrentit. Censuses to determine the presence/absence of these chaparral-requiring bird species (CR birds) were conducted in 37 isolated canyons (Fig. 2). The biogeographic variables that are typically considered in such research (habitat area, isolation, island age) were used, and simple, partial, and multiple regressions were performed to determine the possible influence of these variables on the persistence of the CR bird species in fragmented habitat. Only the results relevant to planning are discussed here.

The variables in this study included the sizes of canyons (AREA), the total area of natural chaparral cover in the canyons (CHAP), the "ages" (time elapsed since they became isolated from adjacent chaparral habitat by denaturation and development—AGE) of canyons (Table 1), various measures of disturbance, and several variables estimating the degree of isolation of canyons from each other and from the closest unfragmented habitat. Much of the information was obtained from aerial photographs, subdivision maps, and city planning maps and records. Besides using these standard variables and sources, we included variables (such as FOXCOY) that represent the distribution of potential predators (see Table 1), and we tested for interactions. We also took a census of birds in unfragmented, "mainland" habitat (Bolger et al. 1991) in nearby, relatively undenatured,

[2] The term chaparral, as used here, includes coastal scrub plant associations.

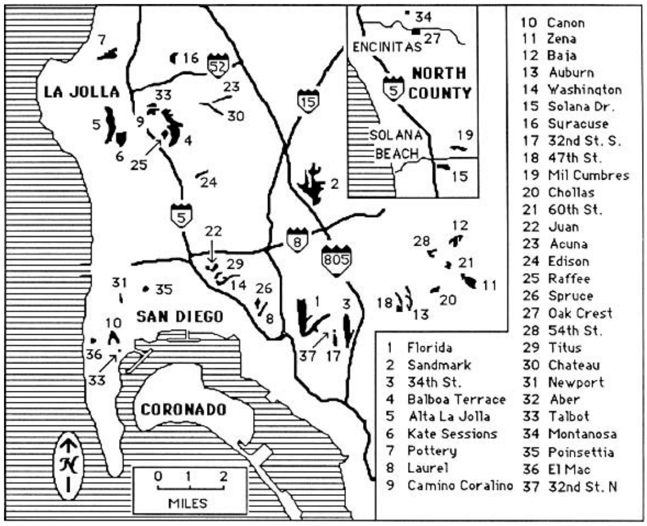

Fig. 2 Location of the study sites (canyon fragments) in the vicinity of San Diego, California. Site 37 was considered a satellite of site 17, and was not included in the analyses described here

areas in southern California, including Camp Pendleton and Tecolote Canyon. The following points summarize the most relevant results (nonsignificant effects are not discussed here):

1. Most canyons lose at least half of their CR birds within 20 to 40 years after isolation, though the larger canyons retain from two to six species (Fig. 3). For canyons less than 50 hectares (about 123 acres), the average number of surviving CR species after 40 years is 0.5. The attrition of habitat due to mechanical disturbance, fire (Westman et al. 1981), and invasion by exotics (MacDonald et al. 1988) must account for some of this loss of bird species. A statistically significant proportion of these local (within canyon) extirpations, however, is independent of the amount of chaparral cover (as shown by partial correlation analysis), and can be attributed to the number of years since the canyon was isolated from a larger tract. Soulé et al. (1988) refer to this temporal component of the extinction process as the "age effect."

   It is likely that the underlying cause of this age effect is the small population sizes of most species in the isolated canyon fragments. Small populations are chronically vulnerable. Theoretical studies (e.g., MacArthur and Wilson 1967; Goodman 1987) and modeling results (Shaffer 1983) have shown that the probability of extinction of small isolated populations increases exponentially below a population size of 75 because of the randomness inherent in demographic (birth and death) processes. Unmanaged populations under 10 or 20 individuals cannot normally be expected to persist for more than a few generations. Empirical studies also establish that population size is the best predictor of local extirpation (Terborgh and Winter 1980; Soulé et al. 1988; Pimm et al. 1988).
2. As shown in Fig. 4, there is also an area effect. That is, the number of CR birds persisting in canyons is correlated with the area of undisturbed, natural habitat (CHAP) in the canyons. This effect persists after removing, statistically, the age effect. Our interpretation of this area effect is that the amount of chaparral habitat that actually exists in a fragment at some point in time limits the number of species that can live in that patch at that time. This result is typical in that an area effect is the statistically strongest interaction in most island biogeographic studies.
3. A third, statistically independant factor, FOXCOY, remained after removing (by partial and multiple correlation and regression) the age and area effects. Canyons frequented by coyotes and lacking grey foxes retain more species of CR birds than canyons without coyotes but inhabited

Table 1 Biogeographic data used in the regression analyses

	CANYONS	SPECIES[a]	AREA (hectares)[b]	CHAP (hectares)[b]	AGE (years)[c]	FOXCOY[d]
1	Florida	6	102.77	67.83	50	1
2	Sandmark	6	84.05	75.65	20	1
3	34th St.	6	53.76	40.32	34	1
4	Balboa T.	5	51.77	38.82	34	1
5	Alta L.J.	6	33.14	16.57	14	1
6	Kate Ses.	6	25.56	15.33	16	1
7	Pottery	5	17.92	10.75	14	1
8	Laurel	0	9.72	.49	79	1
9	Cam. Cor.	4	9.08	8.62	20	3
10	Canon	0	8.66	1.73	58	1
11	Zena	3	8.51	2.55	36	1
12	Baja	3	8.4	4.37	31	1
13	Auburn	2	8.37	2.51	32	1
14	Washington	2	8.07	1.31	74	1
15	Solana Dr.	7	7.64	6.87	11	3
16	Syracuse	5	7.51	6.38	18	1
17	32th St. S.	1	6.36	.95	56	1
18	47th St.	1	6.31	2.52	32	1
19	Mil Cumbres	6	6.23	5.61	11	3
20	Chollas	1	6.22	1.56	36	1
21	60th St.	2	6.11	2.14	37	1
22	Juan St.	2	5.97	2.99	23	1
23	Acuna	3	5.08	1.52	22	2
24	Edison	5	4.75	4.28	8	1
25	Raffee	3	4.74	2.37	19	1
26	Spruce	0	4.28	.43	86	1
27	Oak Crest	6	3.88	1.94	6	3
28	54th St.	2	3.61	1.81	20	1
29	Titus	0	3.5	.25	77	1
30	Chateau	3	3.27	1.80	20	2
31	Newport	1	2.14	1.60	60	2
32	Aber	2	1.6	1.04	15	1
33	Talbot	0	1.41	1.27	55	1
34	Montanosa	5	1.32	1.25	2	3
35	Poinsettia	0	1.2	.30	50	2
36	El Mac	0	1.1	.66	32	1
37	32nd St. N.	1	.4	.10	77	1

a. SPECIES is the number of chaparral-requiring bird species
b. AREA and CHAP are defined in the text.
c. AGE is the years since isolation of the habitat fragment.
d. Under FOXCOY, 1 = coyotes absent, foxes present, 2 = coyotes absent, foxes absent, 3 = foxes absent, coyotes present

by foxes. We attributed this result to the frequently observed inhibitory effects of coyote predation on smaller predators, especially foxes, opossums, skunks, and domestic cats. These smaller "mesopredators" are more likely to prey on birds and bird nests than are coyotes. Foxes, for example, frequently forage by climbing bushes and small trees.

4. There was no statistically significant distance effect. In other words, the persistence of bird species in isolated fragments appears to be unaffected by the proximity of canyons to each other or by the distance to the closest unfragmented "mainland" habitat. Our interpretation of this finding is that the CR birds are virtually unable to cross barriers (streets, freeways, subdivisions), and thus are unlikely to benefit from proximity of other habitat islands. This is not to say that they

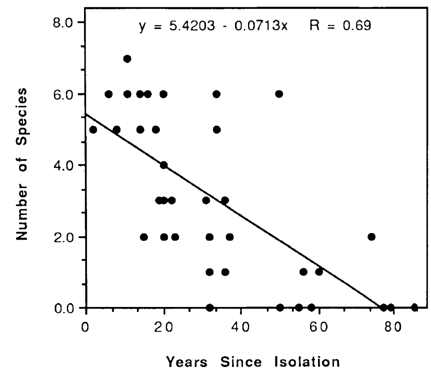

Fig. 3 The relationship between the number of chaparral-requiring bird species and the number of years since canyon isolation in 36 isolated canyons in western San Diego County

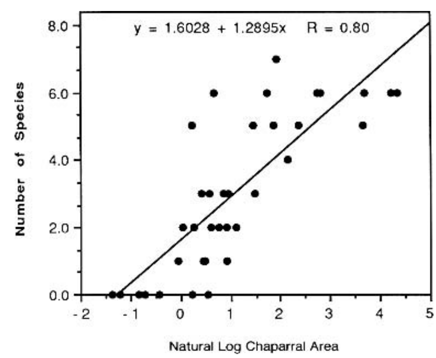

Fig. 4 Species-area relationship for chaparral-requiring bird species in 36 isolated canyons in western San Diego County. Area is actual chaparral cover in the canyons in hectares (natural logs); it does not include disturbed habitat or habitats dominated by alien species

are unable to fly the necessary distances, though many are indeed weak flyers. Rather, the poor dispersal ability of CR birds probably represents an intrinsic aversion to abandoning cover. In any case, recolonization of canyons following local extirpations appears to be rare (Soulé et al. 1988).

The dramatic loss of species in canyons is not limited to birds. The attrition of native mammals, such as rodents, rabbits, and hares, occurs even more rapidly. These native mammal species are replaced in the canyons by non-native (alien) species, notably house mice (*Mus musculus*), black rats (*Rattus rattus*), and opossums, a relatively recent invader from the east. Anecdotal evidence from questionnaires passed out to local residents also suggests a rapid loss of reptiles from isolated canyons. Table 2 contrasts the kinds of birds and mammals that are found in long-isolated, disturbed canyons with the kinds observed in recently isolated, relatively undisturbed canyons.

Anticipating Future Extinctions

Multiple regression is often used to obtain an equation that can be used for predictive purposes. Urban planners and conservationsists wishing to anticipate the fate of CR birds in a particular habitat fragment in the southern California region could use an equation derived from the multiple regression results in Soulé et al. (1988) in order to predict the number of species that will persist in a particular canyon after a certain number of years of isolation. The equation derived from the results is

$$\begin{aligned} S_t = 4.6 &- 1.4\ (\text{in AGE}) + 0.6(\text{in CHAP}) \\ &+ 0.8(\text{in AREA}) + 0.7(\text{FOXCOY}). \end{aligned}$$

Table 2 Species of locally breeding birds and mammals expected to occur in canyons of different ages and degrees of disturbance

Long-isolated, disturbed canyons	Recently isolated, undisturbed canyons
Pigeon	Roadrunner
House finch	California quail
Starling	California thrasher
Mockingbird	Rufous-sided towhee
Bushtit	Bewick's wren
English sparrow	Wrentit
Brown towhee	Brown towhee
Black phoebe	Scrub jay
Flicker	Mockingbird
Grey fox	Bushtit
Striped skunk	Black phoebe
Opossum	Flicker
House mouse	Coyote
Black rat	Jackrabbit
House cat	Brush rabbit
Gopher	Dusky woodrat
	Woodrat
	Deer mouse
	California mouse
	Pocket mouse
	Grasshopper mouse
	Meadow vole
	Gopher

where S_t is the number of species at time t, AGE is the number of years since the isolation of a canyon, CHAP is the area of natural cover in hectares. AREA is the total area of the canyon in hectares, and FOXCOY is a score based on the presence/absence of fox and coyote. [The values for AGE, CHAP, and AREA are converted to natural logarithms (ln) before being multiplied by their respective coefficients.] Estimates of future values for CHAP and FOXCOY can be based on data in Soulé et al. (1988). Note that the numerical values given in the above equation take into account the correlations of the variables, and differ, therefore, from those shown in Figs. 3 and 4.[3]

Say, for example, that a 2-hectare (5-acre) canyon was to be isolated by a pending subdivision. One might want to estimate the number of species of CR birds that would remain in the canyon in five years, twenty-five years, and seventy-five years. Assuming for the sake of simplicity that FOXCOY has a value of 3 (coyotes present, foxes absent), and using the above equation, the corresponding number of CR species that would be predicted to persist following these intervals are 3.53, 1.26, and −1.18 (or zero), respectively. (The 95-percent prediction intervals around these values are approximately plus or minus 1.9 species.) Because nearly all canyons lose natural habitat with time, let us assume that 25 percent of the chaparral is replaced by non-native vegetation in 25 years, and that 50 percent is replaced in 75 years. Recalculating the number of surviving CR species with these reductions in habitat gives 1.1 species in 50 years and −1.59 in 75 years, respectively. Even given the broad prediction intervals, it is unlikely that any CR species will survive for 75 years.

[3] The regression equation is for this bivariate relationship only and should not be used for predictive purposes when other biogeographic information is available.

Such predictions are rough approximations. Nevertheless, this approach can provide estimates of the impact of fragmentation, thus transforming a nebulous warning (“extinctions will occur”) into qualified mathematical statements that can be convincing tools for planners.

Other kinds of predictions can be made. Analysis of the vulnerability of individual species has provided a basis for predicting the sequence in which they disappear. Two factors account for about 95 percent of the variation in persistence among species (Soulé et al. 1988). In order of importance these are (1) average abundance of the particular species in typical habitat and (2) body weight. Thus, the order in which CR species drop out of the isolated canyons is highly predictable: from most to least susceptible, it is cactus wren,[4] black-tailed gnatcatcher, roadrunner, California quail, California thrasher or rufous-sided towhee, Bewick’s wren, and wrentit. Knowing the likely sequence of extinctions could be an important element in long-range environmental planning.

Planning Guidelines for Protecting Wildlife in Fragmenting Systems

The results of the San Diego case study demonstrate most of the principles established by similar research throughout the world (Brown 1971; Emlen 1974; Diamond 1975; Schoener 1976; Diamond et al. 1987; Soulé et al. 1979; Karr 1982; Brittingham and Temple 1983; Blake and Karr 1984; Howe 1984; Lynch and Whigham 1984; Patterson 1984; Lovejoy et al. 1986; Terborgh and Winter 1980; Wilcove et al. 1986; Newmark 1987; Terborgh 1989). The factors that make the San Diego study particularly relevant to planners are its urban setting, the availability of information on the “ages” of the fragments, and the small size of the habitat isolates that contributed to the rapidity of extinctions.

Island biogeographic studies can provide a basis for guidelines on maintaining wildlife and ecosystem values in areas subject to habitat fragmentation. For the planning field, the most important conclusion from this entire body of investigations is that *the best way to maintain wildlife and ecosystem values is to minimize habitat fragmentation*. Where urbanization is occurring, however, habitat fragmentation is virtually inevitable, and one of the only practical mitigation measures is the establishment of corridors of natural habitat or linkages, such as underpasses, that permit dispersal across barriers. There has been some debate about the utility of corridors (Soulé and Simerloff 1985; Simberloff and Cox 1987; Noss 1987), and acknowledgment of their disadvantages in some situations. But this author believes that corridors are the best solution, especially where species are disappearing from small, local fragments in a predictable order, producing nested species distributions based on habitat area (Patterson and Atmar 1986), and where the target species do not disperse well across barriers.

Other caveats may apply, however, especially for plants. Small and isolated habitat fragments might be adequate to protect certain kinds of plants, including endangered or threatened species, assuming that such plants (1) are not suppressed by or dependant on fire, (2) are not subject to inbreeding depression or the loss of genetic variability (Ledig 1986; Shaffer 1981; Frankel and Soulé 1981; Schonewald-Cox et al. 1983), (3) do not depend on animal pollinators or seed dispersers, and (4) compete well in the absence of habitat disturbance caused by large animals and fire. On the other hand, plants that are subject to the above forces, or to the various kinds of edge effects, such as trampling, dessication, wind, over-collecting, competition from weedy species, and cropping by domesticated animals, will not fare well in small fragments unless managed intensively. Vulnerability must be examined on a case-by-case and species-by-species basis.

Figure 5 illustrates the planning guidelines for animals suggested by the San Diego results and those of most other studies. Part A of Fig. 5 illustrates the superiority of large over small habitat

[4] This species was not included in our analyses because it only occurred in one canyon.

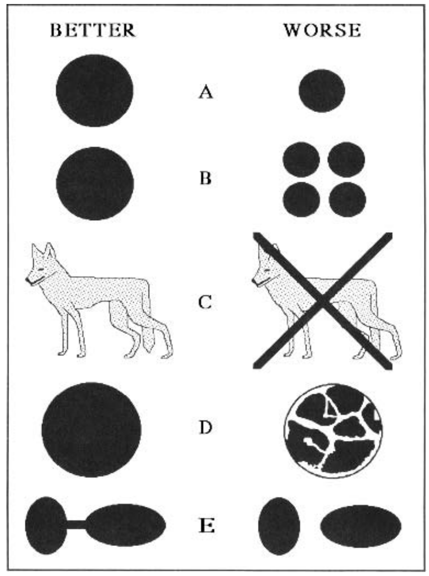

Fig. 5 Summary of planning guidelines based in part on studies of faunal extinctions in fragments of chaparral habitat in San Diego County

fragments. Wherever possible, natural open space elements should be as large as possible and should be made contiguous. As shown in Fig. 4, retention of CR birds is highly correlated with the amount of habitat. One reason for the superiority of large fragments is that they can support a larger number of individuals for a particular species. As already mentioned, the probability of extinction is inversely proportional to population size.

Large fragments also minimize edge effects (see Fig. 1). Some species will never breed in small habitat fragments, even if they use them for foraging. These organisms include those species that require undisturbed (interior, non-edge, old growth) habitats, as well as those that may need a variety of habitats. In the Midwest, many bird species that require forest interior habitat cannot be found breeding in patches of forest that are less than 25 hectares in area (Blake and Karr 1984). For some animals, roads produce formidable edge effects (see, e.g., McLellan and Shackleton 1988). The degree to which these negative effects of edges will diminish the value of a particular site depends on the habitat, the region, and the species under consideration. When in doubt, experts should be consulted.

Part B of Fig. 5 illustrates a more controversial guideline—a single large habitat fragment is superior to several small fragments, at least for vertebrate animals. This principle does not apply to all biological systems, although the canyon data strongly support it, as do data from virtually all studies of vertebrate animals. Our mammal surveys (unpublished data) lead to the same conclusion. The empirical basis for this guideline is the observation that extinctions of vertebrate species in fragments of similar habitat nearly always occur in a regular and predictable order (Patterson and Atmar 1986). In our study, for example, the roadrunner and the black-tailed gnatcatcher always disappear first. At the other extreme are the wrentit and Bewick's wren; they are always the last survivors in older and smaller canyons. On the other hand, if extinctions were random with respect to species, then several small fragments would, collectively, have as many or more species than a single large fragment equal in area to the sum of the small fragments.

Another caveat pertains to some highly mobile animals, including many species of temperate forest birds. For these animals, a multiplicity of habitat (forest) types may be more important than area per se (see, e.g., Beissinger and Osborne 1982). One must bear in mind, however, that attempts to breed by such birds in small habitat fragments often fail (Terborgh 1989) because of nest parasitism by cowbirds (Brittingham and Temple 1983) and nest predation by edge species, such as jays, crows, raccoons, house cats, rats, dogs, skunks, and opossums (Wilcove et al. 1986).

Part C of Fig. 5 symbolizes the advantage of retaining the large carnivores in a system. In the San Diego case (Soulé et al. 1988) and in others (Terborgh 1988), there is indirect evidence that large predators prevent abnormally high population densities of smaller mesopredators (including domestic and feral house cats) that are likely to prey on birds. Unless there are compelling reasons to do otherwise, planners should oppose the "control" of coyotes, bobcats, badgers, and mountain lions (cougars, panthers). An analogous guideline from the ecology field is equally important: manage the system in order to maintain habitat-modifying animals such as tortoises, alligators, moose, beaver, muskrat, and pocket gophers: such animals create and maintain a mosaic of habitats that facilitate the persistence of many other species of plants and animals (Harris 1988).

Part D of Fig. 5 shows the problem of human disturbance. Chaparral is a rather brittle habitat: it is easily and permanently destroyed by trampling, bushwhacking, frequent fires (Westman et al. 1981), or grading. Other sensitive habitats include heaths, wetlands, sand dunes, and some forests that, when "opened up" or "cleaned up," drained, or "improved" by trail or road development, are exposed to accelerating or cumulative changes, including the invasion of weeds and mesopredators. A corollary of this guideline is that development configurations should minimize adverse edge effects. Trails, roads, and similar facilities increase the frequency of human contact, and may eventually lead to the disappearance of sensitive species. In addition, such improvements increase the amount of edge. Deleterious edge effects, such as predation, nest parasitism (from cowbirds), fire, dessication, noise, and invasion of introduced plants and animals, are often mutually exacerbating. Their impacts also increase as patch size decreases.

The apparent contradiction between this anti-disturbance recommendation and the previous mention of the benefits provided by animals that produce extensive habitat disturbance (alligators, beavers, pocket gophers, etc.) is real, and illustrates the contextual nature of all guidelines. Whether disturbance is beneficial depends on many factors, including scale (e.g., the size of the fragment), the habitat type, the likely longevity and objectives of the project, and the kind and degree of disturbance (Pickett and White 1985). Local ecologists should be consulted if there is a question about disturbance dynamics.

Part E of the guidelines demonstrates the corridor principle—maintain continuity and flow between patches of chaparral and other habitats. Corridors, including under-road links, can mitigate some of the deleterious effects of fragmentation (Forman and Godron 1986). Wildlife corridors can be viewed as a kind of landscape health insurance policy—they maximize the chances that biological connectivity will persist, despite changing political and economic conditions. The design of wildlife corridors, however, is a new branch of conservation biology. For this reason and others, there are few, if any specific guidelines. Potential corridors must be analyzed and designed by teams of planners, engineers, and biologists on a case-by-case basis. Admittedly, wildlife linkages involve capital investment up front: but it is considerably less expensive to construct underpasses and other linkage elements for wildlife during the construction of facilities than to attempt to retrofit existing "improvements."

This corridor guideline stems from the inevitability of local extinctions in isolated habitat fragments. Though there has been little research on optimum corridor design (but see Fahrig and Merriam 1985; Fahrig and Paloheimo 1988; Soulé and Gilpin 1991), particularly as it affects the movement of different kinds of organisms, many of the CR birds have been seen moving and feeding in strips of chaparral only a few meters wide (Soulé et al. 1988). Planners should bear in mind, however, that species differ markedly in habitat needs and tolerances, and that the utility of particular corridors for wildlife (Harris and Gallagher 1989) depends on the behavior of the targeted species.

For some highly mobile species, the distance between fragments will be relevant. For the CR birds it is not. Our results suggest that close proximity of fragments does not retard the rate of species loss, unless the patches are separated by less than a few dozen meters (Soulé et al. 1988).

The reason is that the CR birds disperse poorly, if at all, through non-native habitat. Our results (unpublished) for rodents, rabbits, and hares, on the other hand, suggest a minor distance effect, indicating a slight benefit of patch proximity for these mammals. For most nonflying animals in most places, however, proximity of habitat remnants will not retard species loss unless the patches are connected by corridors.

Other Recommendations

The preceding observations suggest that the best way to fight the deleterious effects of fragmentation is to prevent it. Wherever possible, therefore, planners should insist on the linking of habitat elements by habitat corridors. This suggestion obviously assumes that it is possible to do planning on a scale that is larger than the individual housing or commercial development.

Where corridors are not practical, there are other ways to mitigate fragmentation. One is to ensure that open space set-asides are contiguous. Such aggregation of open space is implicit in guidelines A and B above. Even if such open space aggregation is accomplished, however, corridors between these larger aggregates are highly recommended. A second possibility, where both landscape linkages and juxtaposition of open space elements are impractical, is "mitigation banking"—the developer, instead of setting aside tiny parcels that will deteriorate rapidly, deposits money into an account for future open space acquisition.

A third alternative is a permanent committment to the artificial transport of organisms on a schedule that precludes the extinction of isolated populations. Translocation requires less capital investment than highway underpasses dedicated to wildlife, but assumes that jurisdictions and management agencies will commit funds indefinitely for the capture and release of animals. In many cases, however, the infrastructure does not exist to routinely translocate animals, or the procedure is prohibitively expensive. In addition, translocated animals usually do not survive, and expensive monitoring programs are necessary. For these and other reasons, there are few if any programs that routinely transfer wildlife for the purposes of maintaining population viability.

Land use planning involves many variables that are not in the province of the natural scientist. Nevertheless, scientists can assist planners in the analysis of the available land-use options. For example, depending on the stage of development and the kind of habitat, many "improvements," including highway shoulders, the edges of bicycle and foot paths, streams on golf courses and in parks, and utility rights of way may facilitate animal movements. In addition, some species, including large predators, can take advantage of culverts and underpasses, especially if these facilities are designed with animal dispersal in mind. Biologists should be consulted when such alternatives are being considered.

Some conflicts between recreational uses and wildlife values in corridor design are inevitable. For example, cover is important for chaparral birds and other small vertebrates. The public would have to tolerate a certain "untidiness" in open space systems designed for both wildlife and people. Public education about such matters is a perennial requirement.

A question not addressed here is how large is large enough to maintain a population of a species? Questions of this genre can only be answered probabilistically—the larger the population (or the patch size in most cases), the higher the chance that the species will persist over a given interval. Such answers may not be satisfying, but the question of population viability is extremely complex (Shaffer 1981; Gilpin and Soulé 1986; Soulé 1987), and good answers to complex questions are contextual. In practical terms there are no magic thresholds of population or ecosystem viability.

Planners are increasingly called upon and held accountable for the present and future quality of the human environment. One body of information that could help planners ensure a more interesting, more diverse, and more natural environment is that provided by island biogeography. This field, as well as other aspects of ecology, become increasingly relevant where the landscape is usurped and fragmented by humans, and where the remnants of natural habitats are isolated. The preceding results and discussion constitute an attempt to begin a dialogue between planners and conservation biologists.

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Terrestrial Nature Reserve Design at the Urban/Rural Interface

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Keywords: urban planning · fragmentation · population viability · reserves · corridors · island biogeography · meta population

Introduction

Wisconsin had 28 nature reserves five years after the Wisconsin State Board for the Preservation of Scientific Areas was created in 1951. Iltis (1956) remarked that "time is running out," indicating that 280 or even 500 reserves were needed. By 1993, Wisconsin had a remarkable 276 dedicated nature reserves. However, for much of the Midwest and other parts of the United States, time is running out in spite of some remarkable achievements (Figs. 1–2). Awareness is increasing that terrestrial conservation efforts in some parts of the United States must by necessity be on small pieces of habitat, supporting small populations of species (e.g., Mitchell et al. 1990). Small reserves are important in areas where landscape alteration is very high *and* very low (Shafer 1995).

The effects of people on any landscape can be either dramatic or subtle where human populations are dense (McDonnell and Pickett 1993). In California, documentation of the loss of biodiversity (e.g., Jensen et al. 1993) has resulted in endorsement of new planning approaches by the highest state government officials. Perhaps because urbanization is moving closer to our rural and wilderness areas, this interface is receiving more attention in research (e.g., McDonnell and Pickett 1990). Urbanization, with its accompanying loss of native habitat and creation of new habitat, has been correlated with a decreasing number of bird species, increasing avian biomass, and increasing dominance of a few species (Emlen 1974, Beissinger and Osborne 1982).

Urban areas, however, are not necessarily a death knell for all wildlife. In fact, many opportunities for wildlife habitat or corridors in urban/suburban areas, like golf courses, are overlooked. (Terman 1994). Red foxes in Great Britain use railroad corridors to travel in and out of towns and cities (Kolb 1985, cited in Adams 1994), and Adams indicates that white-tailed deer, coyotes, and raccoons are thriving in some U.S. urban areas. Their presence may cause concern when deer browse on home shrubbery and gardens, coyotes attack pets, and raccoons transmit disease or raid home garbage cans. Gill and Bonnett (1973) document how many species occur in London and Los Angeles. "The wolves, mountain lions, bears, salmon, and oysters that were part of Manhattan are gone, but the red fox, opossum (a new resident), flying squirrel, gray squirrel, muskrat, raccoon, several species of bats, and a host of birds remain" (Ehrenfeld 1972, 182–183).

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Fig. 1 A 48-acre virgin beech-maple forest in Indiana. The tree is a 68-inch DBH Shumard's red oak
Source: Photo taken 1973, courtesy U.S. National Park Service

Fig. 2 A 330-acre remnant tallgrass prairie near the eastern margin of Indiana's "Prairie Peninsula"
Source: Photo taken 1973, courtesy U.S. National Park Service

European Precursors

Westhoff (1970) categorized landscapes as natural (undisturbed—no longer present in western and central Europe), *subnatural* (human influenced but still related to the potential natural vegetation), *seminatural* (sites now very different from the potential natural vegetation such as heathlands and moors, undrained/mowed/leveled dry pastures and hayfields, hedges, coppices, or older coastal dunes), and *cultivated* (e.g., a bean field). Seminatural landscapes predominated in western and central Europe from the Middle Ages until the end of the 19th century. Westoff maintains that human influence during this time was more positive than negative. His explanation for this claim is that land management amplified and stabilized biotic variation. Land management methods stayed the same for centuries, people did not travel far from home, and their operations were gradual and of small scale. van der Maarel (1975) gives the distribution of plant species in the Netherlands by degree of naturalness: 20% near natural (e.g., woodlands, bogs, dunes), 60% seminatural (e.g., hay meadows, grazed salt marshes, coppices), and 20% agricultural and suburban. van der Maarel claims many plant species are tied to, or have an advantage in, seminatural environments. Similarly, Erhardt and Thomas (1991) claim a large percentage of British butterflies are confined to human-made niches, like secondary grassland created or maintained by agricultural practices.

Although Native Americans affected the North American landscape before Europeans arrived (Denevan 1992), their influence was much less profound compared with what Europe experienced. Much of the U.S. Midwest resembles the seminatural landscape that predominated in Europe for centuries. Europeans tried to devise conservation strategies for their countries accordingly. We have much to learn from this European example as parts of our country are modified to less wilderness characteristics (see Green 1981). Although we should focus much of our efforts on *remnant ecosystems/natural islands*, other classifications with more urban affinities also harbor some native biota, e.g., urban savanna, mowed/grassland, urban/forest plantation, rail-highway/grassland, and so forth (Brady et al. 1979).

Remnant Persistence

Statistics about how much is left of a particular biotic community are common and depressing. For ecosystems whose spatial extent has declined by more than 98% in the United States, the greatest losses are from grassland, savanna, and barrens (Noss et al. 1995). Only about 2% of California's interior wetlands remain. Some of these communities were not widespread to begin with; for example, vernal pools covered only 1% of the state and 80% are gone (Barbour and Whitworth 1994). Leopold (1949) long ago pointed out that some of the best examples of prairie communities are remnants found along railroad rights-of-way. Betz (1977) explains that many tiny tracts can still be found along fenced railroads, along farmers' creeks, in hay meadows, in cemeteries, and in some suburban areas. Some states, such as Minnesota, have gone to great lengths to inventory their highways for remnant prairies (see Harrington 1994). In southern Saskatchewan, less than 1% of the grasslands ecoregion is in highly protected reserves (Gautheir and Patino 1995). Less than 202 ha of intact oak savanna remain in Wisconsin, which is less than 0.01% of the original 5.5 million acres in the state (Department of Natural Resources 1995). A 1985 inventory of the entire Midwest revealed that only 0.02% of this plant community is left (Henderson and Epstein 1995). These intact remnants (prairie and savanna) are almost exclusively found on marginal soil types (with the exception of railroad rights-of-way).

The most important woodlands in England and Wales—the species-rich "ancient" tracts that have persisted since the Middle Ages—were recently better quantified by Spencer and Kirby (1992). They found that those woodlands still covered a remarkable 2.6% of the land surface, with 83% of the sites under 20 ha. They calculate that 7% of the ancient tracts were lost in the last 50 years. Such detailed baseline information is necessary in setting conservation priorities in human-dominated landscapes. Some British biotic communities are also very rare. For example, 0.1% of the peat fens in eastern England, such as Wicken Fen, remain undrained. The swallowtail butterfly *Papilio machaon* was lost from Wicken Fen in 1952, leaving only one other population in Great Britain. Studies have focused on why reintroductions have not yet worked, presumably because of food plant availability (Harvey and Meredith 1981).

The $140,000\,\mathrm{km}^2$ Western Australian wheatbelt once had $41,000\,\mathrm{km}^2$ of woodland, but only $1,000\,\mathrm{km}^2$ remain. The Australian government designated 639 forest patches as reserves, and thousands of other privately owned patches are scattered throughout the region. It represents an enormous test case in potential cooperation with private landowners to preserve patches and decrease patch isolation. Many species have already been lost as a result of fragmentation, but positive action rather than resignation may permit others to persist (Saunders and Hobbs 1989).

At times native biotic communities can display surprising resiliency, and their persistence is known only after detailed inventory. For example, the Canterbury Plains in New Zealand was assumed to have lost many of its native biotic communities after 100 years of deforestation. In spite of its apparent continuous expanse of farmland and urban areas, a closer look revealed that many of these communities still survived (Molloy 1971). The Crown purchased 2 to 3 ha tracts of vegetation types that once covered approximately 100,000 to 200,000 ha. The best remaining examples of some vegetation types included 526, 16, and 3 ha tracts.

Remnants As Refugia

Species and Area

The theory of island biogeography (MacArthur and Wilson 1963, 1967) has been equated with the beginning of conservation biology (Simberloff 1988). The theory served as a foundation for thinking about nature reserve design in the 1970s and later. The empirical basis supporting such use is slim, and respect for the theory's conservation usefulness has declined (Shafer 1990, Formann 1995). "The inability of ecological theory to predict precisely future population sizes, the rate at which a fauna will collapse following insularization of its habitat, or the response of an ecosystem to a complex series of insults does not necessarily represent failure of the theory . . .. The problem of balancing precision against generality is much more difficult for ecological theoreticians than it is for theoretical physicists" (Ehrlich 1989, 315). Complete agreement in the scientific community about the theory's empirical foundation does not exist even today (Rosenzweig 1995). However, most would agree that much more autecological information is needed for specific reserve design prescriptions. McCoy (1983) pointed out that the minimum area needed by a suite of butterfly species can only be determined by detailed autecological study, and area alone may not be the most important factor.

The species-area relationship (see Williamson 1988), one component of the theory of island biogeography, may have relevance to conservation practice in some situations (see Shafer 1990 for detractors). That species increase with area is well known, with very rare exceptions (e.g., Dunn and Loehle [1988] for plants). Birds are one of the best studied groups in this regard. The number of rural studies that found that bird species increased with woodlot size is substantial: Freemark and Merriam (1986) outside Ottawa, Canada; Opdam et al. (1985) in the Netherlands; Lynch and Whigham (1984) on Maryland's coastal plain; Woolhouse (1985) in Great Britain; Ambuel and Temple (1983) for southern Wisconsin; and Kitchener et al. (1982) for western Australia. For more urban areas, habitat size has been shown to determine species number for birds (Gavareski 1976, Tilghman 1987, Vizyova 1986), for reptiles and amphibians (Dickman 1987, Vizyova 1986), and for small mammals (Matthiae and Stearns 1981).

Small but Not Vacant

That small, urban parks could play a role as nature reserves has often been ignored in our focus on recreational/psychological values (e.g., Seymour 1969), but this is changing (e.g., Spirn 1984, Gilbert 1991, Adams 1994, Platt et al. 1994). Dickman (1987), based on species-area relationships in the city of Oxford, thought mammal species (excluding deer) could be maintained in a system of small 0.65+ ha habitat patches and amphibians and reptiles in 0.55+ ha patches with permanent water. Because there was no temporal dimension to the study, presence may not necessarily mean persistence.

Based on a survey of 72 remnant grasslands in the Chicago area between 1982 and 1994 and other information, Panzer et al. (1995) concluded that around 25% of the insect species are remnant dependent. One small English garden contained 21 of the 70 known butterfly species in Britain (Owen 1978, cited in Adams 1994). However, more than one patch may be needed to ensure their survival (Hanski and Thomas 1994). One square foot of Pacific Northwest old-growth forest soil and litter can yield 200 to 250 species of invertebrates (Moldenke and Lattin 1990).

Some authors have concluded that suburban and urban parks are unsuccessful as avifaunal reserves due to small size, isolation, and vulnerability to human impacts (e.g., Lynch and Whitcomb 1978). An ongoing study of 225 forest fragments in Prince Georges County, Maryland, including some urban tracts, has a minimum size cutoff of 0.5 ha (Robbins, personal communication). Some birds will nest in the smallest tracts, often "suburban" species (Robbins et al. 1989a). In their literature review, Adams and Dove (1989) made predictions for expected species number as a function of habitat size—they thought some woodland and chaparral birds would be present in 1 ha remnants. Such remnants can provide habitat to produce some birds, which is different from providing viable habitat by themselves. Sometimes a small remnant can be the last refuge for a plant or invertebrate species (Shafer 1995). Ehnström and Waldén (1986, cited in Hansson 1992) describe a 5 ha old oak forest in Sweden that is the last refuge for some species of rare beetles.

Population Viability

The persistence of *minimum viable populations* (MVP) has been defined as hinging on genetic, demographic, and environmental stochasticities, and natural catastrophe (Shaffer 1981), although the term MVP was in use earlier (e.g., Frankel 1970). Early on, very rough generalizations emerged about how large a population needed to be to persist for a certain length of time. For example, it was proposed that a mean of 2,000 vertebrates (give or take one order of magnitude) was needed for a 95% expectation of population persistence for 200 years (Soulè 1987b). Soulè and Simberloff (1986, 32) state: "Thus, not only is there no magic number, there is no magic protocol. Intuition, common sense and judicious use of available data are still the state of the art." Thomas (1990) proposed to move Soulè's (1987b) well-known generalization of "low thousands" to a mean of 2,000 to 10,000 (Note: Soulè [1987b] also used the terms "few thousand" or "several thousand.") Lande (1995, 789) argues for an effective population size of 5,000 "to maintain normal levels of potentially adaptive genetic variance in quantitative characters under a balance between mutation and random genetic drift" (*see also* Culotta 1995). A *population viability analysis* or PVA (Gilpin and Soulè 1995), reviewed in detail by Boyce (1992) and Ballou et al. (1995), is far better than relying on any generalizations. As Holsinger (1995) pointed out, only a few endangered species are likely to receive a complete population viability analysis because of the enormous data-gathering work required. Such best-data scenarios will still not allow fine predictions about needed population size. Why is this?

Genetics is presumed the least important component of MVP, and catastrophe the most important (Shaffer 1987). Since catastrophe is so difficult to account for, long-term predictions by PVA are still in the realm of guesswork (Barrow-clough 1992). Additionally, the deterministic human dimension (e.g., human population density, development and pollution, or exotic species and climate change) could overshadow any so-called stochastic events in traditional PVA. Demographic stochasticity was claimed to be more important than genetics (Lande 1988). Similarly, Brakefield (1991) maintains that an insect population size that minimizes ecological extinction (providing effective population size does not go below several hundred individuals and longer-term evolutionary potential is not taken into account) should automatically take care of genetic variation. Nunney and Campbell (1993), in contrast, maintain that genetic and demographic concerns dictate a similar population size threshold. Regardless, we can still be fairly confident that the upper threshold will be dictated by catastrophe.

The Nature Conservancy concluded that 1,678 United States plant taxa (8.4%) are known from five or fewer locations or less than 1,000 individuals (Falk 1991). In spite of the pressing need, addressing MVP for plants is more recent (Menges 1990). Menges suggested that minimum island size may not be important for plants but metapopulation considerations will be. Weaver and Kellman (1981) concluded that area and isolation did not explain tree species persistence or loss in ten Ontario woodlots. Ouborg's (1993) data from the Dutch Rhine caused him to conclude that metapopulation structure, the negative effect of isolation, and population size was important for some plant species. Widén and Svensson (1992) assume that self-fertilizing annual plants that are selected for inbreeding may not be harmed genetically by habitat fragmentation but outbreeding perennials could be. However, they conclude that present empirical knowledge about genetic diversity and population size in plants is still insufficient to confidently devise strategies to thwart habitat fragmentation. Inbreeding depression in plants has been invoked as a cause of poor survival (Menges 1991, Waller 1993). Schemske et al. (1994) found that the primary cause of endangerment for all but 1 of 98 U.S. plant species listed as threatened or endangered by the U.S. Fish and Wildlife Service was human activity. Most ultimate causes of animal extinction today are probably anthropogenic, although the proximate cause (i.e., reason the last individuals die) could be genetic, demographic, by catastrophe, or through direct human action like collecting or hunting (Simberloff 1986b).

Caughley (1994) contrasted the small population versus declining population paradigms, arguing that the latter has received much less attention but is more germane to conservation. Unless these two things are combined in PVA, the factors probably most significant to a population's survival will be ignored. The National Research Council (1995) concluded that all PVAs are limited by data and methods; that most PVAs vary only some important influences, resulting in casual estimates; and that single factor PVAs will underestimate extinction threats. Because "formal population viability analyses are complex and are impossible to conduct on a routine basis" (Ruggiero et al. 1994, 371), these authors recommended a shortcut to allow managers to do some impact assessment. PVA should be made more available and digestible to managers, with or without shortcuts. Better yet, we need to focus on the real driving factors in any PVA, which requires transdisciplinary approaches. The traditional approach at PVA may be more comfortable to biologists (e.g., Remmert 1994) but is not a depiction of the real world. (Note: There is further debate on these points in the August 1995 issue of *Conservation Biology*.)

Metapopulations

The term *metapopulation* is usually attributed to Levins (1968, vi): "any real population [that] is a population of local populations which are established by colonists, survive for a while, send out migrants, and eventually disappear," although Simberloff (1988) pointed out a form of the idea that arose earlier. The rough metapopulation idea involves a set of geographically distinct populations together comprising a larger population. These subpopulations occasionally receive immigrants amongst one another; there can be a "winking" on and off (local extinction) of subpopulations; but the overall metapopulation persists (see Gilpin and Hanski 1991, Wilson 1992). Conservationists have used the metapopulation model as rationale for preserving multiple-habitat patches or reserves, presuming some species are adapted to this population structure. Some others have used it as a reason why it is acceptable to give up some local populations! The degree to which this model has been supported by field data hinges on the rigidity of model definition (see Shafer 1995). However, the idea that some species now exist in small patches is not arguable. For example, Hanski (1994) indicates that the Finland butterfly *Melitaea cinxia* lives in a series of 50 small patches, most under 1 ha.

The important underlying conservation assumption is that one habitat or reserve is not enough if we want to simulate a population's natural metapopulation structure. Bank voles showed a pattern of recolonization following local patch extinction. Extinctions were most likely in woodlots under

0.5 ha, and their abundance decreased as distance increased from woods of more than 25 ha (van Apeldoorn et al. 1992). Even an enormous population of small organisms is not necessarily safe on a small habitat patch. Tscharntke (1992) concluded that populations of 180,000 adult moths *Archanara geminipuncta* cannot persist on 2 ha *Phragmites* nature reserves without nearby reservoir populations. However, most questions about reserve size, numbers, and distance between habitat patches for invertebrates remains a mystery due to lack of dispersal data (Thomas and Morris 1995).

After three decades of research on the bay checkerspot butterfly *(Euphydryas editha bayensis)*, the modeling of Murphy et al. (1990) permitted a reserve design conclusion: small, low-quality, serpentine grassland patches within seven miles of the largest reservoir patch could be as important, or more important, to the survival of the metapopulation than larger, higher-quality patches at greater distances. Computer simulation modeling conducted by Fahrig et al. (1983) led to the following conclusions: links between habitat patches are important and there is a minimum number of patches needing connection.

The metapopulation concept involves *replicates* of habitat. However, the early recognition that more than one reserve is desirable was not tied to metapopulation theory (e.g., Specht et al. 1974) but to intuitive common sense. The replication message became intertwined and perhaps obscured with the academic Single Large Or Several Small controversy (abbreviated SLOSS) that began in 1976 (Simberloff and Abele 1976). The early SLOSS debate centered around whether it is more desirable to have *(but not necessarily retain over a long period)* species in one large reserve or in a number of smaller reserves whose total area equals that of the single large one. Whether one large reserve is better than several small reserves was raised earlier (e.g., Bourliere 1962, 66) but not as a scientific hypothesis. The advantages of replication, irrespective of SLOSS, was occasionally pointed out (e.g., Soulè and Simberloff 1986; Shafer 1990, 1994, 1995). The mean size of scientific areas in Wisconsin (18.8 ha) is smaller than the 50 ha typically affected by individual tornados (Guntenspergen 1983).

Lessons from the Temperate Zone

Moore (1962, 390) implied that a biotic community has a minimum size—"The smallest viable size of a habitat is the smallest which supports a viable population of its weakest species." There have been efforts for some time to gauge it from species-area relationships (e.g., Vestal 1949). However, Usher's (1986) review led him to claim that minimum biotic community size is yet to be determined for any community. This claim has not stopped scientists from providing their best judgements for biotic communities however defined or demarcated. For example, plant diversity declined with heathland fragment size in Dorset, England (Webb and Vermaat 1990), and the authors recommended 55 ha for maximum heathland plant representation. Levenson (1981) estimated a 4 to 5 ha undisturbed tract was needed to secure the future of all plants characteristic of a southern Wisconsin mesic beech-maple forest and 7 to 8 ha was needed for a dry mesic oak forest. The reason was that below this size the invasion rate by edge-adapted, shade-intolerant tree species was too high. Another biotic edge effect is nest predation. Species like cowbirds are severely decreasing the survival of neotropical migrant birds, no longer protected in deep interior forests because of habitat fragmentation (Wilcove 1985). Woodlot edges are also created by human impacts (Matlack 1993). Some think edge effects encompass a plethora of human encroachments on national parks (e.g., National Park Service 1980), but many might best be called matrix effects. Schonewald-Cox and Bayless (1986) proposed an all-encompassing boundary effects model. The intuitive assumption that human impacts would be greater in small tracts arose earlier (Wright et al. 1933, 43).

Mader (1984) indicates that very small tracts (less than 0.5 ha) in West Germany should be disregarded because they are all edge and no core. This does not mean, however, that they have no biological value to conservation. Although very small reserves might not allow long-term persistence for certain species, particularly large mammals, many other species use them and some small

HOW SMALL A RESERVE?

Arctic National Wildlife Refuge, Alaska	7,804,819 ha
Everglades National Park + Big Cypress National Preserve, Florida	796,809 ha
Shenandoah National Park, Virginia	79,055 ha
Congaree Swamp National Monument, South Carolina	6,126 ha
Muir Woods National Monument, California	224 ha
Davis-Purdue Experimental Forest, Indiana	21 ha
Weston Cemetery Prairie, Illinois	2 ha

Fig. 3 Each U.S. protected area is approximately one-tenth the size the one listed above it

species may be able to persist in them (Fig. 3). Small tracts have other values too (Shafer 1995), like education, science, habitat to facilitate dispersal, and providing propogules for restoration.

Really small tracts (e.g., 0.1 ha of vegetation) "do not reveal any fundamental diversity properties of the places or the taxa being sampled" (Rosenzweig 1995, 279). In other words, there is some data to suggest the typical species-area plot is *not* found below some area threshold—not surprising as biotic communities on such small tracts will not be unaltered representative examples of pristine species assemblages.

Lessons from the Tropics

The decline of neotropical migrant songbirds in North America is well known. What is influencing this trend the most—forest tract size for spring breeding in North America versus deforestation in the tropics where they spend the winter—is not yet known (Robbins et al. 1989b). Some neotropical migrant bird species need forest tracts of at least 3,000 ha to breed in North America (Robbins et al. 1989a). However, we should not overlook that some forest-dependent neotropical migrant songbirds do survive the winter on the Yucatan Peninsula in small patches of trees in an agricultural landscape (Greenberg 1989). Patches with eight small trees (ungrazed) and ten small trees (grazed) gave three times more sightings than patches with fewer trees. Schelhaus and Greenberg (1993) provide a good compendium and analysis of tropical literature, some of which I will use here.

Lovejoy et al. (1984) found that Amazon butterflies with uniform distributions needed 10 ha tracts of tropical forest for representative communities but that butterflies with patchy distributions needed 100 ha tracts. Klein (1989) found that dung and carrion beetle communities in the Brazilian Amazon had fewer species as forest patch size decreased—100 ha, 10 ha, and 1 ha. Lovejoy (1987) indicates that the howler monkey *Alouatta seniculus* was able to persist and reproduce in all of their 10 ha isolated Brazilian rainforest patches, though many other monkey species were quickly lost. Lovejoy et al. (1986) reported that tree mortality (over 10 cm DBH) in isolated 1 and 10 ha tropical forest fragments was almost twice as high as in continuous forest. Laurence (1991) believed a tropical forest reserve that is too small may end up preserving species that could have survived outside of the reserves anyway.

A species can be found in a fragment long after its population is presumably too low to persist (Janzen 1988). Some scientists have recorded low extinctions in some tropical forests (e.g., Brown and Brown 1992), though massive extinctions are predicted by species-area relationships (e.g., Simberloff 1986a). There can be a long lag effect. The remaining species could be doomed because the species loss period following deforestation is not immediate. Science journalists (e.g., Mann 1991) might stress that continued species presence over the short term may not invalidate some species-area extinction predictions. On the other hand, these doomed species could form the nucleus of a species salvage effort.

One Danger of Guidelines—Biotic Community Size

The provision of guidelines to planners on how small a tract is too small for a population is useful, if based on thorough research. Guidelines are much more difficult for *communities*. In most cases, we simply do not know how many species and what species a small remnant will preserve. We presume that some plants and insects will fare better than medium-sized mammals and area-sensitive birds. The umbrella-species approach at gauging needed reserve size assumes that the area required to protect viable populations of large vertebrates like bears will automatically be large enough to protect other species with small home ranges (Wilcox 1984, Shafer 1995). Unfortunately, it tells us nothing about the space requirements of smaller species in the biotic community. The umbrella species may have been lost from the region long ago or the isolated remnant has become so small as to make such an approach pointless.

Misuse of size guidelines is a danger. Size guidelines do not mean sites below this size should be abandoned, serving no purpose for a "flagship" species, other species, or for science or education (Shafer 1995). Size is only one consideration. Habitat management, connectivity, replication, and buffering will also greatly influence the perpetuity of species in a habitat patch or reserve.

Value of Dispersal Corridors

Corridors have captured the attention of scientists (e.g., Saunders and Hobbs 1991), elicited guidance to planners (e.g., Smith and Hellmund 1993), and generated grassroots action to create greenways (e.g., Little 1989, Flink and Searns 1993). The pros and cons of corridors are discussed in Noss (1987) and elsewhere. I will highlight here some research that is particularly germane to very human-dominated landscapes.

Studies in the actual use of any corridor by a species is still meager (Simberloff et al. 1992). Based on painstaking research, fencerows in farmland near Ottawa, Canada, appear to allow the dispersal of chipmunks and white-footed mice, allowing populations in isolated woodlots to persist (Wegner and Merrian 1979, Fahrig and Merriam 1985, Henderson et al. 1985). However, looking at fifteen 1 to 25 ha Ottawa farmland woodlots varying from 300 to 500 meters apart, Middleton and Merriam (1983) concluded that only 7 of 86 taxa (trees, herbs, squirrels, or invertebrates) reflected any isolation influence. Most of these species may be adapted to medium-distance movement. Hence, one has to be cautious about making "island" assumptions based only on casual landscape observation. Some species may need corridors but others may not. Soulè et al. (1988) observed some California chaparral bird species occupying 1 to 10 ha ribbons of habitat and then presumed that the ribbons were needed for dispersal between larger tracts.

There are certainly documented barriers to dispersal. Mader (1984) found that some species of beetles rarely cross highways in West Germany, but mice, far more able to navigate this distance, rarely cross either. Eversham and Tefler (1994), however, argue roadside verges in the Netherlands are used by carabid beetles not as corridors but as refugia. Klein (1989) found a 100 m gap in Brazilian rainforest would affect the movement of dung and carrion-feeding beetles. Volant species presumably will be less affected by isolation, but some are poor dispersers. For example, one butterfly species *(Mellicta athalia)* rarely moves between boreal forest gaps of 1 km (Warren 1987). Knaapen et al. (1992) estimated that butterflies would have much more difficulty traversing a built-up landscape (residential, commercial, or industrial, especially with less than 5% forest cover) in the Netherlands than would deer, squirrels, or forest birds. Dispersal is important for the survival of arthropods (der Boer 1990) and barriers do exist (Mader et al. 1990).

Moon (1990) described koalas moving from one park sanctuary to another through open paddocks. These paddocks contained sparsely distributed trees as much as 300 m apart. The koalas commonly used individuals of the tree *Eucalyptus tereticornis* 14 to 18 m in height for movement, suggesting tree plantings might be feasible to enhance corridor appeal for koalas in degraded areas.

Elton (1958, 156–158), discussing the virtues of hedgerows, remarked "I cannot think of any ecological system in Britain that so clearly has all the virtues inherent in the conservation of variety. . .. They form, as it were, a connective tissue binding together the separate organs of the landscape." We need to know the habitat needs of a species to complete its life cycle before we conclude a particular habitat linkage means the difference between extinction and perpetuity. Preferably, research should come first to ascertain whether corridors would be useful for a particular species, and if so, what its dimensions should be (Simberloff et al. 1992, Hobbs 1993). However, that is a luxury often not available in places like San Diego County, California (Mann and Plummer 1995). In lieu of good data, perhaps the best advice is to maintain habitat connectivity until we know more. Elton may have agreed with this logic. Once connectivity is gone, it is very difficult to re-create. England lost roughly a quarter of its hedgerows—96,000 miles—from 1945 to 1985. A further study gave a decline of 53,000 miles between 1984 and 1990 (Bryson 1993).

Whyte (1968, 389–399) said. "The most pressing need now is to weave together a host of seemingly disparate elements—an experimental farm, a private golf course, a local park, the spaces of a cluster subdivision, the edge of a new freeway right-of-way." His reasons were not based on conservation biology but because this linearity created more "visual space" for humans to see. His idea is nevertheless valid for animal movement. Isolation may have some pluses—for example, restricting exotic species, limiting transfer of disease, and denying entry of domestic predators like cats and dogs (Simberloff and Cox 1987)—but the greater danger lies in not being able to re-create natural landscape connections (Noss 1987).

Reserve Design

More Science

Reserve design guidelines for terrestrial ecosystems, reportedly derived from island biogeography theory, were soon advocated for incorporation in the planning process of nature reserves (Balser et al. 1981). Based on a study of chaparral fragments in San Diego, California, Soulè (1991) concluded that at least three of Diamond's (1975) general reserve design guidelines would be applicable to this urbanized setting as well: large is better than small, single large is better than several small (SLOSS), and corridors are better than no connection.

Wildlife conservation efforts in urban areas must proceed based on available scientific guidance, often inadequate in providing explicit directions to planners (Adams and Leedy 1987, 1991). Although they are not a substitute for detailed information on a particular species of concern, Shafer (1994) nevertheless proposed some updated graphic nature reserve design guidance (Fig. 4). These guidelines are a mix of very broad ideas, but are real-world oriented. Most are also germane to densely populated regions.

There may be no single answer about what a minimum viable population is for a species, and hence there is no consensus on the best reserve design (Nunney and Campbell 1993). McCoy (1983) concluded that minimum area and best-choice options for remaining habitat patches can only be determined from detailed information about species natural history and the patches themselves, not from simple species-area equation calculations. Since local extinction of fragmented populations is common, an understanding of a particular species' dispersal characteristics is essential if the most optimal patches for future reserves are to be sought from the pool of remaining patches (Fahrig and Merriam 1994). I think Wright (1990) correctly indicated that general reserve design guidelines

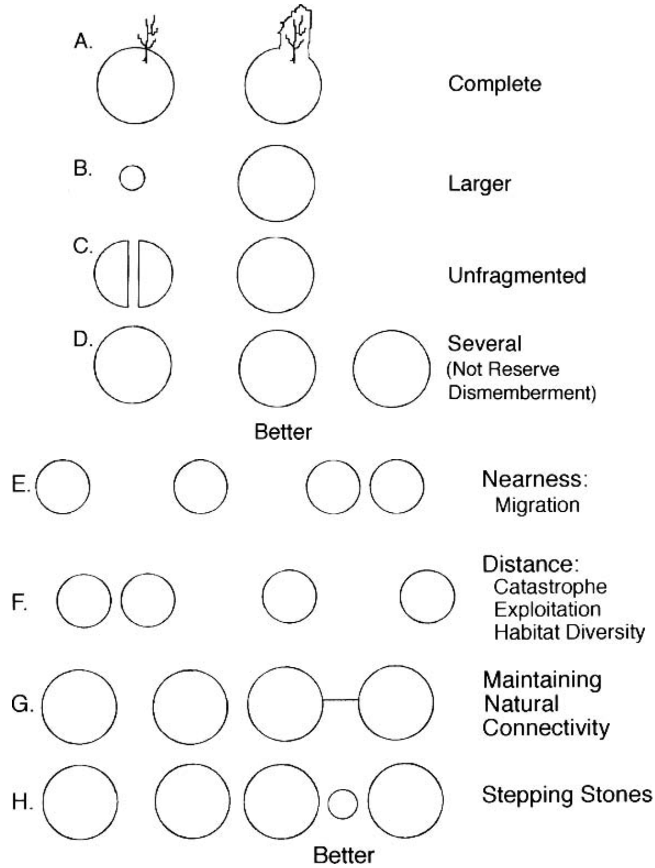

Fig. 4 The option on the right is proposed as better than the one on the left. See text for discussion.
Source: From Shafer 1994, reprinted with permission of Elsevier Science, Amsterdam

- **A.** Complete watersheds, migratory routes, feeding grounds are preferable inside reserves.
- **B.** Larger is better than smaller, especially for wide-ranging large mammals.
- **C.** Unfragmented is better than fragmented.
- **D.** Several reserves (e.g., two reserves, each $1,000\,\mathrm{km}^2$, instead of one $1,000\,\mathrm{km}^2$ reserve) are better because replication guards against catastrophe and human exploitation, and may capture more endemic or patchily distributed species. This is not a recommendation for reserve dismemberment.
- **E.** Nearness is better than being father apart because it facilitates migration to a sister reserve, providing the landscape is traversable by the species.
- **F.** A greater distance may be better, however, to reduce the effects of catastrophe, disease, and human exploitation; increase the likelihood of more habitat heterogeneity and thus more species; and enhance the possibility of more intraspecific genetic variation.
- **G.** Maintaining existing natural connectivity/usable corridors is a far better alternative than no connection.
- **H.** Small stepping-stone reserves, if used, are better than none at all.

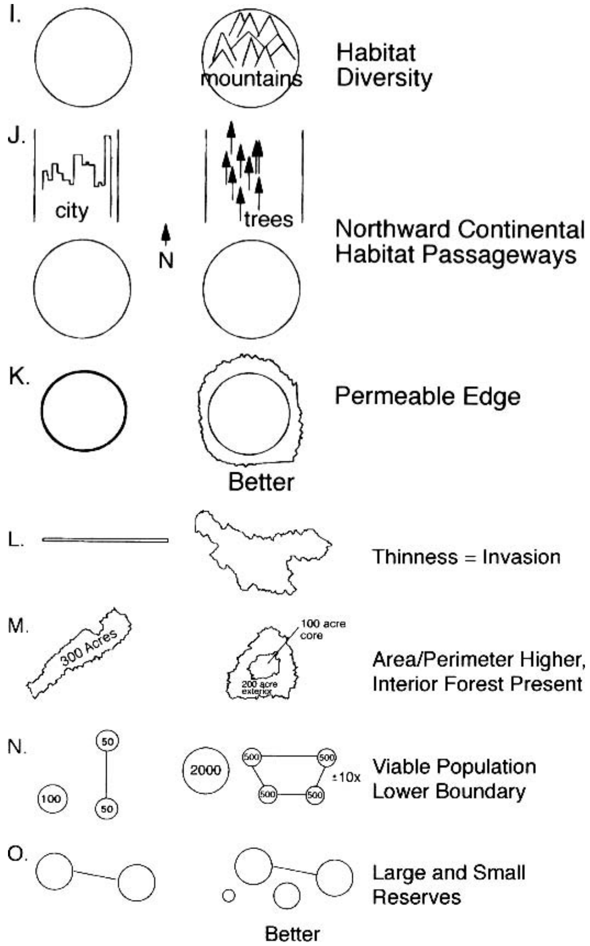

Fig. 4 (continued)

- **I.** Higher within-habitat heterogeneity (e.g., mountains, lakes) should allow for more species due to more habitat. The elevational diversity that mountains provide is also helpful in thwarting climate change.
- **J.** Continental habitat passageways may be vital for overcoming climate change, especially toward higher latitude.
- **K.** A permeable edge encourages animal movement across a park boundary. Edge abruptness, width, vertical structure, and natural discontinuities influence permeability (Forman and Moore 1992).
- **L.** Very thin reserves (e.g., roughly 200–500 m) can encourage invasion by avian predators or weedy species.
- **M.** Similarly, reserves with no or too little core interior forest may lack area required for some U.S. neotropical migrant birds. Theory and some modeling also suggest higher area-perimeter ratios may be better for buffering some external influences and facilitating animal movement across boundaries.
- **N.** A thoughtful guess at the lower limit for minimum viable population size (Soulè 1987b) is better than assumed nonviable ones. However, be forewarned that “there is no single ‘magic number’ that has universal validity” (Soulè 1987a). This is not a recommendation for reserve dismemberment but for reserve connection as needed. Note: some (e.g., Lande 1995) would argue this lower limit should be upped tenfold.
- **O.** Small reserves can provide a useful purpose for some species in any reserve system.

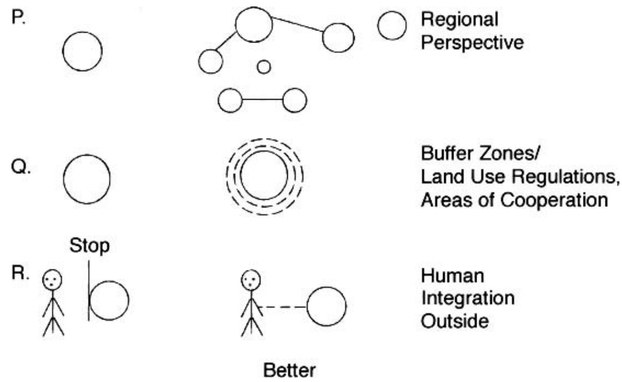

Fig. 4 (continued)

P. A regional perspective rather than a local perspective is crucial in preserving reserve biota.

Q. Buffer zones/land use regulations or areas of cooperative planning outside reserve boundaries are crucial to minimizing human impacts.

R. Human population outside the reserve must be socially and economically integrated into a reserve management plan so the park boundaries are not viewed as an abrupt wall or the park considered an island. Application of social sciences, e.g., Q and R, not just application of ecological theory, is among the foremost nature reserve planning challenges.

derived from model populations are potentially useful only when detailed information on a species' requirements are unavailable.

Bennett (1990) concludes that forest fragmentation in southwestern Victoria, Australia, leads to four recommendations for mammalian conservation: a regional perspective, maintenance of substantial total area of forest, maintenance and enhancement of forest continuity, and protection and promotion of faunal habitat. The broad generalities of needing large tracts of habitat, avoiding fragmentation, and maintaining connectivity were later made by Shafer (1990) and Wilcove and Murphy (1991). These ideas are indeed simple (see Soberon 1992) but their adoption could have a profound positive effect.

Other Realities

Kelly and Rotenberry (1993, 85) said, "In regions that are undergoing rapid urbanization, such as much of Southern California, the question of preplanning the establishment of reserves of sufficient size and configuration to maintain population or community viability is often moot because of high land values and the extent of pre-existing habitat fragmentation." They nevertheless argue for a scientific framework for buffer zone establishment; otherwise the reserves will be "gradually eroded away by external forces." Vast reserves in some parts of California may now be precluded, but networks of smaller ones are not.

San Diego County has initiated one of the most detailed nature reserve design planning efforts in the country, which could serve as a template elsewhere (City of San Diego 1995, Boucher 1995). Gap analysis has been applied to a 235,387 ha area in the southwest portion of the county, identifying core reserves, linkages, and available buffer zones. The analysis involved 15 layers of GIS design information (including land ownership and projected land use) and four proposed reserve design options (Stallcup, personal communication). Observers await its potential adoption in an extremely urbanized and fast-growing region of the country. Reid and Murphy (1995) and Manson (1994) discuss similar efforts in other California counties.

An ongoing effort in Wisconsin, led by The Nature Conservancy, is a good example of reserve planning for multiple small nature reserves within a specific regional landscape. The landscape design for Baraboo Hills, Wisconsin, consists of a plethora of small fragments and buffer zones, connected by other habitat or corridors. The results of this planning effort, and others like it, is not yet known.

Falk (1992, 398) said "The daily practice of conservation is as different from the world of theory and scholarly research as is the blackboard at a military academy from the battlefield. As every conservationist knows, decisions in the field are as likely to be influenced by real-estate transactions, land use, the economics of resource extraction, state and federal taxation, political expediency and the vagaries of public opinion as they are by careful planning grounded in sound conservation biology." The economic, political, and legal considerations gain importance when one is looking at regional landscapes (Shafer 1994).

The issue is therefore not just one of science. The social, economic, and political circumstances must be dealt with (Frankel 1974) or scientific guidance may become irrelevant (Shafer 1990, 1994, in press). Many other factors come into play, such as funding, accessibility, land cost, level of protection, owner attitude, adjacent landowner sympathy toward regional planning, and proximity to large urban areas. Goldsmith (1991) indicated reserve selection criteria for London includes public access, aesthetic appeal, proximity to urban communities, and degree of open-space deficiency.

Role of Nonreserves

Seminatural areas, as we have seen, can contribute to the conservation of biological diversity. In the San Diego County plan, more than just strict nature reserves are being sought. Nearly three-quarters of the remaining coastal sage scrub is on private land. Planners are treating the entire existing landscape as a de facto reserve system. Besides the buffer zones and corridors essential for the strict reserves, homes on large lots can also be a asset (Reid and Murphy 1995). Elsewhere in the United States, some urban parks, county fishing reservoirs, certain resorts, and even golf courses might be assets. We need to think of habitat wherever it occurs, not just in terms of reserves. The agricultural landscape has often been proposed as an alternative approach for biological diversity conservation (e.g., Green 1989), adopted in much of Europe, in part by default. Far more species exist in agricultural/forestry and other human-dominated ecosystems, covering 95% of the terrestrial environment, than in protected reserves (Pimental et al. 1992). In British urban areas, railway embankments, ancient forts, old quarries, gardens, and "wasteland" can provide habitat for some species (Gilbert 1991). In eastern Denmark, a primary function of small *biotopes* is to mark the boundaries of fields and estates. These boundary markers also provide habitat, and permission is required from authorities to disturb some of them. A gross categorization of biotopes include hedges, roadside verges, drainage ditches, small brooks, bogs, marl pits, natural ponds, thickets, and prehistoric barrows (Agger and Brandt 1988).

Insects Are Important

Insect ecology was once studied with the sole aim of potential control (e.g., Price 1984), but a new subdiscipline (i.e., conservation biology of insects) has emerged (Samways 1994). Insect conservation does matter (e.g., Kim 1993), in large part because they constitute 55% of the world's 1.4 million named organisms (Wilson 1992). Insects have been called the "little things that run the world" (Wilson 1987, 344) because of the vital role they play in ecosystem function.

Thomas and Morris (1995) think terrestrial invertebrate extinction rates in Britain have matched or exceeded those of vertebrates or vascular plants during the present century. There are success stories: Thomas (1991) provides accounts of endangered British butterfly populations recovering on small, isolated reserves with proper management. However, the rate of butterfly extinction in Europe is depressing in spite of valiant efforts (Warren 1992).

Looking again at California, the El Segundo blue butterfly *(Euphilotes battoides allyni)* resides on less than 1% of its original geographic range, which once extended about 36 miles along the shore of Santa Monica Bay, from Marine Del Ray to San Pedro. Its survival is closely tied with the Seacliff buckwheat plant. As of 1986, it survived on only two dune remnants—122 ha and 0.6 ha. Chevron, the owner of the 0.6 ha remnant, fenced off this area in 1975 and made it a butterfly sanctuary. Weed removal and outplanting of buckwheat on Chevron's remnant dune habitat occurred in 1983, 1984, and 1986. From 1977 to 1984, the estimated population declined by 70%, but 1985 and 1986 censuses indicated a slowing of this trend. Assuming this was a cause-effect relationship, a recommendation was made to continue outplanting buckwheat and clearing weeds. This was all for the perpetuation of a very small (0.6 ha) but critical piece of habitat for an endangered butterfly species (Arnold and Goins 1987). For their examples in California and elsewhere, see Beatley (1994).

Insect conservation can benefit other species. Launer and Murphy (1994) showed that if all central California serpentine grassland fragments containing the bay checkerspot butterfly *(Euphydryas editha bayensis)* were set aside, about 98% of the native spring flowering plants would receive some protection. The largest number of invertebrate extinctions and candidates for federal listing in the United States is in Hawaii and California (Hafernik 1992). California and Hawaii have the highest number of imperiled species (i.e., 600 or more) in the United States (Stein 1996). Hawaii is, however, our country's extinction capital—two-thirds of all extinct plants and animals come from this one state (Vitousek et al. 1987).

Remnant Restoration

Habitat restoration around habitat fragments can sometimes be accomplished by just allowing natural revegetation to proceed. For example, New York's Onondaga County had only 8% of its area in forest islands in 1930. However, forest cover increased by 40% by 1980, surrounding many older forest islands with younger 50-year-old trees (Nyland et al. 1986). Mladenoff et al. (1994, 752) proposed a design model for harvested Wisconsin forest that incorporates an "old-growth restoration zone surrounding old-growth patches to buffer and enhance forest-interior habitat and link nearby old-growth remnants. A larger secondary zone is delineated for uneven-aged forest management." The important point is that there may be opportunities to enhance remnant viability by allowing succession to proceed outside its boundaries and thereby expand its effective size or connect it to another fragment.

At a much larger scale, Alverson et al. (1994) proposed the creation of *diversity maintenance areas* (DMAs). The center of these idealized DMAs would consist of old-growth remnants. These centers would provide the building blocks for regeneration of large unfragmented tracts of late successional forests adjacent to them. The overall goal is to create mature, wild, old-growth forests and their natural ecological processes and disturbance regimes. The focus of Alverson and colleagues is primarily U.S. Forest Service lands, especially in Wisconsin, but they indicate the idea is applicable elsewhere.

True restoration can take a long time (e.g., hundreds to thousands of years). Tropical lowland dry forest can take 150 years to recover from timber harvest, and tropical lowland wet forest can take 1,000 years (Opler et al. 1977, cited in Reid and Miller 1989). Young successional forest inside or outside a fragment may increase the viability of a fragment in a much shorter time.

Private Initiative

Establishing vast reserves often requires the help of governments, but perhaps less so for smaller parcels. Prime (1992, 11) relates a situation in India, one of the world's most densely populated countries. Historically, between Indian villages three types of reserves were common: forest sanctuaries (raksha), dense forest (ghana), and planted single-species sacred groves (e.g., mango). The forest sanctuaries were off limits to the Hindu people, but the dense forest could be used to collect dry wood, forest produce, and a small amount of green timber. These one- to ten-acre tracts were cared for by the village communities, because they depended on it for their livelihood and they had a tradition of respect for nature. The standard of protection reportedly often exceeded that of current huge government-operated reserves. In the United States, Henry David Thoreau in 1859 thought each town should have "a primitive forest of five hundred or a thousand acres where a stick should never be cut for fuel, a common possession forever" (cited in Udall 1963, 173). Many small reserves are protected by a plethora of private organizations in the United States without the help of government (catalogued in The Nature Conservancy 1982). Throughout the world, private initiative in setting aside reserves is impressive (Alderman 1994).

In 1966, the U.S. Congress held hearings on senate bill S. 2282, a sweeping proposal for a nationwide effort at ecological survey and research. It was an early partial vision of what transpired in 1993—the creation of the U.S. National Biological Survey (later National Biological Service, and still later the Biological arm of the U.S. Geological survey). The role of this organization encompasses some of the need identified in 1966. The potential role it can potentially play is large (National Research Council 1993). The first Natural Heritage Program was created in 1974 under the leadership of The Nature Conservancy (TNC), now expanded to all 50 states, Latin America, and Canada, another example of private initiative at its best. Small tracts were often the focus of early TNC inventories, and some current gap analysis efforts also now consider them (Scott et al. 1993).

U.S. Midwest

It can be argued the U.S. Midwest is a success story in setting aside small remnant tracts. One can always bemoan biotic losses, but it is important to look back at the status of the small protected area enterprise here only 50 years ago. Progress was the result of commitment by thousands of individuals. Reserve location here was the result of an early but still very useful form of gap analysis. Reserve design was based primarily on academic and amateur field notes, common sense, and intuition.

Science-based management can accomplish much, but some problems stem from initial reserve size and layout. In highly urbanized regions, options to create new preserves or improve existing ones may be limited. It can be done, however, using sympathetic landowners, easements, or land purchases. Many states and private conservation organizations already have done this and can continue to do so. Monitoring is needed to gauge preservation success (Drayton and Primack 1996).

Ultimately, reserve design hinges on the value society places on preserving small natural areas. In other words, do we prefer a potentially higher quality of life amidst some native species, or a higher standard of living amidst landscape blight? One recent and able attempt to better educate Wisconsin natural resources personnel (Department of Natural Resources 1995) should be extended to other states. Other such attempts have surfaced (e.g., Nigh et al. 1992). State and federal natural resources personnel, from the maintenance staff to the politically appointed senior executives, need some basic information about what biological diversity means if they are to be effective land stewards in the next century.

An oversimplification of Leopold's (1949) famous land ethic is that humans should treat the outdoors as they would their home. If only this simple but profound idea were adopted as a personal and national principle by the general public, top officials, and politicians. Until then, only proactive foresight will help retain more species and biotic communities in highly impacted landscapes, sometimes their only remaining potential sanctuary.

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Restoration of Fragmented Landscapes for the Conservation of Birds: A General Framework and Specific Recommendations for Urbanizing Landscapes

John M. Marzluff, Kern Ewing

Abstract Humans fragment landscapes to the detriment of wildlife. We review why fragmentation is detrimental to wildlife (especially birds), review the effects of urbanization on birds inhabiting nearby native habitats, suggest how restoration ecologists can minimize these effects, and discuss future research needs. We emphasize the importance of individual fitness to determining community composition. This means that reproduction, survivorship, and dispersal (not simply community composition) must be maintained, restored, and monitored. We suggest that the severity of the effects of fragmentation are determined by (1) the natural disturbance regime, (2) the similarity of the anthropogenic matrix to the natural matrix, and (3) the persistence of the anthropogenic change. As a result, urbanization is likely to produce greater effects of fragmentation than either agriculture or timber harvest. Restoration ecologists, land managers, and urban planners can help maintain native birds in fragmented landscapes by a combination of short- and long-term actions designed to restore ecological function (not just shape and structure) to fragments, including: (1) maintaining native vegetation, deadwood, and other nesting structures in the fragment, (2) managing the landscape surrounding the fragment (matrix), not just the fragment, (3) making the matrix more like the native habitat fragments, (4) increasing the foliage height diversity within fragments, (5) designing buffers that reduce penetration of undesirable agents from the matrix, (6) recognizing that human activity is not compatible with interior conditions, (7) actively managing mammal populations in fragments, (8) discouraging open lawn on public and private property, (9) providing statutory recognition of the value of complexes of small wetlands, (10) integrating urban parks into the native habitat system, (11) anticipating urbanization and seeking creative ways to increase native habitat and manage it collectively, (12) reducing the growing effects of urbanization on once remote natural areas, (13) realizing that fragments may be best suited to conserve only a few species, (14) developing monitoring programs that measure fitness, and (15) developing a new educational paradigm.

Keywords: Birds · conservation · exotic species · fragmentation · landscape · natural disturbance · predation · urban ecosystems · restoration

Introduction

When humans occupy landscapes, we convert portions of the native vegetation to agriculture and urban development, and modify areas to varying degrees by harvesting natural resources (Villa et al. 1992). The ecological implications of our domination of the earth are complex (Vitousek

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et al. 1997), but the effects on land cover are rather straightforward: we *fragment* it. Once continuous mosaics of native vegetation become transformed into disjunct pieces of native vegetation surrounded by a matrix of cement, grass, crops, and degraded lands (Meyer & Turner 1992; Marzluff & Hamel 2001).

Fragmentation of natural landscapes is often detrimental to biodiversity because it involves the removal, reduction, and isolation of native vegetation (Fahrig 1999). As a result, remaining populations of native wildlife are smaller and perhaps exposed to new threats emanating from the human-dominated matrix. Relevant threats for bird populations include (1) competition with exotics such as *Sturnus vulgaris* (European starlings; Kerpez & Smith 1990), (2) exposure to larger populations of predators and parasites such as corvids, domestic cats, and *Molothrus ater* (brown-headed cowbirds; Robinson & Wilcove 1994; Marzluff & Restani 1999), (3) heightened disturbance and mortality from human activity (Johnston & Haines 1958; Knight & Gutzweiller 1995; Evans 1998), and (4) restricted and exposed dispersal corridors (Matthysen & Currie 1996).

The anthropogenic activities that fragment natural landscapes do not affect wildlife populations equally (Fig. 1). Globally, agriculture currently has the greatest effect on wildlife because 32% of the earth's land is planted in row crops or pastures (Houghton 1994). In many regions, agricultural practices are intensifying, which reduces similarity of land cover with native cover, introduces exotics, disrupts nutrient cycles, and adds pollutants (Newton 1998). Urbanization has the greatest local effect on wildlife because of its persistence on the landscape and its dissimilarity to natural land cover. The magnitude of the effect of urbanization (loosely defined here to include human settlement ranging from dispersed rural and exurban villages and homesteads to densely settled subdivisions and cities) depends on the amount of vegetation incorporated in settlements, especially native vegetation (Lancaster & Rees 1979; Beissinger & Osborne 1982; Mills et al. 1989). The pattern of development (clumped versus dispersed housing, for example) greatly affects the resulting fragmentation, but the effects of urban pattern on bird diversity are poorly known (Nilon et al. 1995). Timber harvest produces the least effect of fragmentation because harvested areas may regrow with native vegetation. However, when succession is slow, non-native vegetation is planted, natural structures are simplified, or harvest is done in landscapes not naturally fragmented, the effects of timber harvest can be substantial (Bierregaard & Lovejoy 1989). In general, the causal link between fragmentation and bird reproduction (increased nest predation and parasitism along newly formed edges) is greater when landscapes are fragmented by urban and agricultural development than when they are fragmented by forestry (Marzluff & Restani 1999).

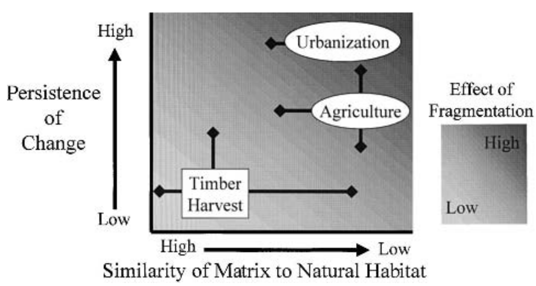

Fig. 1 Relative importance of anthropogenic agents of fragmentation. Urbanization, agriculture, and timber harvest all fragment native habitats, but urbanization is expected to have the greatest local effect on native animals because once an area is urbanized it rarely reverts to a more natural condition (high persistence of change) and the urban matrix is very dissimilar to native land covers. The effect of urbanization may vary depending on the mix of built, exotic, and native elements in the matrix (range of effect is indicated by line ending in diamond). Agriculture is expected to have intermediate effects on native animals that vary depending on the intensity of land conversion and use (range indicated by lines to diamonds). Timber harvest is expected to have the least effect on native animals, but this depends on the natural rate of success and tree growth which affect the persistence of the change, and the natural disturbance regime of the area which affects the similarity between harvested and natural land cover (range indicated by lines to diamonds)

Urbanization is likely to top agriculture as the dominant agent of fragmentation at the global scale. Currently only about 3% of the earth's surface is covered with buildings and other urban structures (Meyer & Turner 1992); however, the growing human population is becoming increasingly urbanized. By 2025 the global urban population is projected to equal today's world population (~6 billion, United Nations 1996). Therefore, the extent of urban development will increase. More importantly, the sprawl associated with many urban centers and the current tendency in developed countries to subdivide and settle formerly extensive ranches and wildlands (Berry 1990; Knight 1995; Buechner & Sauvajot 1996) means that increasingly large portions of the earth are fragmented by some form of human settlement. This is most evident when the pattern of lights produced by human settlement is viewed from afar at night (Marzluff & Hamel 2001). Because of the increasingly extensive, lasting, and large effects of fragmentation resulting from human settlement, and the typical emphasis in the literature on fragmentation resulting from other human activities (usually forestry; e.g., Harris 1984; DeGraaf & Miller 1996; Laurance & Bierregaard 1997; Rochelle et al. 1999), we restrict the remainder of our discussion to restoration of areas fragmented by urban development.

It is unreasonable to assume that habitat fragmentation will subside as long as humans dominate the earth. Therefore, restoration ecologists must determine how best to maintain wildlife diversity in fragmented landscapes. Nearly all suggestions in the conservation biology literature posit two solutions for preserving biodiversity in fragmented settings: (1) establish corridors among native patches or (2) buffer native patches with native habitat to increase their size and amount of interior area (Davis & Glick 1978; Soulé 1991; Shafer 1997). Both goals may be realized by first restoring fragments nearest reserves (Huxel & Hastings 1999), but we suggest that additional options exist and explore some of them that are relevant to birds in urban landscapes. Our objectives are to (1) offer a framework that links individual animal fitness to community composition so that we can better identify when and why fragmentation affects avian community composition; (2) highlight and review the large effects urbanization has on birds inhabiting remaining native habitats; (3) suggest how restoration ecologists, land managers, and urban planners can reduce the impacts of fragmentation in urbanized landscapes; and (4) suggest research that is needed to improve our ability to restore ecological function to urban fragments.

Maintaining Diverse Communities Requires an Understanding of Individual Fitness and Population Viability

Restoring wildlife diversity in fragmented environments will be more successful if restoration ecologists, land managers, and urban planners know how their actions affect the fitness (reproduction and survivorship) and dispersal of individual animals. Simply knowing that a management action increases or decreases the diversity of wildlife communities is insufficient (van Horne 1983). Understanding fitness and dispersal of individuals is important because these are the parameters that determine a local population's likelihood of growth or extinction and its dependency on (or importance to) other populations. Managers who know, for example, that corridors work in sage-scrub habitat because they allow coyotes to limit medium-sized mammals that prey on bird eggs and nestlings (Crooks & Soulé 1999) will be able to handle problems in a different landscape when corridors fail to enhance community diversity. They might see, for example, that a variety of predators use the corridors to the detriment of mammalian predators, as well as birds themselves. A change in corridor configuration or direct removal of some predators might restore bird community diversity. Managers unaware of why avian diversity is declining despite the presence of corridors might guess that the reserve needs to be larger or the corridor wider. These actions would likely be ineffective and possibly exacerbate the problem. In general, managers who know the causal links between their actions and population persistence can more effectively restore and maintain community diversity (Marzluff et al. 2000).

The focus on individuals and populations we espouse is often viewed as simplistic and today is thought to be relevant only for sensitive species requiring special attention (Knight 1990). In contrast, management of communities or ecosystems is perceived to be more balanced, economical, and effective. However, this is an inherently false dichotomy because the ability of local populations to persist in fragments (population viability) *determines* community composition and is *determined by* the ability of individuals of a species to reproduce and survive in fragments and to colonize fragments (Fig. 1). In other words, because communities are collections of species, they require detailed understanding of each species for effective management.

Community composition is unlikely to be a simple reflection of the individualistic responses of species to their environment (Fig. 2). Instead, a wildlife community is determined by the physiology, ecology, morphology and behavior of individual species (Gleason 1926), the constraints of various biological interactions (predation, competition, mutualism, parasitism), ecosystem function, past disturbance, and chance (Wiens 1989). However, the population dynamics of each species in the community is the fundamental determinant of community composition because a population's dynamics reflects the effects of all these factors on survivorship, reproduction, immigration, and emigration. This preeminence of population dynamics challenges the manager to monitor survival, reproduction, and dispersal, not just community diversity.

Appreciating and measuring population-level processes in highly fragmented landscapes often emphasizes the importance of dispersal to population persistence. Moreover, this often requires a regional, rather than local, perspective. For example, community diversity in fragments of midwestern forests may be high simply because of immigration from large forest blocks in the upper Midwest (Robinson et al. 1995). The diversity in fragments cannot be maintained unless large forests continue to be productive (i.e., fragments are sinks and distant forests are sources; Pulliam 1988). The manager of a fragment who monitors only local community diversity will blissfully think he is doing a great job until diversity inexplicably drops. The manager of a fragment who measures reproduction, survival, and dispersal will understand that dispersal is critical to his fragment's diversity and can be working to enhance reproduction and survivorship in the fragment, while also securing populations in distant, large blocks of forest.

Acknowledging the importance of individual fitness and population viability to community composition focuses our discussion of restoration in fragmented environments. To restore avian diversity in fragmented landscapes we need to determine if and how fragmentation affects reproduction,

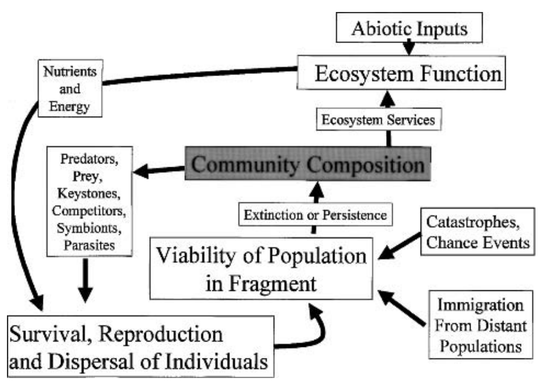

Fig. 2 How individual, population, community, and ecosystem properties interact to determine community composition. Survival, reproduction, and dispersal by individuals determine local population viability. This may be affected by immigration from distant populations and chance events. Population viability determines whether a population persists or goes extinct, and therefore defines community composition. Emergent properties of communities and ecosystems modify community composition through their effects on individual fitness. Thus, the restoration ecologist interested in maintaining diverse communities must restore, maintain, and monitor individual fitness and dispersal

survival, and dispersal of individuals in local populations. Likely pathways for these effects include (1) disruption of nutrient cycling and energy flow, (2) increases in predators, parasites, and competitors, and (3) reduction of immigration from distant populations (Fig. 2). The influence of fragmentation on each of these processes will be highly site-specific, but these are the processes that should determine individual fitness and population viability (Fig. 2). Individual fitness, dispersal, and population dynamics must be monitored to determine if restoration prescriptions are successful (i.e., correct the aberrant processes in a particular site and enhance survivorship, reproduction, and successful dispersal).

When Does Fragmentation Affect Bird Communities?

The effects of anthropogenic habitat fragmentation on fitness, population viability, and therefore community diversity and composition depend on two basic attributes of the landscape. First, the temporal frequency and spatial extent of the natural disturbance regime determines the natural patch dynamics (patch size, connectivity, and persistence) of an area. Patch dynamics are important agents of natural selection that determine the sensitivity of birds to unnatural fragmentation. Small volcanic flows, meandering rivers, localized windstorms, and low-intensity fires naturally fragment landscapes. As a result, birds have evolved good dispersal characteristics and utilize small patches and edges in such settings. Examples of these types of landscapes are active volcanic islands, intermountain forests of the western United States (Sallabanks et al. 1999), bottomland forests of the southeastern United States (Walters 1998), and natural wetland complexes. In these areas, bird (and other wildlife) diversity is reliant on fragmentation, not sensitive to it. Fragmentation provides a mix of small, short-lived habitat patches, each of which is used by distinct assemblages of birds. Removing the agent of fragmentation homogenizes the landscape and lowers bird diversity (Askins 1998). In contrast, large, homogeneous, interconnected patches of vegetation are produced by infrequent disturbances or extensive disturbances such as hurricanes, typhoons, and high-intensity fires that affect entire forest stands. As a result, natural selection favors birds that are sedentary and able to utilize resources provided in the patch. The poor dispersal abilities that have evolved predispose birds inhabiting such landscapes to being affected detrimentally by fragmentation (Soulé et al. 1988; Knick & Rotenberry 1995). Temperate and tropical rainforests, western United States shrublands and thornscrub, and tundra are examples of these landscapes.

The second factor determining the potential effects of anthropogenic fragmentation is the similarity of the land cover created by humans to natural land cover. When we surround remaining habitat fragments with natural vegetation (such as young forest during timber harvest operations) birds often show little response. For example, rates of nest predation (DeGraaf & Angelstam 1993; Marzluff & Restani 1999), nest parasitism (Tewksbury et al. 1998), and community diversity (Sallabanks et al. 1999) have been shown to be unaffected by forest fragmentation resulting from timber harvest. However, when we replace forests, grasslands, and wetlands with crops and urban settlement (land covers grossly different from natural ones) bird communities rapidly lose diversity (Wilcove 1985). The specific reasons for these losses and suggestions for minimizing them are discussed below.

How Does Urbanization Affect Birds in Remaining Fragments of Native Habitat?

Urbanization produces homogenous, dense, and often exotic communities of birds and mammals within settlements worldwide (Erz 1966; Bezzel 1985; Knight 1990; Marzluff et al. 1998; Blair 2001; Marzluff 2001). These communities and the human activity associated with urbanization affect birds that remain in relatively natural fragments by (1) increasing their natural predators and parasites; (2) attracting exotic predators, diseases, and competitors; (3) affecting trophic structures by removing top-level predators; (4) obstructing traditional migratory and dispersal routes and increasing the dangers of dispersal; (5) removing key resources like standing and downed deadwood, ground cover, and shrub patches; and (6) disrupting nutrient and hydrological cycles. We discuss each below with an emphasis on how they affect individual fitness and resulting community composition.

Increase of Native Predators and Parasites

Native species that are able to tolerate human activity benefit by the ameliorated climate, abundant food, and reduced predator populations in cities. Foremost among such species are corvids, small- to medium-sized mammals, brood parasites like brown-headed cowbirds, and a few raptors (Eden 1985; Fraissinet 1989; Marzluff et al. 1994; Bird et al. 1996; Konstantinov 1996; Danielson et al. 1997). These species forage to varying degrees in nearby fragments of native habitat and may even establish large populations in such habitats where they use native vegetation for nesting and exploit nearby urban food sources. The result of their increase is an increase in predation pressure on the eggs, nestlings, and occasionally adults of birds in fragments. These "subsidized predators" often are most abundant at the edge between urban lands and fragments, and therefore they often impact productivity of birds in fragments surrounded by urban development most severely at the fragment edge (Marzluff & Restani 1999). The result is that populations of native birds rarely reproduce well enough to be viable along the edges of fragments or in very small or linear fragments that have high ratios of edge to interior area (Wilcove 1985; Robinson et al. 1995).

Introduction of Exotic Predators, Diseases, and Competitors

Humans often purposefully or accidentally import exotic animals when they colonize new lands. This is a major cause of avian extinction (Diamond 1989) and a serious threat to any native birds in fragments near human settlements. *Felis domesticus* (house cats), *Rattus* spp. (rats), European starlings, and various diseases dramatically reduce reproductive success and survivorship of birds. On oceanic islands they are common agents of extinction. For example, after the ship *Makambe* ran aground on Lord Howe Island in 1918, rats escaped, colonized the island, and caused the extinction of five bird species (Hindwood 1940). In a similar way, introduced cats exterminated the *Cyanoramphus novaezelandiae erythrotis* (Macquire island parakeet; Taylor 1979), and malaria contributed to the extinction of at least four bird species on Hawaii (van Riper et al. 1986). On habitat islands the role of these agents is less well documented, but likely to be the same. Cats alone were estimated to kill 7.8–217 million birds per year in Wisconsin (Coleman & Temple 1996)! Cat populations are greatest in rural settings, but still substantial in urban areas (e.g., 1.5 million cats live in urban Illinois, compared to an estimated 4 million in rural Illinois; Warner 1985). As with native predators, exotic mammalian predators often inhabit forest-field and forest-development edges (Danielson et al. 1997; Mankin & Warner 1997). Avian communities may persist on habitat islands in spite of exotic predators, but the long-term prospect of persistence may be illusionary. Their persistence may be due only to immigration from less impacted "source" populations (Pulliam 1988; Robinson et al. 1997). Detailed study of reproduction, survival, and dispersal is the only way to determine this and identify effective management solutions.

Removal of Top-level Predators

While we supplement many predators, we remove many others, especially if they are viewed as a threat to our safety or that of our stock. Thus, at this time lions, tigers, bears, and wolves inhabit few urban areas. We remove other top predators in more indirect ways by interfering with their hunting or providing too little space for their needs. In this way, many large raptors, weasels, native cats, and canids are excluded. The result of predator removal is an increase in corvids, domestic dogs and cats, rats, and mice that then prey on native birds and their nest contents. The keystone role of predators was demonstrated in the coastal California scrub, where avian diversity dropped when *Canis latrans* (coyotes) were excluded from the landscape (Soulé et al. 1988; Crooks & Soulé 1999).

Installation of Dispersal Barriers

Dispersal is the aspect of avian population biology that we know the least about, but one we are certainly affecting with our building and development activities (Walters 1998). Roads, lights, buildings, and subdivisions do not provide the cover or food resources that birds need when they disperse from their natal habitat. Moreover, they include many obstacles capable of killing thousands of birds annually (Johnston & Haines 1958; Evans 1998). Tall (> 200 ft) towers alone are estimated to kill 2–4 million songbirds each year in North America (Evans 1998). Untold numbers are hit each year by automobiles. Interference with dispersal reduces community diversity because it lowers colonization of fragments and reduces the ability of healthy source populations to contribute breeders to distant habitat. Because of the long lifespans of many birds, the effects of reduced dispersal will often take decades to detect unless it is explicitly monitored.

Removal of Key Resources

Many humans abhor untidy landscapes. We remove and simplify ground cover, trim branches from lower reaches of trees, cut down dead trees, and fastidiously rake, haul, burn, or grind up fallen dead material. The result is a sterile landscape lacking nesting cover, cavity nest resources, and diverse insect, reptile, and amphibian communities that many birds depend on. This is an important reason that ground, shrub, and cavity-nesting birds are the first to disappear from human-dominated ecosystems (Emlen 1974; Gavareski 1976; Horn 1985; Tilghman 1987; Konstantinov 1996).

Disruption of Ecosystem Processes

Perhaps the greatest long-term effect of human domination on birds is our disruption of nutrient and water cycles, and diversion of primary productivity (Marzluff et al. 1998). Humans are estimated to use 25% of the earth's primary productivity (Vitousek et al. 1986). Thus, less is available to other species. Our tendency to suppress fires and overuse water resources in proximity to dwellings changes bird habitat and reduces nutrient loads. Such degradation of the landscape extends far beyond the boundaries of cities (Wackernagel & Rees 1996), which may reduce reproduction and survivorship of birds in nearby native habitats.

Management of Native Fragments in Urbanized and Urbanizing Landscapes

Conserving biodiversity in urban landscapes requires two basic activities: (1) the design and establishment of a system of native vegetation reserves, and (2) the maintenance and restoration of ecological function in reserves. The first activity dominates the manager's agenda in landscapes where sprawl is just beginning to encroach on large areas of native habitat. Much has been done to relate the theory of island biogeography to reserve design in such settings (Davis & Glick 1978; Knight 1990;

Soulé 1991; Shafer 1997). These authors uniformly recommend that (1) the area and numbers of reserves be maximized, (2) the amount of edge and degree of fragmentation within reserves be minimized, (3) the connectivity between reserves be maximized, (4) buffers be maintained around reserves, and (5) the scale of reserve planning be expanded beyond the local area to include entire watersheds and bioregions. However, the rapid pace of urbanization, the high cost of land acquisition, the diminishing availability of large parcels of native habitat, and increasing threats to existing reserves reduce the utility of reserve design guidelines to managers in urban settings. Rather than deciding on the optimal properties of fragments, managers are increasingly forced to decide whether to acquire new reserves or restore and manage existing ones (Schwartz & van Mantgem 1997).

In the fragmented landscapes that typify those settled by humans, managers must acquire as much native habitat as they can. Priority should be given to acquiring the largest parcels (Robinson et al. 1997) of native vegetation (Schwartz & van Mantgem 1997) in proximity to existing reserves occupied by sensitive species (Huxel & Hastings 1999). Small fragments of native habitat can conserve biodiversity (Schwartz & van Mantgem 1997), but mobile organisms like birds may require large, undisturbed reserves in some part of a bioregion to allow any fragments to be utilized (Robinson et al. 1995, 1997). Acquisition of native habitat may not be enough to conserve wildlife because reserves attract predators and parasites and are often heavily utilized for recreation by humans. As a result, birds may be attracted to reserves but may not be able to maintain viable populations in them (Robinson et al. 1997). Regional planning and active management and restoration of reserves is necessary. Given this premise, we provide a combination of short- and long-term restoration approaches and policy structures that we suggest will aid the conservation of wildlife diversity in general and avian diversity in specific in human-dominated ecosystems. No one approach is adequate. All the approaches considered together provide the planner, habitat manager, and restoration practitioner with tools for providing an environment that retains native species diversity.

Restoration of Ecological Function in Urban Landscapes

Little has been written about how to actively manage and restore fragments in urbanized settings. Here we offer 15 specific recommendations that would improve the suitability of reserves for birds. Above all, it must be assumed that the degree of fragmentation in the landscape will remain stable or increase. Therefore, the manager must be relentless in determining how individuals are performing (reproducing, surviving, and dispersing) in reserves and aggressive in restoring population viability within reserves. It may be prudent to prioritize the restoration and management of fragments nearest areas occupied by native species of concern (Huxel & Hastings 1999); however, we would caution managers first to make sure that populations in such areas are viable.

Managers have their greatest latitude to restore ecological processes within reserves, but much can also be done to manage the matrix surrounding the reserve. Therefore, we direct our suggestions to restoration within fragments and around fragments. Key elements that should be present in restored fragments include: standing deadwood, complex woody debris, complex vertical and horizontal structure, protected interior areas, undeveloped riparian zones, undeveloped slopes and cliffs, high native plant diversity, invasive plant control, minimal lawn area, high diversity of shrubs that produce berries, nuts, or nectar, control of exotic mammals including house pets, reduced supplementation of native predator and parasite populations, monitoring programs that measure fitness and dispersal, and integrated education, research, and outreach activities to foster citizen support. These are important regardless of fragment geometry and serve to underscore our general message that the ecological processes that produce viable populations, not just the habitat structure associated with viability, must be restored.

(1) Increase the foliage height diversity within fragments. MacArthur and MacArthur (1961) noted long ago that bird community diversity was positively related to foliage height diversity. Lancaster and Rees (1979) confirmed this association in native fragments imbedded in an urban landscape. This occurs because habitats with many vegetation layers (herb, shrub, sapling, canopy) provide more nesting, feeding, and hiding spots than habitats of uniform structure. Providing dense and variable ground and shrub cover is especially important in human-dominated ecosystems. These are the areas we tend to clean and simplify first, yet they are the areas that harbor native birds most likely to breed successfully in the face of strong nest predation (such sites are usually well hidden from avian nest predators). For example, urban woodlots in Massachusetts that had vertically and horizontally diverse vegetative structure contained richer avifaunas than simpler woodlots (Tilghman 1987).

Restoring or maintaining plant structural diversity may be an intermediate management goal. It keeps the manager's options open because a greater diversity of birds may find some portion of the reserve attractive. However, later the manager may want to emphasize and increase specific structural attributes needed by birds able to maintain viable populations in urban fragments (see [14] below). Structural diversity may also be a useful criterion for prioritizing parcels available for purchase.

(2) Maintain native vegetation and deadwood in the fragment. Structural diversity in vegetation may promote bird diversity, but simply maintaining the structure of the fragment is not adequate. Exotic vegetation must be actively controlled and native plants maintained and restored if native birds are to be retained in the fragment. Native vegetation has been found to be important to bird diversity in ecotypes ranging from deserts (Mills et al. 1989) to riparian zones (Rottenborn 1999) to deciduous forests (Beissinger & Osborne 1982).

When restoring small parcels, aim for diversity in plantings, in vertical structure, and in downed wood. Diverse wood on the ground provides high diversity of habitats for decomposers, invertebrates, and small mammals—all-important foods for birds (Wood 1989; Schuman & Belden 1991). Restoration must allow for continued recruitment of standing dead- and downed wood. This means that trees must be allowed to age, lose branches and upper portions, and become infected with organisms that decay their interiors (e.g., heart-rot fungi like *Phelinus pini* and *Fomitopsis cajanderi*). Nest boxes may be useful in the short term for secondary cavity nesters, but many cavity nesters also require natural foraging and nesting substrates provided only by dead-wood. A variety of active management techniques exists to speed the development of cavity resources and foraging sites needed by primary and secondary cavity nesters. These include topping large trees, killing trees, and injecting heart-rot fungi (Conner et al. 1981; McComb & Rumsey 1983; Bull & Partridge 1986; Baker et al. 1996).

(3) Manage the landscape surrounding the fragment (matrix), not just the fragment. In urban settings the primary reason that birds in native fragments fare poorly is that predators, parasites, competitors, chemicals, kids with BB guns, recreationists, and the like *from the matrix* intrude into the fragment. No amount of habitat restoration within the fragment will be adequate if the destructive forces in the surrounding landscape are not identified and reduced (Aronson et al. 1993). In most settings regulation, enforcement, education, and a variety of buffers and barriers (see [4] and [5], below) will be needed to accomplish this. A key issue to reduce in the matrix is food supplementation to exotic and native nest predators.

(4) Design buffers that reduce penetration of undesirable agents from the matrix. The standard approach to providing interior conditions is to design reserves to include "interior areas" (those more than 50–200 m from an edge: Soulé 1991; Shafer 1997; Rochelle et al. 1999). However, simple spatial buffers do not guarantee the safety of fragment interiors in urban areas because (1) native predators and parasites that live in fragment interiors are supplemented by resources in the matrix, and (2) many of the exotic predators, competitors, pollutants, and other agents we try to contain in the matrix are very mobile and able to penetrate buffers, especially if the composition of the buffer is suitable to them. Often, native habitat is suitable, if not optimal, to predators, parasites, and competitors (Marzluff et al. 2000). To prevent the problem of buffers acting as "wicks" that allow exotic and native predators and parasites to flow between the fragment and the matrix, we suggest buffers be as impermeable as possible. Often this means as unattractive to, and as devoid of, any wildlife as possible. Extremely dense, simple structured forest works well in this regard in the temperate rainforests of the Pacific Northwest (Marzluff et al. 2000). Heavily urbanized or intensively farmed areas would also insulate fragments from biotic invaders. If such buffers are proposed one should make sure that wildlife does not utilize these areas, and that polluted emissions and runoff, and noise are minimized.

(5) Recognize that human activity is not compatible with interior conditions. Interior (core) areas must be protected from humans as well as predators, parasites, competitors, and diseases. Fencing reserves, or making them difficult to enter in other ways, may help, but in multiple-use reserves, recreational use of interiors and buffers must also be limited. Buffers that include trails, for example, may foster the travel of disruptive agents from the matrix into the fragment. Even innocuous activities like nature watching need to be excluded from core areas because they can disrupt breeding birds (Knight & Gutzweiller 1995).

(6) Make the matrix more like the native habitat fragments. In the application of island biogeography theory to reserve design, the size and closeness of islands become less and less important determinants of animal diversity as the matrix becomes more similar to the habitat islands. The reason for this is that the matrix interferes with normal movements between fragments in proportion to its dissimilarity with the habitat preferences of animals. Increasing the similarity between the fragments and the matrix can be accomplished by leaving or creating many small areas of native habitat in the matrix, with plant and structural complexity and protected in some way from intrusion (Berger 1993). In the city these habitat areas can be created at street ends, along streams, in parks, on slopes, between rail and street corridors, on municipal property such as at treatment plants, along shores, around wetlands, on airports, or on islands. Private landowners with large lots can contribute to this effort by planting small portions of their land with native vegetation. Urban planners and managers could create stepping stones between reserves by promoting naturalization of the matrix in strategic locations. This may help retain biodiversity in reserves (Shafer 1997), provided that such remnants do not act as sinks. Special attention should be paid to naturalizing and protecting naturally linear areas like riparian zones that may be normal dispersal corridors (Mankin & Warner 1997).

(7) Actively manage mammal populations in fragments. When humans remove medium- to large-sized predators from ecosystems, small and medium-sized mammal populations often increase (Soulé et al. 1988). In urban areas many of these mammals are exotics that are efficient bird predators (Crooks and Soulé 1999). The house cat is the single most important mammal in this respect. A single cat can wipe out populations of entire species within fragments. An important component of habitat restoration includes restoration of a balanced small- to medium-sized native mammal community. This will require trapping and removing all exotic mammals, educating and regulating surrounding landowners to keep their cats indoors, and intensive monitoring of mammal activity within fragments.

Avian nest predators and parasites (e.g., corvids, brown-headed cowbirds) may need to be controlled in special instances (e.g., to protect endangered species like *Dendroica kirtlandii* [Kirtland's warblers] Marshall et al. 1998), but we suggest that reducing food supplementation in the matrix and designing effective buffers are more permanent solutions to these problems.

(8) Discourage open lawns on public and private property. The typical American lawn is an ecological disaster that reduces biodiversity, contributes to global warming, stresses water supplies, uses global fossil fuels, and encourages the use of pesticides and herbicides (Bormann et al. 1993). The organic soil, litter, woody debris, herbaceous layer, shrub layer, and tree saplings excluded from lawns provide a complex habitat substrate for microbes, invertebrates, small mammals, and birds. Urban planners and managers should discourage open lawns on public property and reward lawn minimization on private property. Encourage landowners to reduce lawn and diversify their ground and shrub cover by providing information on the importance of these resources, creating backyard wildlife refuge programs, and providing property tax incentives. Residents of King County, Washington, for example, receive substantial tax breaks (50–90% reductions in property taxes) for maintaining natural undisturbed areas on their properties (King County Public Benefit Rating System).

(9) Provide statutory recognition of the value of complexes of small wetlands. Complexes of small wetlands may be especially important in urban settings (Adams 1994). Because they are considered to be of little consequence individually, the small wetlands may be subjected to piecemeal loss from land development or its consequences. Smaller wetlands tend to fall by the wayside because they may not be inventoried by any regulating agency; they may be unreported or below the threshold size for regulation (10–20 ha). In addition, landowners may simply not recognize them as wetlands when development or site modifications are contemplated. However, small complexes have many positive values such as extensive aquatic-terrestrial interfaces, maximization of productive shallow, saturated areas, complicated canopy and edge structure, aggregate water storage capability, and seclusion. They may promote dense nesting colonies of birds (Weller 1994), and, when clustered, they provide nearby destinations when waterfowl are flushed. They add to landscape diversity (Adams 1994).

(10) Integrate urban parks into the native habitat reserve system. Parks have multiple users, but it must be recognized that multiple uses cannot be carried out on every square foot of a park without severely degrading the habitat quality for wildlife. We suggest that parks with good habitat or adjacent to good habitat should have an emphasis on wildlife values, and sites with poor habitat potential should be used for more intensive human recreational sites (e.g., ball fields, picnic areas). Trails can be built through or adjacent to wildlife areas, but the integrity of such areas should be kept in mind when park design is accomplished (see [5] above).

(11) Anticipate urbanization and seek creative ways to increase native habitat and manage it collectively. Increasing reserve systems in urban landscapes will be difficult but necessary to maintain biodiversity. Two promising ways to accomplish this task are: (1) creation of public-private partnerships to reserve substantial amounts of native land cover (e.g., in Wisconsin and Illinois; Herkert 1997; Stearns & Matthiae 1997), and (2) establishment of "mitigation banks" where developers mitigate habitat losses by contributing to a fund for purchasing available open space (Soulé 1991).

Incentives for participants and coordinated management of individual parcels at local, regional, and national levels are necessary to reduce redundancy and conserve the greatest possible share of species. An example from an urbanizing landscape of utilizing the inevitable fragmentation in a land use system driven by economics is the Lower Rio Grande Valley National Wildlife Refuge (NWR) in south Texas (Jahrsdoerfer & Leslie 1988). Two self-contained refuges exist in the area around the mouth of the Rio Grande: The Santa Ana and Laguna Atascosa NWRs. The Lower Rio Grande Valley NWR was formed primarily to manage fragmented parcels, and currently contains almost 100,000 acres, stretching from the last dam on the Rio Grande to its mouth (about 150 miles). Parcels with intact thornscrub are purchased when possible, but parcels are also purchased from farmers who can no longer profitably farm them (because of salt or lack of water). Farmers are paid to continue to farm while converting 10 to 20% a year back to thornscrub. The farmers are paid to prepare and plant the restoration sites. The vegetation is quick growing and the corridor of parcels provides a matrix of satisfactory habitat dispersed among the farmland. Bird watching has become a major economic industry in south Texas, with the legislature allocating funds this year for a primary birding center with three satellite centers.

This example points out the importance of restoring native habitats and preserving them before extensive urban development (and the associated increase in property value) occurs. Since inception of the habitat buy-back plan, the lower Rio Grande has become the third-fastest growing area in Texas, and two urban centers (McAllen and Brownsville) now contain over 845,000 people (Texas State Data Center, Texas A&M University). Urbanization is certain to increase because seven new international bridges are proposed along the area's 50-mile section of the Rio Grande. As property values increase, farmers are likely to benefit more from subdividing for development than from restoring thornscrub (Wuerthner 1994). Successful expansion of the reserve area will become increasingly difficult as urbanization proceeds. Conservationists should anticipate urbanization in currently rural areas and create public-private partnerships to reserve and restore native land cover while property value is low. Using incentives for participants and coordinating reserve management in urban landscapes, where development is always profitable, will be more difficult but creative managers and planners can find opportunities to purchase strategic properties and encourage nearby landowners to restore a portion of their land.

(12) Reduce the growing effects of urbanization on once remote natural areas. Large areas of native habitat on the fringes of development are likely to be important sources of colonists for fragmented habitat. Their preservation is crucial. The manager interested in maintaining avian diversity in a small local fragment has much to gain by maintaining large, distant sources (Robinson et al. 1997). Growth management policies, economic incentives to reduce subdivision, mitigation banking to purchase distant open space, and incentives that allow farmers, ranchers, and foresters to resist selloff and conversion to urban developers are important tools to minimize the global effects of urbanization (see [11]).

(13) Realize that fragments may be best suited to conserve only a few species. As the matrix becomes more hostile to native birds, those in fragments will likely have difficulty reproducing and surviving well enough to maintain viable populations (Murphy 1988). A few species may be relatively successful, and we encourage the manager to identify those and restore habitat to make sure they continue to succeed. Generalists, concealed nesters (often grassland and shrub nesters), aerial foragers, and flocking species may be especially suitable for such management. Robinson et al. (1997) suggested that Illinois managers in chronically fragmented landscapes focus on preserves of grassland and shrubland rather than forest because viable populations of forest birds were unlikely to be maintained in fragments near agriculture. Such difficult decisions will be necessary and should be part of a regional management planning effort so that birds not protected in fragments are protected elsewhere in large reserves (Robinson et al. 1997).

(14) Develop monitoring programs that measure fitness. Restoration must include a monitoring component for determining if our well-intentioned activities actually work. As we argued earlier, simply measuring the diversity of birds using restored areas is not adequate. Managers must monitor the reproduction, survival, and dispersal of individuals to accurately gauge their progress. This information is difficult and expensive to obtain, but is essential. Mist netting and color banding can provide information on survival and movements (DeSante & Rosenberg 1997), but information on reproduction is also needed. An approach that combines monitoring of diversity, reproduction, survival, dispersal, and predator populations is feasible and is described by Donnelly and Marzluff (in press). Monitoring programs are ideal ways to encourage landowner and local conservation group participation.

(15) Develop a new educational paradigm. The public needs to understand how humans affect wildlife and what they can do to minimize their effects. Traditional extension, outreach, summer nature camps, and school programs are helpful, but they are only a start. To further engage the public in wildlife conservation, we suggest that high-profile, collaborative efforts among [or involving] reserve managers, urban planners, K–12 schools, local management agencies and municipalities, universities, and conservation organizations are needed. For example, in Phoenix and Seattle new programs (funded in part by the National Science Foundation) are encouraging researchers to work with managers to design experiments and monitoring programs that address urban ecological questions. Graduate, undergraduate, and K–12 students, local residents, and conservation groups are used to collect and analyze data. The greater public participation fostered by these programs and the stronger link between research and management should increase public support, applicability, and utility of the knowledge gained.

Research Needed to Improve Fragment Restoration

The restoration of functioning ecosystems in severely degraded areas is one of the greatest challenges facing conservation biologists. Although we suggest many ways for restoration ecologists and land managers to increase the ecological functioning of native habitat fragments in urbanized landscapes, research will be needed to guide the way. The following areas of uncertainty would benefit immediately from creative investigations.

(1) **Are corridors used by dispersing birds and do they facilitate the functioning of metapopulations?** Debate exists about the functioning of corridors as travel ways. We would benefit by knowing how to make corridors that funnel rather than trap moving birds. The use of corridors by nest predators also needs more study to determine if the benefits to dispersal outweigh the detriments of increased nest predation. The central question really is: are bird populations in fragments maintained by dispersal from distant sources and, if so, how do we facilitate this "rescue effect" without simultaneously dooming the survivors with increased disease transmission, higher predator loads, and greater risk of catastrophe?

(2) **Does increasing native vegetation of the matrix help?** We have called for managers to recognize the importance of the matrix to the fragment and increase its natural components. But how much is needed to increase the functioning of fragments? What types of natural components minimize the effects of fragmentation most of all? Are small patches of native shrubs or scattered standing dead trees useful?

(3) **How does the pattern of housing affect avian population viability in surrounding fragments?** Although we cannot reduce the amount of urban space, we can control how densely it is settled. To lobby for development most compatible with avian conservation, we need to know whether clustered and dispersed housing developments affect birds differently. One study from the deciduous forests of Missouri suggests that dispersed housing developments have less effect on native bird communities than clustered developments (Nilon et al. 1995). The generality of this result needs confirmation.

(4) **How do we design effective buffers that shield birds in fragments from the disturbance of the matrix?** Distance will likely not be a sufficient buffer, but how do we make buffers that inhibit predators, competitors, and humans that also do not unduly constrain the species we are trying to conserve inside fragments? The search for the ultimate "semipermeable membrane" to buffer fragments should be conducted in the form of replicated experiments that test various combinations of land covers separating fragments from the extensive matrix.

(5) **How does urbanization affect insect communities?** Insects are critical food resources for birds, especially during the breeding season, yet little is known about their composition and abundance in urban areas. Even less is known about how they relate to bird populations. Urban insect communities are rich and include many exotics (Lutz 1921). Richness may be related to abundant ornamental plantings in urban areas (M. Deyrup, Archbold Biological Station, personal communication, 1999). Community composition of urban and wildland insect communities differs in subtle ways. For example, streetlights favor some moths over others, and

loss of large rotting wood reduces rotten-log insect communities (M. Deyrup, Archbold Biological Station, personal communication, 1999). Loss of rotten-log communities may affect some woodpeckers, such as *Dryocopus pileatus* (pileated woodpeckers), because carpenter ant numbers would decline. Connections between such changes in community composition and bird population viability are basically unknown. Urban insects usually are met with insecticides, and the secondary effects of these on birds are mostly anecdotal.

(6) **Is it possible to use some non-native plant species to reduce invasions by species known to be disruptive to ecosystem function?** For example, *Rosa nutkana*, the native rose, is prone to blackberry invasion, but the non-native *R. rugosa* is not. Could *R. rugosa* be used to create a synthetic landscape that would not allow invasives to penetrate? Would the non-native plant naturalize and become a problem? Would it function as a food source in the landscape like its native congener? Do the benefits of limiting invasives outweigh the potential costs of increasing non-native plant abundance?

(7) **Can fragments of native habitat in urbanized landscapes make tenable contributions to avian conservation?** If urban fragments require continuous input of colonists from distant sources, are they really helping conserve birds or are they functioning as ecological traps? A modeling approach that would determine the contribution of urban fragments to regional bird population viability could suggest important aspects of populations in urban fragments that should be quantified.

(8) **What are effective means of encouraging citizens to conserve birds and their habitats and reduce their impacts?** Many Americans want a variety of birds in their yards and parks but never think twice about letting their cats roam or expanding their prized lawns. How do we inform their decisions with current science? Collaborations between ecologists, policy scientists, and urban planners will be fruitful in this arena (Alberti et al. unpublished.).

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Steps Involved in Designing Conservation Subdivisions: A Straightforward Approach

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It is best to divide the process of development planning into two broad phases, one dealing with basic information collection and analysis and the second organizing this information and making judgments about the shape of the development itself. While the first is more objective, the second clearly involves more subjective decisions, which should usually be based upon certain design principles that provide a defensible rationale.

Background Stage

The first phase, the "background stage," involves four distinct steps: understanding the locational context, mapping special features, prioritizing objectives, and integrating the information layers.

Understanding the Locational Context

Within most townships or counties there are a variety of different locational contexts, some of which are more significant than others in terms of their relevance to the design process. Perhaps the most important is the site's proximity to traditional small towns or villages. New development within or adjoining such settlements should reflect and extend the historic streetscape and street pattern, especially in terms of their regularity and interconnectedness. Relationships between dwellings and streets are also important, in terms of modest, land-conserving front setbacks, sidewalks, and continuous rows of shade trees lining both sides. Special opportunities also exist here to avoid large off-street parking lots in higher density developments by designing streets with parallel parking spaces on both sides.

An interesting example of neo-traditional village design is the one prepared for Romansville, a hamlet within West Bradford Township, Chester County, Pennsylvania, where 150 dwelling units are proposed to occupy approximately 60 acres of a 160-acre site adjacent to this country crossroads settlement (a density consistent with local zoning standards). Dwellings are predominantly single-family detached, with about a dozen semi-detached homes and several apartments above shops or offices. The design is notable for its variety of lot sizes (5,000 to 30,000 square feet); its numerous commons, greens, and playing fields; and its extensive greenbelt of fields, woods, and trails (see Fig. 1).

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Fig. 1 Site plan for the proposed expansion of Romansville, an historic hamlet within the rural/suburban township of West Bradford, Chester County, Pennsylvania. The author's design (at right) retains the entire density of the developer's original "cookie-cutter" plan (shown on the left) but arranges the development in a more compact village-like manner that preserves a substantial greenbelt of woodlands and farm fields around its perimeter. Due to their very compact nature, neo-traditional village layouts do not have the same high proportion of "view lots" that are commonly found in well-designed "conservation subdivisions." To compensate for this, the Romansville design includes five internal greens or commons (plus two ballfields), a relatively high number for a development of approximately 150 houselots

The way neighborhood streetscapes would actually look in new developments that are patterned upon traditional small towns is perhaps better illustrated by the perspective sketch in Fig. 2, from a conservation subdivision proposed in Dutchess County, New York.

In other more rural locations it is not imperative that the "traditional neighborhood" principles described above be observed, although it would be difficult to imagine a situation where they would *not* be appropriate (except in the midst of several conventional suburban subdivisions). On outlying parcels it is often equally fitting to follow more informal, irregular, or "organic" layouts, such as the one shown on the schematic plan in Fig. 3, depicting the recent "River's End" subdivision on land bordering Deep Creek, near its confluence with the Nanticoke River, just outside the town of Seaford, Sussex County, Delaware, River's End is the county's premier example of an "open space subdivision" modeled on the principles of golf course development design, but without the course (which would be expensive to build and maintain). As with its conceptual prototype, the great majority of lots abut or face onto protected open space, including greens, meadows, ponds, wetlands, and woods. Altogether the 142 lots occupy a little less than half of the 245-acre property.

Although this particular development features commodious $^{3}/_{4}$-acre houselots, this approach is particularly appropriate when smaller lots are involved, because *the adjoining open space psychologically enlarges their actual dimensions to include some of those meadows, woodlands, or wetlands that are within direct view of the houses. In addition, the open space creates a welcome buffer on at least one boundary of each of these lots, which is preferable to being closed in on all sides by other people's yards*. Interestingly, this successful subdivision was not laid out by a landscape architect or developed by a professional developer. Rather, it is the creation of a retired economist who returned to his native Sussex with a vision for building a better place to live. As an observant layman who had thought quite a lot about the subject but who had never studied it formally, Ron Hastings concluded that the most pleasant kind of rural neighborhood he could create would be one in which about

Fig. 2 Perspective sketch of streetscape featuring simple vernacular homes in Squire Green, a new conservation subdivision in the town of Pawling, Dutchess County, New York. Note the shade trees, front porches, modest front yards, and familiar feel of this new subdivision designed along the lines of traditional neighborhoods in the classic American small town. (Courtesy SCI Real Estate Development, and Do Chung Architects of Stamford, Connecticut.)

half the land remained in its natural state. And, as he says succinctly, "*Open space sells*." Having established a successful start at River's End with its upscale homes, Hastings has recently begun planning a second open space development. This development will be for Sussex County residents with moderate incomes, demonstrating that this design approach meets the needs of a wide variety of people, not just golfers and retirees with comfortable pensions.

Figure 4 shows a perspective sketch of a typical scene from a "conservation subdivision" designed in a more open and less formal fashion for a rural site out in the countryside (as contrasted with a "neo-traditional" village or an extension to a preexisting settlement).

Mapping Natural, Cultural, and Historic Features

Every new development should be based upon a fairly thorough (but not necessarily costly) analysis of the site's special features, both those offering opportunities and those involving constraints. All too often such efforts are limited to identifying legally unbuildable areas to avoid, and moving as many units as possible onto the remaining land. That type of "short-circuit planning" could be

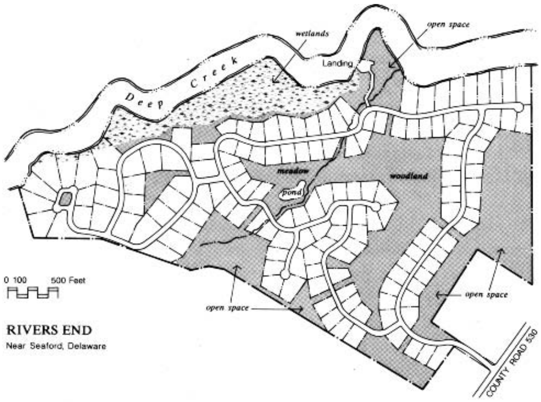

Fig. 3 The site plan for River's End, near Seaford, Delaware, shows the relationship of its 142 lots to the 100 acres of open space preserved in this layout, including woodlands, meadows, ponds, streams, wetlands, riverbanks, and a neighborhood boat landing. Very few non-golf course developments in Delaware contain significant open space features such as these

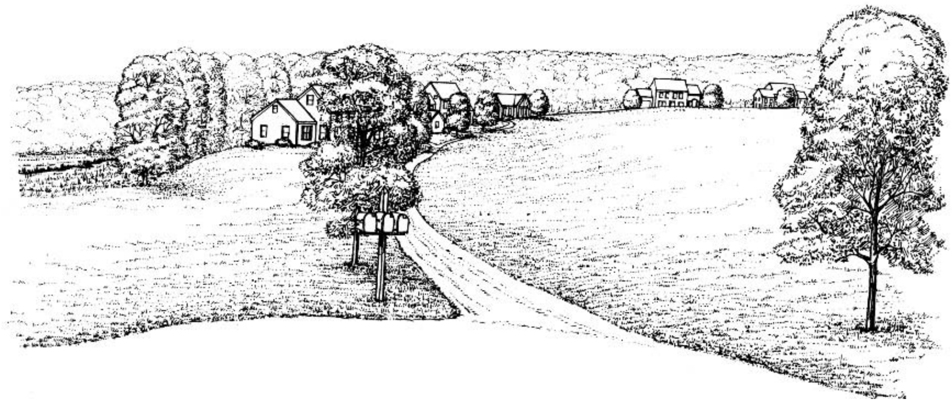

Fig. 4 View across a protected meadow toward a group of new homes built at the edge of the woods. This view, from a township road, typifies the pattern of conservation and development represented by the examples illustrated in Chapter 7

discouraged through a new design standard for residential development, one that requires "Existing Features and Site Analysis Plans" to be submitted for review and that requires applicants to be prepared to demonstrate how they have followed the four-step design process described in the second half of this chapter.

Many of the special features of interest to site designers will be known to the landowner, who should always be consulted. A country landowner will know the fields that remain damp until late spring, places where the waterfowl nest, and the hollow trees where raccoons make their homes. After walking the site several times, from end to end in different directions (preferably in early morning or late evening, when wildlife is apt to be more visible), or in the early spring when groundwater levels are highest and vernal pools might still be present, the site designer is ready to relate to the published or readily available data. To this material he or she will then bring a personal familiarity with the land in question, including an appreciation of the visually most significant aspects of the property in terms of views into the site from existing public roads, and outward prospects toward external landscape features (such as meadows, marshes, and hills).

Listed below are the factors that development designers should include in their site analyses. An asterisk (*) denotes that the particular resource should be considered to constitute part of the site's "Primary Conservation Area" described in Chapter 2. Other resources fall into the category of "Secondary Conservation Areas." If your local government or land trust has completed a community-wide natural resources inventory, that set of documents would be the best place to start. If such information has not already been compiled for your area, individuals or groups interested in eliminating that deficiency should consult *Where We Live: A Citizen's Guide to Conducting a Community Environmental Inventory* (Harker and Natter, 1994, Island Press).

1. Soils

When on-site sewage disposal is proposed, the most suitable soils for filtering effluent (whether from individual or community filter beds, or from "spray irrigation" systems) are one of the most significant resources around which development should be organized. These locations should be identified and targeted for such purposes, including "reserve areas" for use if primary areas eventually become saturated. The most favorable soils for septic disposal are those where the seasonal high water table or the impervious layer are four feet or more from the surface and which possess a medium texture, not being either too fine and silty (impeding drainage) or too coarse and gravelly (providing little filtration).

Other typical limiting conditions involve steepness or stoniness. *Medium-intensity soil survey maps* are available from local county agents of the USDA Natural Resource Conservation Service (NRCS), formerly the Soil Conservation Service (SCS). These maps are usually quite accurate down to about two acres, meaning that areas smaller than this can differ from the mapped category by being either less favorable or more suitable for the intended purpose than the map portrays. This level of mapping is generally quite adequate for identifying, in a broad-brush manner, the "Potential Development Areas" shown on the drawings for the case-study sites in Chapter 7. For purposes of identifying soils that would be suitable for subsurface sewage disposal, the accuracy of the SCS maps can be either greater or lesser than the two-acre standard, depending on the internal consistency of individual soil types, which can vary from region to region. When dealing with soil types that are highly variable over short distances (such as the shallow-to-bedrock soils of northern New England, for example), special site-specific "high-intensity" soil surveys, accurate down to one-tenth of an acre, are strongly recommended at this stage in the design. (As this kind of detailed information will eventually be needed during the review process, when individual house sites and lot lines are proposed, such a requirement would not add to total project costs.)

On-site testing would also be desirable when the soil types occurring on a property are borderline in their suitability for sewage treatment. For example, in areas where septic system regulations require a minimum depth of 24 inches of *natural* soil above the impervious layer or the seasonal high water table, NRCS soil survey categories that include soils ranging from 18 to 36 inches in this vertical dimension will not be sufficiently detailed for the site designer, who will need the results of some on-site testing before he or she can do a good job of identifying viable house sites and lot boundaries. (This is true even when the septic systems are proposed to be situated within the undivided open space, because the locations of the best soils for such facilities play a role in determining where homes can or should be sited.)

Even when sewage will be discharged off site, the medium-intensity maps are a valuable resource, as they will show locations where basements can be built without flooding, and where *wetlands* can be expected. While they are not a substitute for a detailed wetlands analysis, these NRCS maps will show approximate locations of wetlands through their "very poorly drained" classification (which means that the land is occupied by standing water for at least several months every year). The next wettest soil class, "poorly drained," is a similarly good indicator of the presence of hydric soils where seasonal water tables close to the surface make cellars impractical and road construction somewhat more costly.

2. Wetlands

Both tidal and freshwater wetlands should be identified, together with dry, upland buffer areas around them. To the extent that land in such buffer areas would be buildable under federal, state, or local regulations, full density credit would be granted for applicants to use in other locations on their sites. As noted in several other parts of this handbook, these buffers perform a number of significant functions–filtering stormwater runoff, providing critical habitat at the land-water interface, and offering opportunities for wildlife travel corridors and informal walking trails for the immediate neighborhood.

Although a good general idea of their location can be determined by consulting the medium-intensity soil survey maps described above, the National Wetlands Inventory maps published by the U.S. Fish and Wildlife Service of the Department of the Interior, and/or wetlands maps published by state planning or environmental agencies, an on-site delineation by a wetlands specialist will be necessary at some point in the process to provide greater detail. If the applicant simply wants to sketch a rough layout first, to get an approximate idea of the site's potential for open space design, these materials will probably be sufficient. However, if he or she wishes to submit a concept plan on which more detailed layouts will be closely based, on-site investigations by appropriate specialists are advisable. Since these investigations will eventually be required, they might as well be done as early in the design process as possible, to improve the accuracy of every planning step along the way.

3. Floodplains

Although there is a long-standing tradition in some coastal areas and river valleys of permitting new structures elevated on specially engineered piers in areas prone to slow-moving floodwaters (but not in high-velocity floodways), this handbook recommends against continuation of that practice because it is inherently unsafe and is contrary to broadly accepted principles of sound planning. Because such areas are sometimes deemed to be buildable in those communities, a density bonus–in addition to full density credit–should be offered to encourage developers in those areas to set their buildings, whenever practicable, beyond the 100-year floodplain, as shown on maps published by the Federal Emergency Management Agency (FEMA). On unwooded sites, views to the water will remain essentially the same, while on parcels with intervening woodlands, views can be substantially opened by removing lower tree limbs, an accommodation to developers that strikes a better balance than would otherwise be achievable. Unless wetlands are also present, construction could begin fairly close to the edge of these floodplains. A more effective measure would be to amend zoning to require that new buildings be set back 50 to 100 feet from the edge of floodplains wherever feasible, with appropriate, internally transferable density credits to avoid the "takings" issue.

4. Slopes

Due to their high potential for erosion and consequent sedimentation of watercourses and waterbodies, slopes over 25% should be avoided for clearing, regrading, or construction. Slopes of between 15 and 25% require special site planning and should also be avoided whenever practicable. Although slope maps are not published, they can be easily prepared by an engineer, planner, or landscape architect working from readily available topographic sheets printed by the U.S. Geological Survey.

5. Significant Wildlife Habitats

Habitats of threatened or endangered wildlife species form part of the "Primary Conservation Area" of any site and should be designed around and buffered. Likely travel corridors linking the areas used as food sources, homes, and breeding grounds should likewise be protected by including them in the conservation areas designated within the development. Locations that have been officially documented by state or county agencies should be identified on the development plan and buffered with additional open space for added protection whenever feasible. One of the greater challenges facing wildlife managers today involves minimizing the continued fragmentation of natural areas caused by new development, which at best often safeguards only isolated "islands" of habitat, without maintaining essential land and water connections needed on a regular basis by native animals. The importance of creating continuous greenways along waterbodies and water-courses lies primarily in their habitat conservation benefits (in addition to water quality protection and recreational trail opportunities). When isolated wildlife populations dwindle below a certain number (because their habitat has been fragmented and diminished to the point where it is unable to provide adequate food, water, and shelter), there is great danger that they will fall below their "minimum viable population" level and will disappear entirely from the locality.

Habitats of lesser significance should be placed in "Secondary Conservation Areas" to whatever extent is feasible, so that most of them will be safeguarded as well, reinforcing the "web of life" in the area's natural ecosystem. In both cases, of course, full density credit is allowed for all otherwise buildable land designated for conservation uses.

6. Woodlands

In areas where the majority of original forest has long been cleared away for commercial agriculture, woodlands may be described as remnants, often located in lower lying areas with relatively damp soils or on the steeper slopes. Despite–and perhaps because of–their small areal extent, these small woodlands play a particularly pivotal role for wildlife. Those woodlands growing on wetland soils or on steep slopes are addressed in items #2 and #3 above and should be designated as "Primary Conservation Areas" on the Site Analysis map. Those on higher, flatter terrain often consist of mature upland forest where the land is easily buildable. To the maximum extent practicable, such areas should become "Secondary Conservation Areas" to be designed around and spared the chain-saw and bulldozer blade. In other parts of the country where woodlands constitute the primary land cover, Secondary Conservation Areas might include the most mature stands, or places where unusual species or special habitats occur. In recent years concern has risen among conservation biologists and others who point out that decreases in the number of some species of "neo-tropical" songbirds (that summer in this country and migrate to Central and South America every fall) have been caused in part by both the reduction and the fragmentation of our temperate woodland habitat. Because of rising costs of woodland clearing and stump disposal–estimated to be $9,000 per acre by one Pennsylvania developer–and due to a growing preference among many homebuyers for wooded houselots offering greater privacy and requiring less maintenance, the goal of minimizing woodland clearing is likely to be abetted by market forces in the future.

The best sources for defining the extent of woodlands, hedgerows, or tree-lines are the vertical aerial photographs that are commonly available through county offices of the USDA Natural Resource Conservation Service. These may be ordered as enlargements at working scales (such as 1 inch = 100 feet) and are indispensable in accurately locating not only tree stands but even individual trees (in meadows or fields, or alongside roads). This kind of detail enables site designers to take maximum advantage of these landscape elements, which can add immense value and enjoyment to new neighborhoods. Even a simple line of trees between abandoned fields is a feature worth designing around–for its value in privacy screening, the welcome shade it casts in summer, and the limited habitat it provides. Aerial photos can also be helpful in locating the relative positions of coniferous and deciduous trees, even when the latter are in leaf, due to the darker coloration of the conifers as registered on black-and-white film.

7. Farmland

According to many environmental officials, commercial agriculture frequently contributes to water quality problems in the groundwater and surface waters of many farming counties. There are a variety of techniques that could be made better known to farmers about ways in which they could operate their farms for high crop yields in an environmentally sensitive manner. Spreading manure in appropriate amounts and at the right time, and allowing untilled filter strips to grow along stream-banks and drainage channels, are a few such ways. Some would argue that conversion of farmland to residential development is environmentally preferable because it is relatively easy to control nutrients in runoff from new subdivisions. Apart from the debate on the environmental impact of commercial farming upon surface water and groundwater, it is relatively difficult to maintain viable agriculture on the relatively small parcels spared through conservation subdivision design, especially if the land continues in traditional low-value pursuits such as field corn, soybeans, or dairying operations. For this reason, farmland preservation is not one of the principal goals of this handbook.

Environmentalists in farming areas also point out that it is usually preferable to develop farmland rather than woodland because the latter provides a much richer and diverse habitat for wildlife. Also, most of the original forest in farmland areas has already been cleared away for commercial agriculture, creating hot, dry, well-drained monocultural fields in place of shady woodlands, wetlands, and natural meadows that support a wider variety of wildlife.

However, former fields that were managed in an environmentally unfriendly manner, with heavy doses of agrichemicals to boost monocultural crop yields, can be easily converted to wildlife meadows where many species of native grasses, wildflowers, and shrubs can provide cover, food, and habitat for birds and small mammals (as has been done at the Gwynedd Wildlife Preserve of the Natural Lands Trust in Upper and Lower Gwynedd Townships, Montgomery County, Pennsylvania). In metropolitan fringe areas it is also possible to retain some of this land in specialized high-value crops (such as vegetables, fruit, and nursery stock). Such arrangements seem to work best on larger sites when overall building densities are relatively low, in the range of one acre or more per dwelling. One example is the highly successful "Farm-view" development in Bucks County, Pennsylvania, where Realen Homes built on half-acre lots (half the size usually required under existing zoning), leaving 137 of its 300 tillable acres in crops (and donating that conservation area to a local land trust). This is the fastest-selling development in its price range in the county, largely because people are buying permanent views of open space when they purchase lots in this subdivision. Because they can offer relatively attractive terms (low rents just to cover property taxes, and long leases), land trusts are often in a better position than most rural landowners to set conditions regarding the use of pesticides, manure, and so on, and to be more selective about whom they lease to.

However, in areas with serious, viable commercial agriculture, scattered large-scale residential development of any kind (including "open space designs") should be discouraged, and prime farmland should be comprehensively preserved through mechanisms such as urban growth boundaries, the purchase of development rights, the transfer of development rights, and combinations and variations of these approaches (such as the "density exchange option," as practiced in Howard County, Maryland).

8. Historic, Archaeological, and Cultural Features

Published documentation on the location of buildings or other resources with historic, archaeological, or cultural significance is far from complete. Therefore, after reviewing official lists such as the National Register of Historic Places and the historic or archaeological site inventories compiled by state and county offices of historic preservation and cultural resources, landowners and local historians or historical groups should always be consulted. In most cases, old buildings, ruins, cellar holes, earthworks, stone walls, burial grounds, or other resources will be of local rather than county-wide or regional importance. Nevertheless, as with small tree groups or nesting areas of relatively common waterfowl, it is worthwhile to steer roads, houses, and lawns to other parts of the development site to avoid impacting them when other, more suitable, locations exist for these new uses.

Because even outstanding structures listed on the National Register are not protected from demolition (unless federal funds would be involved, or unless they are also governed by a strict local historic district ordinance), these resources possess none of the legal status accorded to environmentally sensitive wetlands or floodplains and therefore should be considered as part of the "Secondary Conservation Areas." Features such as stone walls marking old field patterns and sites of known battles would be classified in the same way–placed within the open space so that they may remain intact and buffered wherever appropriate.

As no building density value is lost through this approach, it makes good sense even from a business point of view. On a wooded tract in Spotsylvania County, Virginia, one developer located his lot lines and houses to avoid disturbing or too closely encroaching upon an old mill site and lengthy earthen trenches used during the Civil War. He later capitalized on these features by erecting large cast-iron historic marker signs describing their significance and by incorporating the historic theme into his marketing strategy. In the absence of any land-use regulations prohibiting development on top of these resources, staff at the Fredericksburg-Spotsylvania National Military Park have applauded his initiative. They are also supporting proposals to incorporate "conservation subdivision design standards" into new county regulations governing development of sites containing battle-related resources.

Similar steps are being taken in Currituck County, North Carolina, where the owner of a development parcel bordering Currituck Sound has expressed interest in utilizing these creative design techniques to avoid impacting a significant Woodland Era Native American site, while also increasing the number of new homes that would face onto the water across a waterside conservation area. County officials have expressed similar interest in incorporating these design principles into their new land-use codes, based upon a demonstration design for the above site prepared by the Natural Lands Trust as part of the Albemarle-Pamlico Estuarine Study (see Site G in Chapter 7).

9. Views Into and Out from the Site

This aspect of site design is often one of the most important from the perspectives of both the developer and the general public, who tend to see properties from different directions. Developers usually wish to maximize attractive views outward from potential homesites, while the public typically desires that new development be as visually inconspicuous as possible. Although these two objectives can easily conflict, it is often possible for development to be sited or buffered in such a way that everybody's principal interests are accommodated.

From a developer's point of view, it is desirable for sales purposes to maximize the number of homes with attractive views. This can often be achieved in creative ways that are less disruptive than the results produced through conventional platting. In areas with visually prominent ridges on which homes may be perched, Secondary Conservation Areas might include the ridgetops, requiring that new development be located sufficiently below the crest so that the horizon will continue to be defined by the ridgeline, rather than by rooflines. In situations where this is not feasible due to steeply sloping hillsides or parcel configurations, houses should be designed with a low profile, and sufficient woodlands should be retained (or planted) around and behind them to soften their visual impact. Large clear-cuts to open up panoramic views should also be prohibited, and cutting should be limited to "view tunnels" from principal rooms and/or thinning of lower limbs to create "view holes" through the foliage.

In lakefront, riparian, or coastal locations offering views of waterbodies or wetlands, the design procedures recommended in this handbook would generally allow a greater number of such lots, with views through a wooded greenway where lower limbs may be removed so that the water (or wetlands) would be visible from living room windows. In addition to these "view lots," a very large proportion of the remaining lots in a well-designed conservation subdivision will abut or face onto other types of open space, such as commons, greens, ponds, meadows, and woodlands. Given the options of a conventional development, where one-third of the lots have immediate views of the water and the other two-thirds have immediate views of their neighbors' picture windows or backyards, and a conservation subdivision, where the vast majority of lots enjoy views of water, meadows, greens, woods, or other natural features, the choice seems clear. The larger total number of "view lots" in a conservation subdivision outweighs the somewhat filtered and less immediate water views available through green-way buffers. Also, the high proportion of interior lots with views of other kinds of open space makes those lots much more desirable than they would otherwise be–if simply facing other houselots.

This design approach benefits not only developers and realtors but also future residents and the general public. Greenway buffers provide the best of both worlds, helping to screen new waterfront development while not obstructing important views. One of the best examples of this is Woodlake in Midlothian, Virginia, 18 miles southwest of Richmond. Home sales have been brisk in both waterfront and interior locations in this award-winning development. It uses a 75-foot deep greenway running along the edge of the water, between Woodlake's most expensive homes ($650,000 to $700,000) and the Swift Creek Reservoir, to provide a delightful walking or bicycling experience for both abuttors and residents of interior lots (where single-family homes sell for as little as $80,000). The water is clearly visible through the wooded buffer from all abutting homes, while habitat and water quality are protected to a much higher degree than would have been the case with conventional development.

Recognizing the economic value of maintaining clean, clear water in the Inland Bays (in terms of tourism, recreation, fisheries, and real estate), a growing number of realtors have joined conservationists in advocating greenway buffers for subdivisions as well as for PRDs in Sussex County, Delaware. The highly successful Woodlake example demonstrates that providing water views and greenway buffers are not mutually exclusive, and it suggests a new planning principle for waterside development: each site should be laid out with greenway buffers, *as if the adjacent waterbody were a reservoir*.

As pointed out in item #7 above, views of preserved farmland can also add value to new houselots. And to the extent that home sites are located away from existing public roads, at the far edges of fields as seen from those thoroughfares, some rural character can be maintained with each new development.

10. Aquifers and Their Recharge Areas

The term "aquifer" refers to underground water reserves occupying billions of tiny spaces between sand grains and other soil particles, including gravel. They are "recharged" with surface water seeping downward through coarse sandy or gravelly deposits, and/or at low points in the landscape where wetlands frequently occur. Present groundwater levels in many farming areas are several feet lower than they were before drainage ditches and tiles were installed to make formerly wet ground suitable for commercial agriculture. These areas are buildable today for structures without basements and where sewage is disposed of through public sewers or with central sewerage linked with a private disposal facility (such as spray irrigation) located on higher, drier ground on other parts of the site (or on a neighboring property). Since stormwater retention ponds often dip into areas of high ground-water, runoff entering them can recharge the underlying aquifer with dissolved pollutants (typically excess nutrients from agricultural or lawn fertilizers), requiring special buffering along drainage swales to remove as much of these substances as possible.

Although many aquifer recharge areas consist of soils that are not inherently unbuildable (such as excessively drained sands and gravels, and certain of the less severe hydric soils), they should be avoided for construction when other parts of the property are available and are less constrained by environmental factors. As with all other kinds of buildable land that are placed into natural open space in a creative development plan, full density credit should generally be allowed for these soils (when their buildability is not in question, and typically when wastewater is proposed to be treated in a central location or off site). When it is not feasible to rearrange the development pattern within the site to minimize such impacts, density transfers to neighboring properties (under a "landowner compact" agreement between two or more adjoining landowners, or under a TDR plan involving nonadjacent parcels) should be thoroughly explored. Sometimes these strategies can be combined, each playing a partial role in the process of creative development and land conservation.

Integrating the Information Layers

Once all the pertinent features have been identified, located, and evaluated in terms of their significance, they need to be drawn onto overlay sheets (typically tracing paper) and looked at together.

Because ten sheets of even the lightest tracing paper would be too dense to show all the underlying information, even if they were placed on a light table (or taped to a large window on a sunny day), it is recommended that several types of features be drawn onto the same sheet–preferably features that do not coincide in terms of their location on the site. A composite map can eventually be prepared by looking at all the information layers together to see the overall pattern of potential conservation areas.

This is essentially the same basic technique used by generations of geographers and planners; it is sometimes referred to as "sieve mapping" because all the most suitable land for development becomes apparent as those areas that drop through the "sieve" of information layers. All *buildable* land will be in those areas *not* limited by the basic constraints posed by the "Primary Conservation Areas" (wetlands, floodplains, and steep slopes), and these will emerge clearly as the appropriate sheets are placed together. (This technique was substantially expanded and refined in the 1960s by Ian McHarg and given the name "ecological planning" in his widely acclaimed book *Design With Nature*. See also *The Living Landscape* by Frederick Steiner.)

After integrating those information layers, which typically comprise only a small fraction of any site, the remaining land is examined with regard to the other layers. Because two of these layers include farmland and woodland, it is obvious that all the remaining land to be considered for development will usually be entirely covered by one or more resource types. This is not a problem because these are simply information layers at this point, and *there is a basic commitment in this design approach to accommodate the entire amount of development that would otherwise be legally possible under conventional design*. As will be discussed in the next section, these other resource types must be prioritized to determine which are the most critical, significant, or irreplaceable. Those that meet such tests are placed into "Secondary Conservation Areas." This typically consumes no more than half the buildable land on the site, leaving the other half for homes, yards, and streets.

Therefore, the two steps of "integrating information layers" and "prioritizing objectives" are not entirely separate and sequential. It makes sense to look at the information layers first, then begin thinking about priorities for conservation, and then revisit the information layers to prepare a composite map showing the location of both Primary and Secondary Conservation Areas.

Prioritizing Objectives

As a rule of thumb, those features listed above with an asterisk–wetlands, floodplains, and slopes–take first priority for inclusion in the designated open space, as they represent highly sensitive environmental resources that are generally considered to be un-buildable in a legal sense, in a practical sense, or for reasons of common sense. As mentioned above, because of their limitations or inherent unsuitability for development, they should be placed in "Primary Conservation Areas," the first type of open space to be drawn on any site plan.

Within the second broad category of open space, called "Secondary Conservation Areas," resources vary more widely in importance, vulnerability, or fragility. Within each type of resource there are examples of *greater and lesser significance*, whether one is looking at woodlands (from large and/or mature stands or unusual species, to woods that are young, diseased, already thinned out, or degraded by invasive vines, for example), farmland (soils rated from "prime" to "of local significance"), or sites of historic, archaeological, or cultural interest (from inclusion on a federal list, to a typical pristine example of local vernacular building traditions, to a much altered older house missing many original features).

Within the elements or features listed above that are not marked with an asterisk, those ranking among the top of their category (such as *mature* woodland or *prime* farmland) should always be included in the open space protected as "Secondary Conservation Areas." When decisions must be made regarding the sacrifice of one resource to preserve another (such as developing fields to save woodlands, or vice versa), they should be based upon broad township-wide or county-wide considerations. For example, if one resource type is scarcer or more unusual than another, or if it contributes to biodiversity or water quality in a more compelling way, that could provide the basis for deciding which is to be spared.

In short, *priorities for conserving or developing certain kinds of resources should be based upon an understanding of what is more special, unique, irreplaceable, environmentally valuable, historic, scenic, etc., compared with other similar features, or compared to different kinds of resources altogether*. Although this process will always contain some subjectivity, a ratings approach can help to reduce inconsistent and arbitrary choices. Within each category it is often fairly obvious which features are the most worthy of preservation. The harder decisions usually involve comparisons between different categories, such as whether a small isolated woodland or a historic house should be designed around, when it is impossible to save both.

It is the overall recommendation of this handbook that natural areas generally take precedence over human artifacts, except in situations where the latter are clearly more exceptional, such as most archaeological sites. The reason is that buildings can often be reconstructed or moved, and they can certainly be photographed and documented with measured drawings. On the other hand, it is more difficult to re-create a wetland or a mature forest because of the interrelationships among plants, animals, soil, and water that comprise each natural site. There is also a growing body of evidence that it may be nearly impossible, without intensive management, to regenerate a mature deciduous woodland in the mid-Atlantic region, due to invasive vines and alien species of shrubs and trees (such as Oriental bittersweet, rosa multiflora, Japanese honeysuckle, wild grape, Tartarian honeysuckle, and Norway maple,) that seed themselves and infest newly afforesting areas. Of course, in other areas with more numerous and significant historic locations, such as pastures and ridges that once witnessed major Civil War conflicts, battle-related resources could take precedence for conservation over other types of buildable land, such as prime farmland or mature woodlands.

There will generally be special reasons in each township or county for favoring one resource type over another. In New England where forests cover most of the land, and fields are relatively uncommon, the most widely favored approach is to locate new development among the trees and to leave farmland intact. In much of the Mid-Atlantic region between southern New Jersey and the Chesapeake Bay, the reverse landscape pattern exists, providing a logical rationale for a policy preference that is exactly opposite of the one which New Englanders typically choose. Taken in its own context, each policy makes sense for the area in which it is applied.

To sum up, in the Mid-Atlantic states where the Natural Lands Trust is active, it is recommended that preference generally be given to natural areas over human-made features in the landscape, and that within the natural world, buildable woodlands be afforded greater protection than buildable farmland when one must decide which to favor. (In addition to their greater wildlife habitat value and stormwater filtering capacity, woodlands typically do not pollute watercourses and waterbodies as do farm fields with their greater nutrient loads, pesticides, and erosion-sedimentation problems.) This recommendation should not be interpreted as favoring natural areas, especially woodlands, in any situation, for there may be occasions on specific sites where cropland conservation and historic preservation could assume relatively greater importance than woodland habitat protection.

Design Stage

After completing the somewhat tedious but essential steps involved in the "background stage" described above, it is time to start *the four-step design process*, which is where the fun begins. Since the quality of the design result depends in large measure upon the accuracy and completeness of the information layers prepared previously, the findings on those sheets are critically important, and the majority of time and effort is typically spent on that background stage. Once this information is in place, it is a relatively easy process to create a conservation subdivision design, because the overall pattern of open space and development appropriate for each site is frequently rather obvious when the various layers are collated.

At this point, readers are encouraged to jump ahead to examine the site plans in Chapter 7, where the recommended approach to designing conservation subdivisions is graphically illustrated in a step-by-step manner. It is generally advisable to look at those drawings in conjunction with the textual description of the steps that makes up the remainder of this chapter.

If the maximum legal development density has not yet been calculated (on the basis of wording in the zoning ordinance relating to areas that must be excluded), or through the "yield plan" approach (in which a realistic conventional layout has been drawn), this should be done at this time. Of particular relevance here is the unbuildable land shown on the "Primary Conservation Areas" map. These areas (e.g., wetlands) should be excluded from the yield plan houselots to the extent that zoning restrictions normally prohibit them from being considered for density. The number of dwellings that would ordinarily be buildable on the property is then adopted as the number to be accommodated in the conservation design. Examples of yield plans can be seen in Chapter 7.

The following four subsections describe the basic steps involved in designing conservation subdivisions. They are applicable to both major schools of thinking current in rural planning discussions today: proponents of "rural clustering" and advocates of "neo-traditional" hamlets and villages. Whether one's design preference is for more organic layouts and loosely configured groups of houses, or for the more formal streetscapes and street patterns associated with traditional neighborhood development (based closely upon local historic precedents), the "four-step approach" described on the following pages makes good sense. However, in the case of neo-traditional village or town design, Steps Two and Three are generally reversed since the design of streetscapes and squares is of greater significance than the location of house sites (which predominate more in lower density rural conservation subdivisions, where lots tend to be larger than those in village layouts).

The following descriptions are relatively brief for two reasons. First, their brevity reflects the fact that the conceptual design stage is typically much less time-consuming than the information collection and analysis stage. Second, the text is supplemented with explanatory illustrations in Chapter 7, where seven different sites are evaluated for their conservation and development potential, culminating in broad concept plans showing proposed locations of houselots, streets, greens, commons, meadows, woodlands, and other types of open space.

Step One: *Identifying All Potential Conservation Areas*

The heart of the design process can be summarized as *four sequential steps* beginning with the all-important first one: identifying the conservation land that should potentially be protected.

These features of the property, as mentioned above, consist of the unbuildable wetlands, floodplains, and steep slopes (the "Primary Conservation Areas"), to which are added that part of the buildable uplands that are most sensitive environmentally, most significant historically or culturally, most scenic, or which possess unusual attributes that cause them to stand out from the rest of the property as areas that the average observer would miss most if they disappeared under new houselots and streets.

As mentioned earlier in this handbook, this is the general approach used by designers of highly successful golf course developments, with the basic distinction that here we are advocating preservation of natural areas as fields, meadows, and woodlands and the creation of informal public open space in the form of neighborhood commons instead of fairways, sand traps, and putting greens.

Whether one is interested in building homes around a facility for a single sport or arranging them in a parklike setting full of natural features that all can enjoy (including wildlife), the only practical way is to begin by mentally defining the open space first.

When the site plan is first sketched, the site designer should not be reluctant to include more land than he or she thinks will eventually be designated as open space, so that no potentially desirable area is prematurely left out, excluding it from consideration in the design process. If zoning provisions allow one to save about half the site by reducing lot sizes from two acres to one acre (or from one acre to 20,000 square feet), for exploratory design purposes it is recommended that two-thirds of the parcel be tentatively sketched as conservation land, at least initially. If zoning allows houselot reductions of only 25%, one should aim for 35% to 40% conservation in the first "rough cut" on sketch paper.

This exercise will quickly identify where the core areas of future development are likely to lie on the property. One should then work outward from those cores, being careful to recommend for development only those other areas that appear to be least important to conserve, looking at the site as a whole (including its relationship to neighboring parcels, as described in the next chapter). This analysis may suggest to the site designer a creative way to reduce the "development footprint" through a more compact layout than the community's clustering regulations would ordinarily allow, while saving additional land that most people would appreciate being protected and at the same time securing full legal density for the client. If so, that possibility should be further discussed, first with the client, then possibly with a realtor, and then the planning staff, all on a tentative basis before presenting it as an option at a public meeting. (There are some fairly easy ways to reduce the extent of the developed area without sacrificing any marketability, livability, or safety. These concern building setbacks and lot depth, which are discussed below in the subsection on Step Four.)

Step Two: *Locating the House Sites*

As with golf course developments, the next design step is to identify potential house site locations. Since a developer's fundamental motivation is to make money by selling either houselots or lots with houses newly built on them, and since it is well known that most people prefer (and are often willing to pay extra) to see open space from their windows, it makes economic sense to create as many "view lots" as possible and to ensure that usable open space is located within convenient walking distance from other houses in the subdivision.

One obvious way to maximize the number of view lots is to minimize their width and to maximize the livability of the homes built on them through creative modifications (such as designing houses with a windowless side wall virtually abutting one side lot line, and another sidewall containing windows facing onto a wider side yard–and the "blind" side of the next house). Such arrangements enable the development portion of the site to be utilized nearly as efficiently as if semi-detached ("twin") houses were involved, while offering buyers genuinely detached homes. Where market conditions are favorable, however, semi-detached and multi-family dwellings should be considered. Those containing just two or three units can often be designed to resemble large single-family homes, through careful attention to their bulk, massing, window arrangement, and "front" doorway locations (which can sometimes be internalized inside common entry vestibules, or situated on the sidewalls).

Another way to increase the number of houses with views is to design several flag-shaped lots (sometimes called "pork chop lots" or "pipestem lots" because of the long narrow strip of land connecting them with the street). These lots are especially useful as a design tool in odd corners of a neighborhood, such as at the end of a cul-de-sac or where a road takes a sharp turn. This kind of lot is essentially a variation on the triangular or wedge-shaped "pie lots" common in these situations, but because they need to be only wide enough to accommodate a driveway, they can have minimal street frontage (usually 20 to 25 feet is sufficient). And since their shape in the area where the house is situated tends to be more or less rectangular, they often provide more usable yard space than does the less efficient "pie lot" alternative.

Although flag-shaped lots are most appropriate in relatively low-density subdivisions where the overall density is one acre or more per dwelling, they can be useful at higher densities and should generally be permitted in all developments, with certain restrictions. To curb potential abuses, they should be limited to no more than 15 or 20 percent of the total number of lots (for instance), and when the "flag" portion is less than 10,000 or 15,000 square feet the planning board or commission should be authorized to require adequate visual screening between adjoining lots (particularly those that share a front/back boundary).

Although it is rarely possible to design layouts so that every house has a view over major open space, it is often feasible to give nearly every house a view of at least a minor open space, such as a small neighborhood common or village green, or several acres of trees and grass around a small pond doubling as a stormwater retention facility, attractively landscaped with native species such as red-twig dogwood shrubs. To the extent that residents of these homes live only a short walk away from a larger open space, hopefully including a network of informal trails through woodlands or around wildflower meadows, the neighborhood will offer much more than standard subdivisions (and also more to the non-golfing majority than do golf course developments).

According to research conducted at the University of Washington by zoologist Gordon Orians, most people's ideal dwelling location consists of a home set on a rise of ground offering long views over parklike terrain dotted with large trees with broad crowns (not unlike many golf courses). As Harvard biologist E.O. Wilson has observed,

> It happens that this archetype fits a tropical savanna of the kind prevailing in Africa, where humanity evolved for several million years. Primitive people living there are thought to have been most secure in open terrain, where the wide vista allowed them to search for food while watching for enemies. Possessing relatively frail bodies, early humans also needed cover for retreat, with trees to climb if pursued (Wilson 1994).

These possibly innate landscape preferences provide yet another reason supporting the appropriateness of the conservation subdivision approach, which enables development to be designed around site features that people generally like to see and be around.

It is clear that identifying house sites before lot lines and streets allows building locations to be carefully selected so that natural features worth preserving can be avoided, including large trees and prominent rock outcrops, as well as historic or cultural features such as stone walls, cellar holes, battle trenches, and archaeological remains. Because it is not always possible to draw the Secondary Conservation Areas sufficiently large to include all these features, some of them will probably fall into those parts of the site slated for development. However, the flexibility of this design approach enables the majority of such features to be "designed around."

Step Three: *Designing Street Alignments and Trails*

After the conservation land has been at least tentatively identified and potential homesites sketched in, the third logical step is to determine the best way to access every residence with a street system.

Areas with relatively level or rolling topography pose few street design challenges from an engineering standpoint, the major considerations being to avoid crossing wetlands and to minimize the length (and cost) of new access streets. There are further considerations from an environmental perspective, such as avoiding large trees, mature tree stands, or wildlife habitats that might happen to be within the proposed development area, or which could be in part of the open space that must be traversed to access the proposed house sites. Sometimes it is possible to split the travel lanes so that they curve apart forming an elongated, boulevard-style island between them, where a certain large tree or other natural or historic feature may be preserved and given visual prominence. (When the preservation of large trees is involved, it is essential that the entire area under the canopy's outer "drip line" be kept undisturbed from heavy construction equipment, which can easily cause permanent damage to root systems. To achieve this, temporary construction fences should be erected along such drip lines until all construction activity has been finished in the tree's immediate location.)

From an aesthetic and speed control perspective, it is important to avoid long straight street segments. Curving roads in an informal rural cluster layout, or shorter straight segments connected by 90-degree and 135-degree bends in a more formal or traditional town-like arrangement, are preferable. (Variations that combine elements of these approaches are also possible, such as short curvilinear segments terminating in frequent intersections where the choices are to turn left or right, thereby making it more difficult for motorists to travel at excessive speeds. Such practices, also including use of Y-shaped intersections, are a hallmark of many late nineteenth century subdivisions designed by Frederick Law Olmsted, such as in Brookline, Massachusetts, and Riverside, Illinois.)

Whenever possible, street systems should be designed so that their curvature or alignment produces "*terminal vistas*" of open space elements, such as village greens, water features, meadows, or playing fields. This technique will maximize the visual impact of such areas so that residents and visitors will correctly perceive the conservation emphasis that has guided the development design and recognize the subdivision as contributing positively to the community's open space goals (see Fig. 1, as well as the conservation development designs appearing in the case studies in Chapter 7).

The use of *"reverse curves"*[1] in street design is advised because of their grace and beauty. However, they should be employed in conjunction with relatively long horizontal curve radii (at least 250 feet) and on streets where traffic speed will not generally exceed 30 mph. The common prohibition against reverse curves in municipal street standards is a carryover from the highway design manuals on which many such ordinances were based. While reverse curves without intervening straight sections (or tangents) can be unsafe for high-speed traffic, a completely different situation exists for local access streets in residential subdivisions. Hardly anything destroys the grace of a street curving through a rural "conservation subdivision" more than the introduction of long, straight tangents between curves.

Another design approach that has proven to be of value in both land conservation and real estate marketing is the use of *"single-loaded" streets*. This is a technical term describing streets having houses on only one side. When lots are trimmed down in width (with homes designed more compactly to fit onto them easily, as illustrated in Appendix D), developers can easily reserve certain street lengths for single-loading–such as alongside conservation areas or around village greens or commons–without increasing their average houselot to street length ratios. In other words, the street savings gained by reducing lot widths can be used to create single-loaded situations in other parts of the subdivision, where homes can be allowed to face onto open space.

Single-loading provides homebuyers with views that are more uplifting than their neighbors' garage doors staring back at them. It also provides all subdivision residents with welcome views of their conservation land as they drive, bike, jog, or walk through their neighborhood on a daily basis, increasing everyone's quality of life as well as their property values. Such designs can be seen in most of the case studies illustrated in Chapter 7. Sales records in subdivisions featuring single-loaded streets show that homes located there sell faster and for premium prices compared with similar houses elsewhere in the development. Not surprisingly, when all the streets in a subdivision are double-loaded (as is often the case in many unimaginatively designed "cluster" developments),

[1] *Reverse curves* are consecutive left and right curves of a street in a serpentine fashion without a straight segment separating them.

conservation areas are essentially hidden behind continuous rows of houselots and the streetscape takes on a very ordinary appearance, much like those found in conventional "checkerboard" subdivisions.

One of my favorite ways to employ single-loaded streets in open-field situations is to use them in creating "*foreground meadows*" bordering the public road that serves the development. Upon entering a conservation subdivision laid out in this manner, one's first view would be of a wildflower meadow (or horse pasture) with homes located at its far end and facing this landscape feature (see Figs. 4 and 5). If such a meadow or pasture were bordered instead by a double-loaded street curving around behind it, the view from the public road, the subdivision street, and the meadow would be of house-backs, typically dominated by sliding glass doors, pressure-treated wood decks, and asymmetrical arrangements of windows (perhaps further graced by swing sets and tool sheds as well). Not only do most new houses look far better from the front (where builders spend extra money creating "curb appeal") but residents also prefer the backyard privacy provided by not turning their rear walls toward the public road in this far-too-typical manner.

Whatever layout approach is taken, every effort should be made to *connect each street with another* so that dead ends will be minimized. Interconnected streets provide easier and safer access for fire engines, ambulances, school buses, and garbage trucks, while distributing traffic more evenly and helping to avoid conditions where certain residential streets become "collectors" with everyone in the entire development funnelling through them. In circumstances where cul-de-sacs are unavoidable (typically for topographic reasons), they should always be provided with pedestrian and bike linkages to other nearby streets or to a neighborhood trail system. Where space permits they should also be designed with a central island where existing trees have been preserved or where native specie trees, shrubs, and wildflowers can be planted. Where additional off-street parking is needed, these cul-de-sacs can also function as well-treed "parking courts."

Streets serving new developments should, whenever possible, be designed to *connect with adjoining properties* that are potentially developable in the future. Although many developers strongly

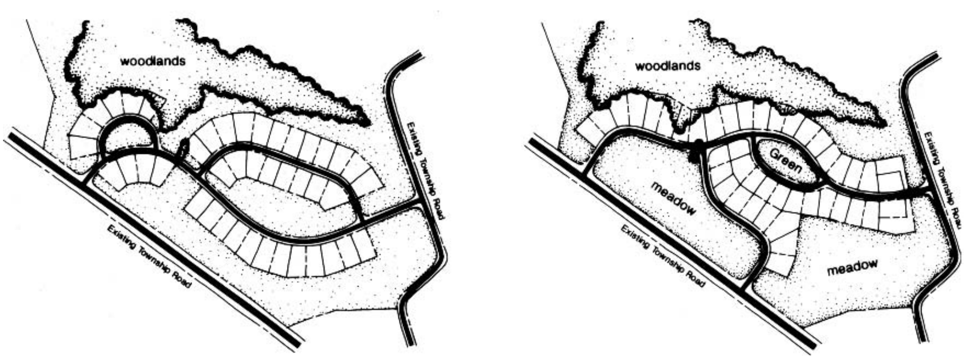

Fig. 5 *Foreground meadows* offer special opportunities to provide attractive buffers between new homes and existing thoroughfares bordering the subdivision. Following this approach, homes located along a single-loaded street typically look out over a meadow, so that the view from the township road (or rural highway) is one of a large grassy area and house fronts, which are always visually more appealing than rear elevations. This arrangement also ensures that backyard privacy will not be compromised by house backs facing onto busy thoroughfares, and it avoids the suburban artifice of the landscaped berm (which usually symbolizes a design failure). In the above two examples, where a typical "suburban cluster" approach on the left is contrasted with a "rural conservation design" with the same number of houselots, it is worth noting that the preferred approach on the right does not require any additional street length, nor does it utilize lots that are narrower. Another unusual feature, not central to the concept of foreground meadows, is the use of two flag lots, on the extreme right, paired with two frontage lots, so that all four homes would face the existing township road across a smaller grassy expanse. This provides a slightly more formal and attractive secondary entrance to the subdivision than would a view of side yards. (See also Fig. 4 for a perspective sketch of a foreground meadow.)

resist such connections, preferring to market their houses as being in self-contained neighborhoods, the lack of connecting streets between developments ultimately frustrates normal travel between neighborhoods, forcing everyone back out onto the township's or county's principal road system to travel to their friends' homes in adjacent subdivisions. In most of the examples shown in Chapter 7, cul-de-sacs have been provided with "stub-street" extensions to the adjoining properties to facilitate future connections.

Step Four: *Drawing in the Lot Lines*

The fourth and final step is the easiest–once the conservation areas have been delineated, the house sites located, and the road alignments determined. At this point in the design process, drawing in the lot lines is usually little more than a formality (one that is unnecessary in condominium developments where all land is jointly owned). Clearly the most significant aspects of a development, from the viewpoint of future residents, are how their houses relate to the open space, to each other, and to the street. Lot lines are the least important element in the development design process, yet they and the street pattern are typically the first items to be set down on paper.

Maintaining livability on the somewhat smaller lots needed in conservation subdivisions does not pose much of a design problem in zoning districts where the normal required lot size is one or two acres. The challenge increases as density rises and lot sizes become more compact. As mentioned above in the subsection describing Step Two, lot lines in high-density single-family developments can be drawn fairly close to side walls with few or no windows, enabling larger and more usable side yards to be provided on the opposite side of the house. This approach can be taken further by building on one of the side lot lines ("zero-lot line" construction), and these lot lines can follow zig-zag patterns (so-called "Z-lots").

The issue of appropriate lot depth is related directly to the presence or absence of open space along rear lot lines. When conservation land is located immediately behind them, there is good justification for shortening proposed houselots since the open space visually extends the perceived depth of backyards.

Therefore, a logical argument can be made to reduce both the width and depth of lots where houses are located off-center (i.e., closer to one side line, thereby maximizing one side yard) and where lots abut conservation areas behind them. In developments with public sewerage or with private central treatment facilities (such as "spray irrigation"), where zoning densities allow one dwelling per 20,000 square feet of land, 75% open space can be achieved by designing houselots of 5,000 square feet. These smaller, village-scale lots are often deemed to be more desirable than conventional half-acre lots by several distinct groups of potential homebuyers–such as empty-nesters, young couples, and single parents with a child or two–who want some private outdoor living space but who also wish to minimize their yard maintenance responsibilities. These lots are especially popular when they back up to protected open space, which psychologically enlarges the dimensions of the actual lot.

Architects, landscape architects, and site designers have for many years recognized that the most efficient use of a houselot occurs when the house is located "off-center and up front." Equal side yards generally produce two functionally useless areas on lots narrower than 80 feet, and front yards are practically useless in any case because they are almost always within the public view. Unless homes are located along heavily travelled streets with considerable traffic noise, there is little need for deep front setbacks to provide buffering. Placing homes where front porches or stoops are within conversational distance of sidewalks helps create conditions for friendlier neighborhoods, where passersby can exchange pleasantries with porch-sitters on weekend afternoons or summer evenings. The illustrations in Appendix C, "Detailed Houselot Designs at Higher Net Densities," show how houses, driveways, garages, and livable backyards could be accommodated even on the smallest lot size recommended in this handbook for single-family detached houses (in Site C, where base zoning is two dwellings per acre and where lots of between 5,000 and 6,000 square feet could be utilized to conserve three-quarters of the site as open space).

Note: The above sequence of steps may be modified in situations where a more formal, "neo-traditional," or village-type layout is desired. In such cases Step Two becomes the location of streets and squares, followed by the location of house sites. Whereas the relationship between homes and open space is of the greatest importance in conservation subdivisions, the relationship between buildings, streets, and squares is the dominant design consideration in the neo-traditional approach to site design. Both design approaches place more emphasis on the designation of public open space and on the provision of sidewalks, footpaths, and trails–in an effort to foster a pedestrian-friendly community atmosphere–compared with conventional suburban "cookie-cutter" layouts offering just houselots and streets.

Linking Conservation Lands in Future Subdivisions to Create an Interconnected Open Space Network: A Greener Vision

Area-Wide Maps for Conservation and Development

From the standpoint of people who are interested in how their township or county will look and feel in 10 or 20 years–as a place in which to live, raise families, and conduct business or vacation–*possibly the most important aspect of the development approach known as conservation subdivision design is the opportunity it offers to create an interconnected network of protected lands.*

Rather than simply preserving isolated pockets of greenery here and there, which do little to protect water quality and which are of relatively little use to wildlife, people in your community have within their reach the chance *to create a true fabric of open space that flows among any number of new subdivisions, as more and more properties are converted from fields and woodlands to residential developments*.

Without such a comprehensive approach, wildlife habitat will continue to dwindle and become increasingly fragmented and nonfunctional, opportunities to connect informal neighborhood trails into an area-wide greenway system will be lost forever, and water quality could be jeopardized over the longer term.

Open space connections will not be automatically preserved simply by following the general design approach described and illustrated in this handbook, on such a site-by-site basis.

The *critical unifying element*, one that can be readily created by paid professional planning staff working together with landowners, developers, conservationists, and state agency officials, *is a tool called an area-wide map of conservation and development*. This tool is essentially a composite map that brings together all the published information and the data, which are readily available from state and federal agencies, pertaining to natural features and other limiting factors that should be avoided when first sketching the broad "footprint" of future development on individual properties. Although all this information is on the public record and is freely available to those who are interested, the task of "designing around" such elements would be made considerably easier if these data were collected and charted on a series of area-wide maps that together would ultimately cover one's entire township or corner of the county. Whether the appropriate geographical divisions for these maps should be based upon watershed "divides," the convenient grid that is the basis for U.S. Geological Survey maps, or some other factors is not important to decide here.

The information that should be recorded on these area-wide maps includes the following items:

1. Wetlands (tidal and fresh)
2. The 100-year floodplains (high velocity zones and areas with only slow-moving shallow water)
3. Steep slopes (25 percent or greater)
4. Habitats of species that are endangered, threatened, or considered by state or federal agencies to be significant at the state or county level, and other ecologically unique or special areas
5. Historic, archaeological, or cultural sites listed on the National Register of Historic Places, and on state or county inventories
6. Active farmland with soil rated as prime or of "state-wide importance" by the USDA Natural Resource Conservation Service
7. High-yielding aquifers and their recharge areas
8. Woodlands of a size that makes them locally significant, and mature woodlands of one acre or more in extent

All these items should be mapped separately (including related resources grouped together above, such as fresh and tidal wetlands, which are typically regulated in different ways).

The usefulness of this tool to site designers would be greatly increased if these maps also showed *existing tax parcels*, so that potential development sites could be easily located with respect to the natural features and other constraints mapped on them. *Although area-wide maps cannot show all the features that are important for site designers, they would provide a basic understanding of the most critical elements existing on each site, especially as they relate to similar elements on adjoining parcels that may also become developed with open space designs in the future.*

In addition to these elements, others that should be considered by site designers–but which cannot be incorporated into area-wide maps–include (as mentioned in earlier chapters) views into and out from the subject property and the landowner's own knowledge of the land, including the location of areas that are special to him or her on a local or personal level, even though they might not show up on county, state, or federal maps and inventories.

After the ten data layers listed in Chapter have been compiled, the broad outlines of an *open space network* should begin to emerge. Since 50% of the buildable land on any development site can often be conserved simply by grouping new development in a more compact, efficient, and neighborly manner–without reducing overall density or profitability–at least half of each potential development parcel could be shaded green on the new Area-wide Maps of Conservation and Development without adversely affecting landowners or developers.

The great advantage of taking a broader view of future patterns of development and conservation is the opportunity it offers to pre-identify the most logical and fruitful ways of connecting conservation lands in new subdivisions. From the perspective of maintaining *functional habitats*–which include travel corridors for native wildlife as they move from nests or burrows to areas where they hunt, feed, or breed–it is essential that natural areas be linked together to the greatest extent feasible. The resulting greenways frequently offer an additional benefit that is potentially very useful to realtors and residents alike: the creation of informal walking trails through woodlands, or alongside meadows, creeks, or other natural features. The demonstrated sales advantage of homes that are located adjacent to or near open space–including greenways–constitutes convincing evidence that this is a development pattern that is sound economically as well as environmentally. (Please see Appendix F, "Sample of Real Estate Ads Mentioning Proximity of Homes to Greenways.")

Inspired by the first edition of this handbook, planners in Orange County, North Carolina (the Chapel Hill area), prepared county-wide computerized maps showing the location of lands that they have classified as possessing the characteristics of either “Primary Conservation Areas” or “Secondary Conservation Areas,” using a variation on the criteria offered in Chapter . These maps, which are a recent product of the county’s geographical information system (GIS), are intended to serve as guidelines to developers using the new “flexible development” zoning and subdivision regulations (which were based on the model language contained in the handbook’s appendix). Their county-wide network of potential conservation lands is illustrated in Fig. 1. Fig. 2 provides a visual impression of the way that various elements of the rural landscape could be protected in one neighborhood if several adjacent parcels were developed according to the conservation design principles described in this handbook.

On a more local level, officials in West Manchester Township, York County, Pennsylvania, have prepared overlay maps showing the preferred location of conservation areas with respect to existing tax parcel boundaries. By adding this information layer to its land ownership maps, the township

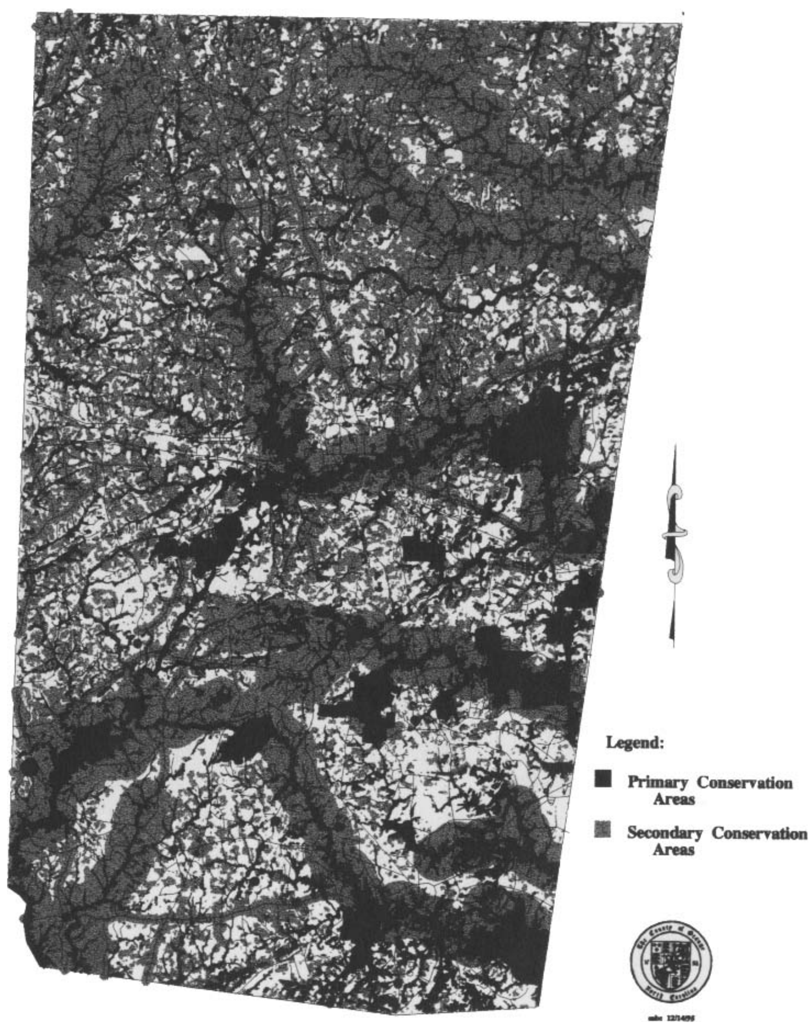

Fig. 1 The county-wide Map of Primary and Secondary Conservation Areas prepared by the Orange County (North Carolina) Planning Department, using its computer mapping capability, provides developers and their site designers with the information they need to lay out conservation subdivisions that will ultimately produce an interconnected network of open space

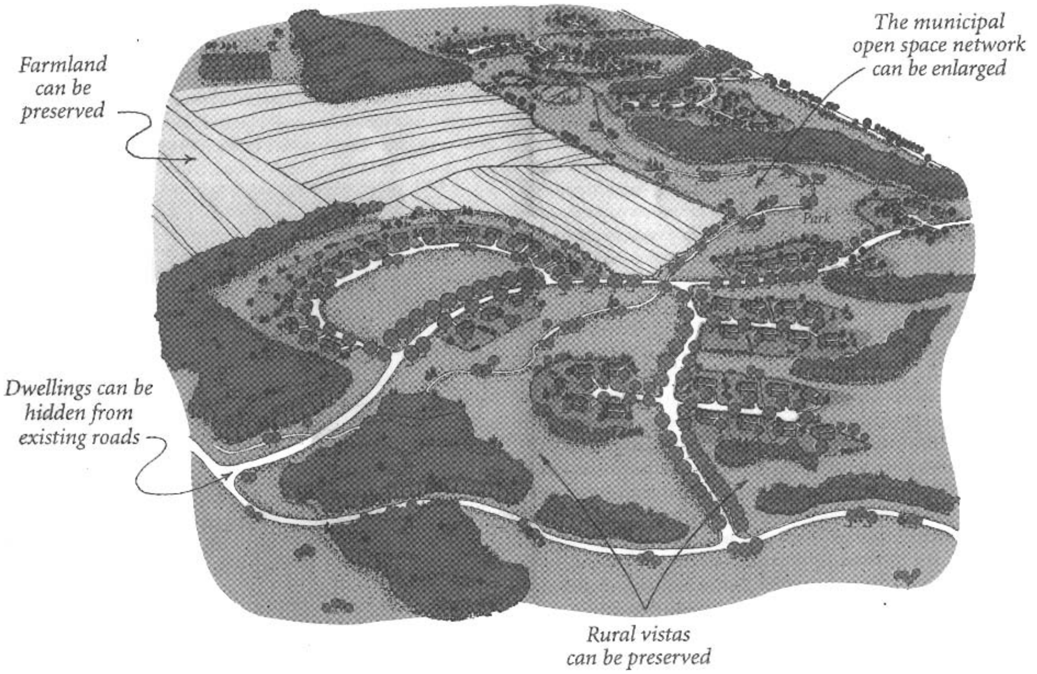

Fig. 2 This aerial perspective sketch illustrates the multiple benefits that can be achieved when conservation design is used in laying out new subdivisions on several adjoining properties. Prepared by the Montgomery County Planning Commission in southeastern Pennsylvania, this drawing shows how a conservation fabric of protected lands could be woven together to form an interconnected network of open space meeting a number of related community objectives, including the protection of woodlands, fields, scenic vistas, cultural landscapes, and additions to the municipal open space system of parks and trails

is effectively showing each property owner where the conservation land should ideally be located in any new subdivision subject to its new open space zoning provisions. This map represents a "rebuttable presumption" that the conservation areas should be laid out in this manner. Subdivision applicants may propose other configurations, but variations from the official map must be approved by the township supervisors. In the largely agrarian landscape of West Manchester, the remaining woodland areas are deemed to be of critical conservation value, and the township's map of preferred open space areas reflects a desire to preserve and enlarge existing tree groups, especially along the Little Conewango Creek, a tributary of the Susquehanna River and ultimately the Chesapeake Bay (see Fig. 3).

In Figure 4, the potential for a stream corridor greenway is illustrated in a conservation subdivision design, along with a previous conventional development where that linkage has been interrupted (and where access agreements would have to be negotiated after the fact with all riparian lot owners– a daunting prospect). The third property shows a land trust preserve that serves as a conservation node along the greenway.

While the creation of such area-wide maps would benefit both developers and conservationists, the site design principles of this handbook can certainly be implemented before they are compiled. Site designers and township or county staff members should both look at the larger picture of what is happening ecologically in the vicinity of each development as the conservation lands within it are being determined.

Fig. 3 Tax parcel maps form the base onto which officials in West Manchester Township, York County, Pennsylvania, identified those parts of each property that should ideally become designated as undivided open space in new conservation subdivisions, to protect and enhance existing woodland habitats and buffers, in this largely open agricultural landscape

Eight Self-Diagnostic Questions

Each of the following questions has been framed to help municipal officials examine a different aspect of their community's abilities to manage growth in a way that fosters land conservation. For many people, simply posing these questions will help them obtain a clearer understanding of some of the critical activities their township or county needs to undertake if they are to increase the effectiveness of their land conservation efforts. These questions, which have been posed by Michael Clarke of the Natural Lands Trust for use in the Trust's Community Land Stewardship program, are aimed at helping local leaders discover and identify areas that their community needs to work on.

1. *The Community Resource Inventory*. Has the community adequately inventoried its resources, and does the public have a sufficient understanding and appreciation of them?
2. *The "Community Audit."* Is the community monitoring and assessing its likely future under its current growth management practices, and is it taking steps to change what it does not like?
3. *Policies for Conservation and Development.* Has the community established appropriate and realistic policies for land conservation and development, and do these policies produce a clear vision of lands to be conserved?
4. *The Regulatory Framework.* Do the community's zoning and subdivision regulations reflect and encourage its policies for land conservation and development?
5. *Designing Conservation Subdivisions.* Does the community know how to work cooperatively and effectively with subdivision applicants?
6. *Working Relationships with Landowners.* Does the community have a good understanding of working relationships with its major landowners?
7. *Stewardship of Conservation Lands.* Does the community have in place the arrangements required for successfully owning, managing, and using lands set aside for conservation purposes?

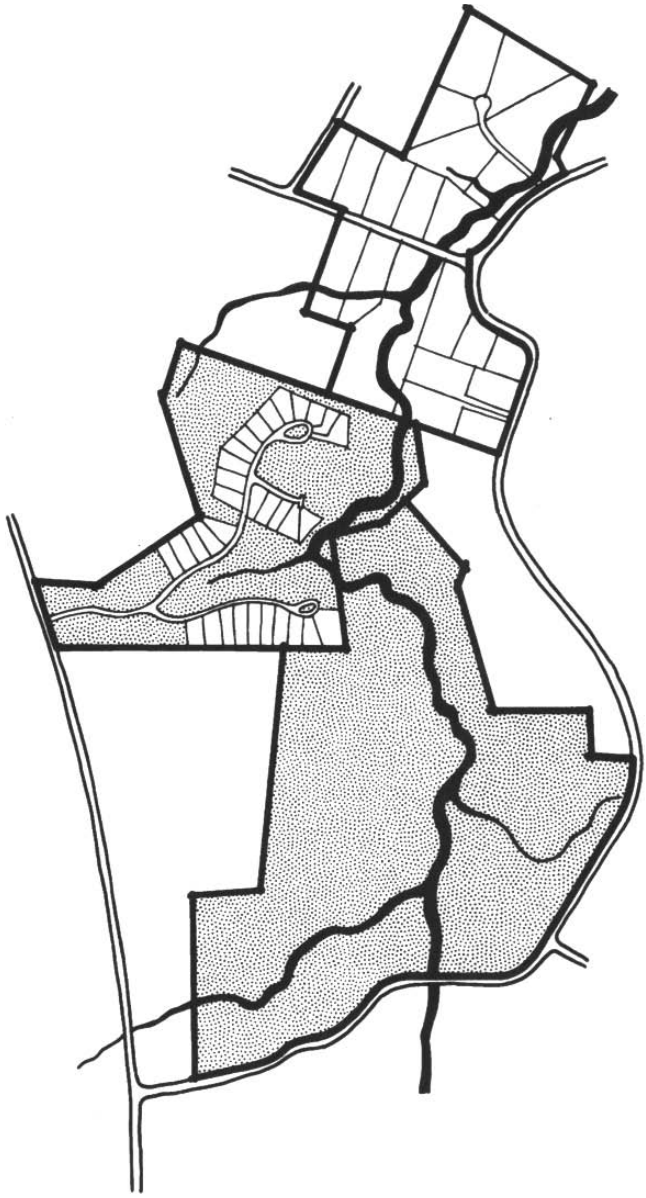

Fig. 4 Three parcels of land located along a stream illustrate how opportunities to conserve open space networks are typically lost when developments are conventionally laid out, and how such connections can be incorporated into the design of conservation subdivisions. This illustration also shows the role of a land trust preserve (or of a public park) in protecting additional segments of the stream corridor

8. *Ongoing Education and Communications.* How are local officials and the general public maintaining their knowledge of the state-of-the-art in managing growth to conserve land?

Simply asking these questions, quietly and to oneself, is likely to stimulate considerable thought about subjects that are typically not in the forefront of issues on the minds of many local officials, who are generally too busy dealing with day-to-day affairs to keep one eye on the horizon. Part of the usefulness of these questions is that they enable people to see important areas that are generally not focused upon by anyone in the community. They can help residents and officials take stock of where they are heading as a town, township, or county and to propose a mid-course correction, if necessary. It is my observation that many communities are essentially drifting, without a clear sense of direction or an established means of getting there. As the saying goes, “If you don’t care where you end up, any road will take you there.”

However, it is the rare community that cares little about its ultimate future situation. The usual problem is that, before such a list of questions is posed, most people living in areas with moderate to high growth rates often do not realize that their communities are drifting steadily in the direction of haphazard suburbanization produced by conventional zoning and subdivision codes. Each year this process permanently forecloses more and more opportunities to conserve special areas and natural lands and to create interconnected networks of open space throughout the community. That is why this list, or one similar to it, should be considered and discussed by members of local planning boards and governing bodies at least once each year.

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Beyond Greenbelts and Zoning: A New Planning Concept for the Environment of Asian Mega-Cities

Makoto Yokohari, Kazuhiko Takeuchi, Takashi Watanabe, Shigehiro Yokota

Abstract Asian mega-cities have realized explosive growth in the post-war decades. Such growth, however, resulted in serious environmental problems including air and water pollution and a lack of adequate urban infrastructure. This growth also created a chaotic mixture of urban and rural land use in the fringe of the cities. Western urban planning concepts such as zoning and greenbelt additions have been applied to the cities to encourage controlled urban growth. These landscapes located in the fringe of Asian mega-cities indicate that such attempts have not achieved significant success. Asian cities historically place land use patterns of urban and rural character next to each other. These vernacular landscapes have in the past demonstrated a workable relationship between the urban and rural environments.

It is therefore perceived that a planning concept, which respects the mixture of urban and rural land uses, should be developed and applied to encourage an ordered growth of Asian mega-cities. Farm and wooded landscapes provide key ecological functions, visual amenities and cultural services that help justify the continued relationship of rural and urban land use mixes. A planning concept that respects the vernacular landscape of the past can help provide new stability to the Asian urban environment of the 21st century.

Keywords: Environment · Land use patterns · Asian mega-cities

1 Introduction

The 21st century was to be a prosperous era for Asia. The economic crisis has changed that belief. However, despite the disappointments Asian mega-cities continue to grow to accommodate people flowing in from surrounding rural areas. Hall (1984), using Tokyo as an example, describes major problems caused by the rapid accumulation of people in Asian mega-cities and categorizes them into three groups: housing; basic services; and transportation. The future of the Asian urban environment cannot be realized without sufficient control of these three concerns.

Most Asian mega-cities have attempted to keep growth under control and encourage well-ordered developments by applying urban planning concepts that were originated in western nations. There are several successful experiences in some Asian cities, but in most cases the attempts should be regarded less than successful. This paper will discuss concepts in urban planning that reflect characteristics of Asian mega-cities by referring to the history of urbanization and the attempts made that aimed to induce ordered growth.

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2 Explosive Growth of Asian Mega-Cities in the 20th Century

The explosive growth of population in Asia in the post-war decades is already well documented. According to Institute of Population Problems Ministry of Health and Welfare (1991) the total population of Asian nations was 1.4 billion in 1950, 3.1 billion in 1990, and is estimated to reach 4.7 billion at the year 2020. This number means that almost 58.1% of the total population on the globe will be living in Asian nations at the beginning of the 21st century.

Another feature in the population of Asia is the rapid migration of people from the countryside into the cities. The United Nations reports that the population growth rate in urban areas between 1950 and 1960 in Europe 2.4%, America 3.7%, Africa 4.4%, and Asia 4.5%. The trend on population increases in individual cities is obvious. Figure 1 illustrates the population growth of London and major Asian cities. The population of London grew to 8 million over the last century, while Tokyo, Shanghai and Seoul grew to 8 million over the last 25 years.

The rapid urban growth of Asian mega-cities has resulted in a number of environmental problems. Cities in Japan including Tokyo, Kawasaki, Yokkaichi and Osaka have experienced serious air and water pollutions, as well as soil contamination, and odor, noise and vibration pollution during 1960s. These 'Seven Major Pollutions' were successfully decreased in the 1970s. However, problems in housing and transportation are still a problem. Overcrowded train systems and poor but costly housings in the Tokyo Metropolis are typical examples (Fig. 2).

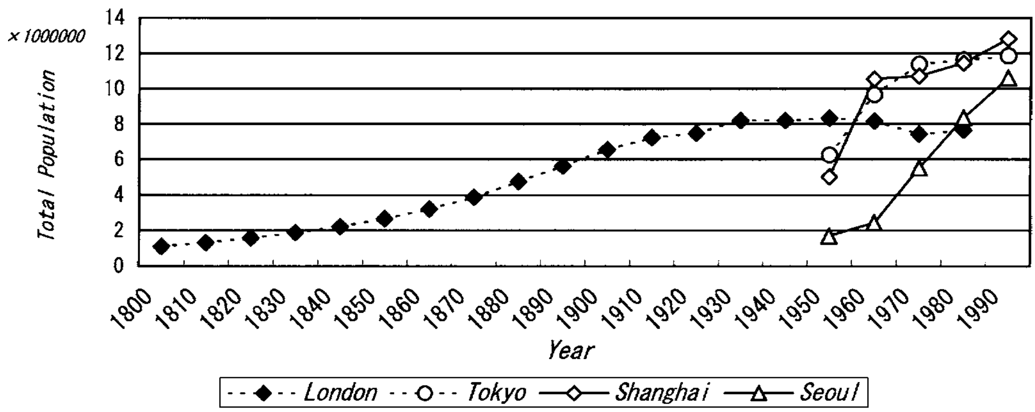

Fig. 1 Population growth in London and major Asian cities

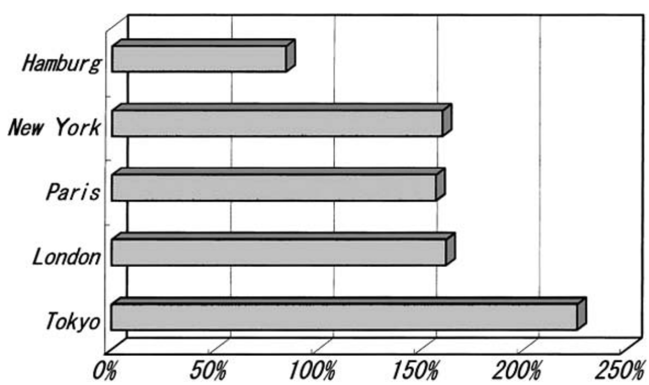

Fig. 2 Congestion ratios of subways in major cities in the world (National Land Agency, 1998)

Environmental problems are identified not only in Japanese cities but are common to most of Asian mega-cities including Bangkok, Thailand. Hayashi et al. (1993) report that urbanization in Bangkok, initiated in early 1970s, has resulted in inadequate transportation systems, poor sewage and drainage systems, and air and water pollution. Today, the traffic in the Bangkok Metropolis is regarded as one of the worst in the world (Kidokoro and Hanh, 1993).

3 Application of Western Planning Concepts

3.1 Greenbelts

When observing contemporary urban landscapes in Asian mega-cities, one may hardly realize that there have been attempts to apply western planning methods on land use to keep explosive urban expansion under control. Chaotic landscapes identified in the fringe of mega-cities are one of the clearest examples that document the absence of effective controls (Fig. 3). However, Asian mega-cities did, and still do, have physical urban plans including land use and zoning plans. Greenbelt is one of the most commonly applied concepts to Asian mega-cities.

3.1.1 Tokyo

Tokyo installed a comprehensive parks and open space master plan in 1939. The plan included parks and open space in various scales in Greater Tokyo area of 9600 km^2; from urban parks, cemeteries and allotment gardens in the central district to scenic beauty areas and national parks in remote mountains. The plan is regarded as the most ambitious plan in the history of parks and open space plans in Japan (Yokohari et al., 1996).

The plan included a greenbelt on the boundary of the Ward Area of Tokyo (Fig. 4). The Amsterdam Declaration in 1924 by the International Federation of Housing and Planning (IFHP), which identified the need for establishing greenbelts when planning for urban expansion, was the theoretical basis of the installation. The greenbelt, total 136 km^2, consists of farmland and coppice woodland, was planned on a 15 km radius to restrict disordered expansion of densely inhabited urban areas. The belt was associated with radial green corridors planned along river ravines flowing into downtown. Recreational paths such as pedestrian and horse riding trails were planned in these corridors (Minomo, 1992).

Succeeding the 1939 plan, a new open space plan for Tokyo was decided in 1943 to meet the needs of air defense during the World War II. The concept of the plan was to create open

Fig. 3 Disordered land use changes in the fringe of Bangkok, Thailand

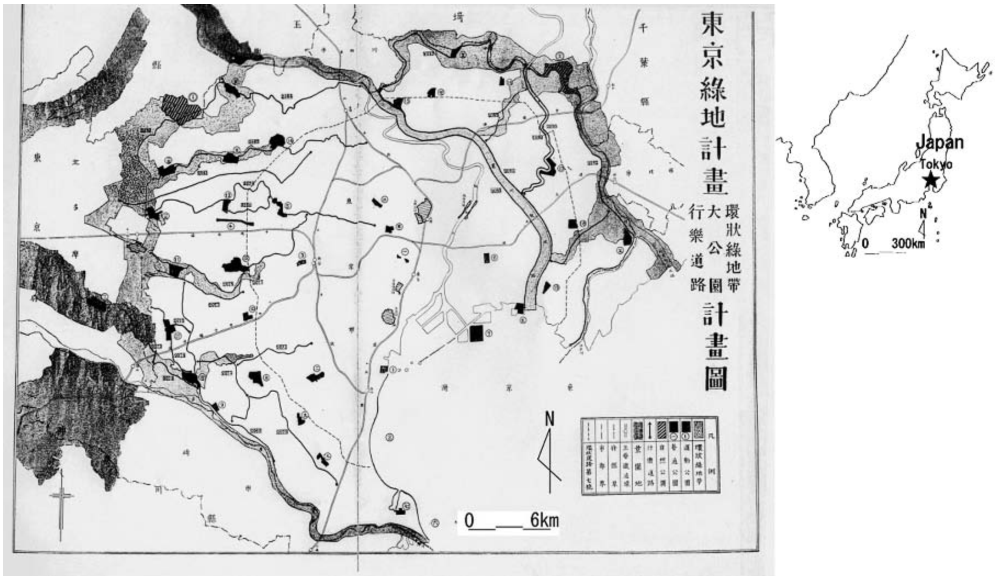

Fig. 4 Greenbelt of Tokyo in the Parks and Open Space Plan 1939. A green corridor, mostly consists of farmland and woodland, located on a 15 km radius

areas to stop the spread of fire caused by bombing and provide refuge and escape routes. The focus was to create green corridors. In addition to the greenbelt, an inner circular corridor was planned on a 10 km radius to surround urbanized area at the time by connecting major urban parks planned in the 1939 plan. Radial corridors along river ravines connected outer and inner radial corridors. The double circular and radial fluvial corridors in Tokyo reached 123.5 km (Kimura, 1990, 1992).

The air defense open space plan was terminated and succeeded by the post-war rehabilitation open space plan of 1947. In this plan, the focus was again given to the creation of circular and radial corridors (Mori, 1992). The double-ring circular green corridors, including a greenbelt and a network of radial green corridors along trunk roads, rivers and railroads were planned to connect urban parks.

If the plan was fully implemented, central Tokyo might have been one of the richest green cities in the world with over 200 km^2 of green spaces in the central district. However, as the urban landscape of Tokyo today clearly represents, the plan was poorly implemented. Only a few fluvial corridors were realized, while the circular green corridor gradually decreased and completely abolished in 1969 (Ishida, 1992). Today, only 4%, 24 km^2, of the Ward Area is ceded as parks and open space.

3.1.2 Seoul

The greenbelt surrounding Seoul, the capital city of Korea, may be nominated as one of few successful greenbelt experiences in Asian mega-cities. The greenbelt in Seoul consists of farmland and woodland, and is designated on a 15 km radius surrounding densely inhabited areas of the city (Fig. 5).

The plan to install a greenbelt was first proposed in 1963, but was not taken seriously until the end of 1960s when the explosive urban expansion became a major public concern. In 1970, the Urban Planning Act of Korea was enacted. This act was the legal basis for the creation of the Development Restriction Region, commonly known as a greenbelt. The land area totals 1567 km^2, 29% of the National Capital Region.

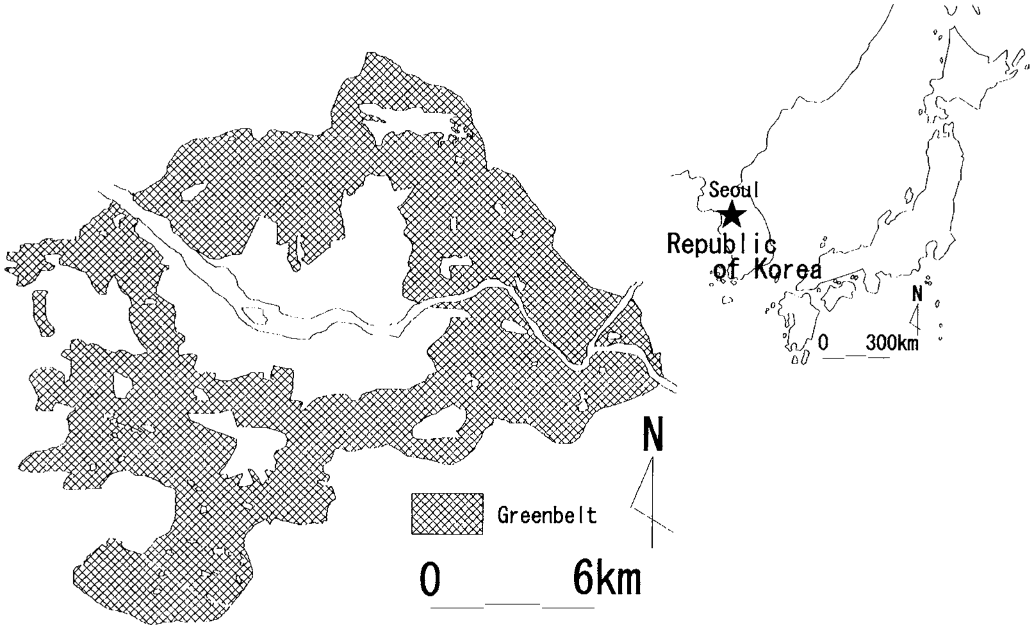

Fig. 5 Greenbelt of Seoul. Massive circular green of 1567 km^2, 29% of the whole region, located on a 15 km radius

The Seoul Greenbelt is successful due to strong legal controls in the land use of the designated zone. Tashiro and Ye (1993) point out that the development restriction in the zone is so strict that it should rather be called ‘prohibition’. Such strong control has, so far, succeeded in conserving a vast circular green corridor only 15 km away from the center. Tashiro and Ye (1993) also suggest that the migration of people from countryside into major cities was from the national security point of view a major concern, and this concern has enforced the validity of the act. It may therefore be concluded that the martial situation of Korean peninsula has supported the success of the greenbelt.

After a quarter century history of the Seoul greenbelt, we may identify several fundamental problems now emerging. The greenbelt did succeed in restricting the explosive population growth of Seoul, controlling urban sprawls into surrounding rural areas, and conserving natural/semi-natural environments near the densely inhabited district. According to the national questionnaire survey conducted in 1985, more than 85% of Korean citizens support the greenbelt. Yet, it should be noted that the strong legal control on privately owned lands within the greenbelt seriously affects the welfare of landowners and farmers. The survey reports that 67% of the citizens living in the greenbelt have negative opinions on the development restriction policy (Tashiro and Ye, 1993). It is also noted that the success of the greenbelt in restricting the growth of Seoul has ironically encouraged urban sprawl in the satellite cities located immediately outside the greenbelt.

The greenbelt in Seoul, so far, may be evaluated as one of few successful greenbelt experiences in Asia. However, it should be noted that the success is backed with the martial control, and that several fundamental problems related to strong control on land use are now starting to emerge. A close examination over time may be needed before giving a final decision on the Seoul greenbelt.

3.1.3 Bangkok

The greenbelts, or more precisely ‘green zones’, of Bangkok are located on both the eastern and the western outskirts of the city. As illustrated in Fig. 6, unlike those in Tokyo and Seoul, the greenbelt in Bangkok is not a continuous circular greenway that surrounds the core city but is a series of three isolated zones.

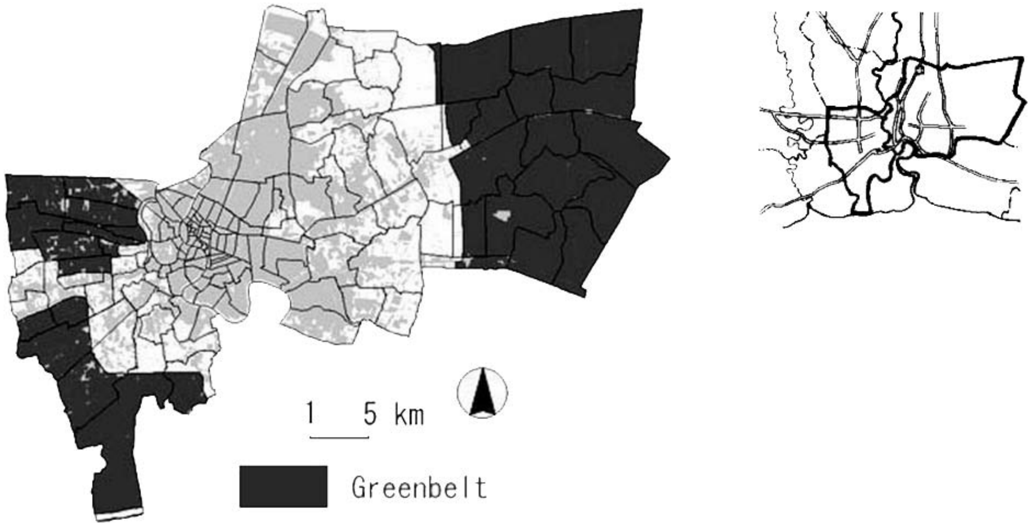

Fig. 6 Greenbelt of Bangkok. Three protected zones, mostly consist of rice paddies, designated primarily for flood control

A plan to install a greenbelt in Bangkok was first submitted in 1960. The plan, submitted by an American urban planner, was based on a finger plan concept, which allocated a series of development areas along radial transportation corridors. The basic urban development plan was then revised in 1971 by the Thai government. The concept of the finger plan in 1960 was replaced by that of a concentric circles plan, which symbolized the centralization of administrative power of the time. The idea to install a greenbelt was succeeded. The greenbelt, located on a 25 km radius with 700 km^2 of rice paddies, was finally established in 1982 with the Metropolitan Bangkok Regulations serving as its legal basis (Kidokoro, 1997).

What characterizes the greenbelt in Bangkok is that the primary function of the belt is not to restrict urban expansions but to control flooding by maintaining vast green open space. Bangkok is located on the low delta of Chao Phraya River. Rice paddies included in the greenbelt are regarded as reservoirs which store flood-water and safeguard Bangkok. In fact, even dams were constructed between the greenbelt and the core of Bangkok to enhance the function of the greenbelt as reservoirs.

While many mega-cities in Far-east Asia such as Tokyo and Seoul are reaching their limit of the growth, mega-cities in South-east Asia including Bangkok are still growing. The population of Bangkok, which used to be 3.1 million in 1970, has reached 5.7 million in 1998 and it is still increasing. Because of this ongoing population growth citizens need more housings, therefore densely populated areas of Bangkok are expanding sharply. Kidokoro (1997) reports that the area, which recorded the highest population growth rate, has shifted from the zone between 5 and 10 km radiuses in 1970s to the zone between 10 and 20 km radiuses in 1980s.

Today, the greenbelt, established on a 25 km radius, is still maintaining vast green open space. However, this is not due to successful regulatory measures but simply because the urban sprawl has not reached the area yet. Continuous and careful monitoring of urban growth in and out of the greenbelt is important for the future of the Bangkok greenbelt.

3.2 Zoning

One common observation of urban fringe areas in Asian mega-cities is the presence of micro-scaled mixture of urban and rural land uses. Such mixture has been regarded by modern urban planning as a symbol of chaotic land use, as it may result both in incomplete urban infrastructure and inefficient agriculture. Zoning was introduced to Asian mega-cities to bring order into urban developments and thus clearly separate urban areas from surrounding rural areas.

Greenbelts are established to promote the creation of well-ordered urban areas by restricting urban expansion into surrounding rural areas. Zoning aims to create well-ordered urban areas by promoting urban developments. The approaches of ‘restriction’ and ‘promotion’ have been introduced to control chaotic urban expansion of Asian mega-cities.

3.2.1 City Planning and Zoning Act of Japan

1960s was a time when Japanese economy grew rapidly. The gross national product (GNP) was ranked second place globally. Economical success, however, brought about serious pollution problems and uncontrolled human settlements. In 1965, more than 63% of the total population of Japan was living in cities, while almost one-third of local municipalities, 1100 out of 3300, were underpopulated. People kept flowing into cities from countryside during 1960s.

Such rapid and massive migration of people into cities resulted in uncontrolled land use in the suburbs. Japan’s City Planning and Zoning Act was enacted in 1968 by the national government to control the situation, by having traditional European cities, where urban areas are sharply separated from surrounding rural areas by a clear boundary line, as a target image of the act. Two types of areas were promoted in the planning district; urbanization-promotion areas and urbanization-control areas. Urbanization-promotion areas are zones that include existing urban areas, and areas that should be urbanized within $\approx$ 10 years time. Urbanization-control areas are areas that include rural areas without urban developments, except for public facilities including hospitals and schools.

30 years have past. The act, to some extent, did succeed in encouraging the creation of well-ordered urban areas. However, pressure by landowners was brought to bear on many local governments and thus vast urbanization promotion areas were designated. Consequently, a number of segmented farmland patches, which had to be turned into urban areas within 10 years from the designation, still remain in urbanization-promotion areas. The act, which aimed to introduce well-ordered urban areas, ironically encouraged uncontrolled urban developments in urbanization-promoted areas.

3.2.2 Town Planning Act of Thailand

The town planning Act of Thailand was enacted in 1975, succeeding the Town and Country Planning Act. The 1975 act was the first legislative measure that introduced zoning concept to Thailand. Designation of General Plans and Specific Plans were specified by the act. However, it was not until 1992 that the first general plan of Bangkok was established. A specific plan, which has stronger controls on land use, has never been established. Moreover, the controls specified by the act are quite lax. Land use categories are the only issues under supervision. The floor area ratio is substantially uncontrolled.

4 Uncertain Urban Edges

Although many measures have been passed to control urban expansions and uncontrolled land use mixtures, landscapes with uncontrolled urban developments and segmented farmland patches still dominate the area. Even in Japan, where explosive expansion of urban areas is no longer observed, uncontrolled mixture of urban and rural land uses still dominates Japanese urban fringe areas. In growing Asian mega-cities including Bangkok, Jakarta, and Manila, such chaos may multiply even quicker than what it did in the past.

4.1 Cities with Farmland

However, such mixture of urban and rural land uses is identified not only in contemporary Asian mega-cities but also in their history. Kyoto, one of the oldest cities in Japan, was established in the 8th century with a grid road system introduced from China, yet a number of blocks in the city remained as agricultural lands throughout feudal eras. Edo, former Tokyo, was also a city with rich green acreage. Edo was founded at the beginning of the 17th century on a marshy delta of several rivers flowing into Tokyo Bay. When the population started to increase in the 18th century a number of major land reclamation and landfill operations took place to meet demand for land. Figure 7 is a part of the map of Edo published in the early 19th century. The area illustrated in the figure, a neighborhood ≈4 km east of Edo Castle, is a typical residential area with warriors' residences and citizens' housings.

What should be noticed in this map is a series of paddy fields patches scattered in the residential area. These are not the remains of farmland that used to dominate the area, as the area was mostly under the sea before the land reclamation operation for urbanization took place. It is therefore assumed that these segmented paddy fields were 'planned' to be there, so as to play certain roles in the neighborhood. As the land elevation was at or near sea level, the neighborhood could have easily been flooded by storm surges. The agricultural fields that provided rice to Edo citizens, are assumed to have played a role as reservoirs that prevented the occurrence of floods.

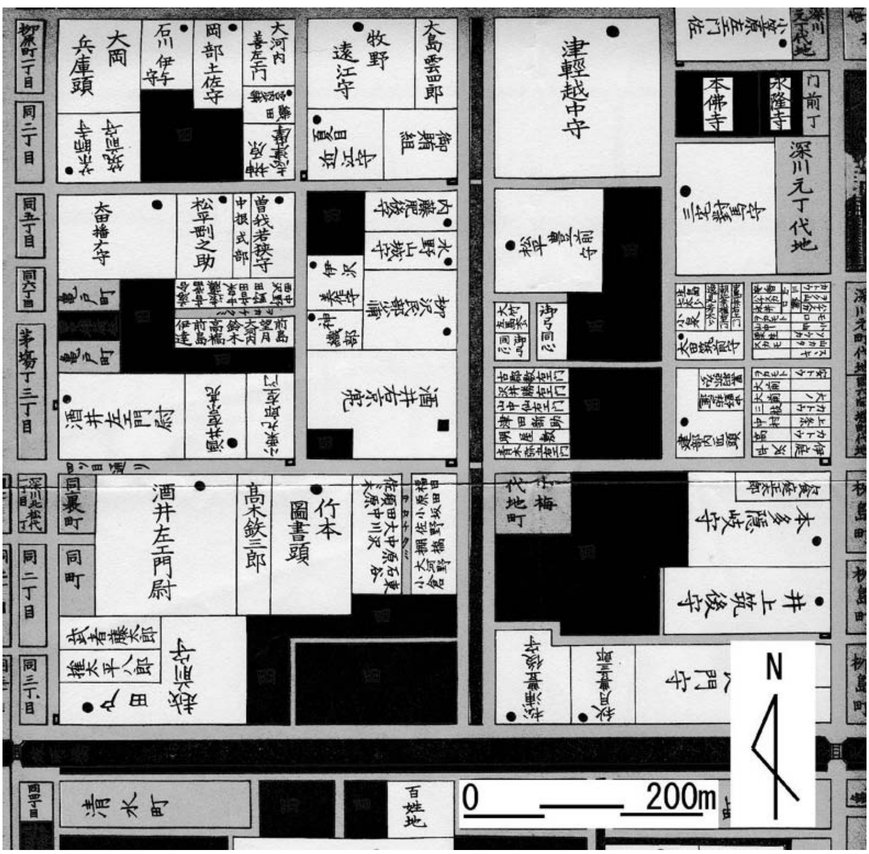

Fig. 7 Typical residential neighborhood of Edo, former Tokyo, in the 19th century. Isolated patches of paddy fields, hatched in black, surrounded by residences and housings, are found

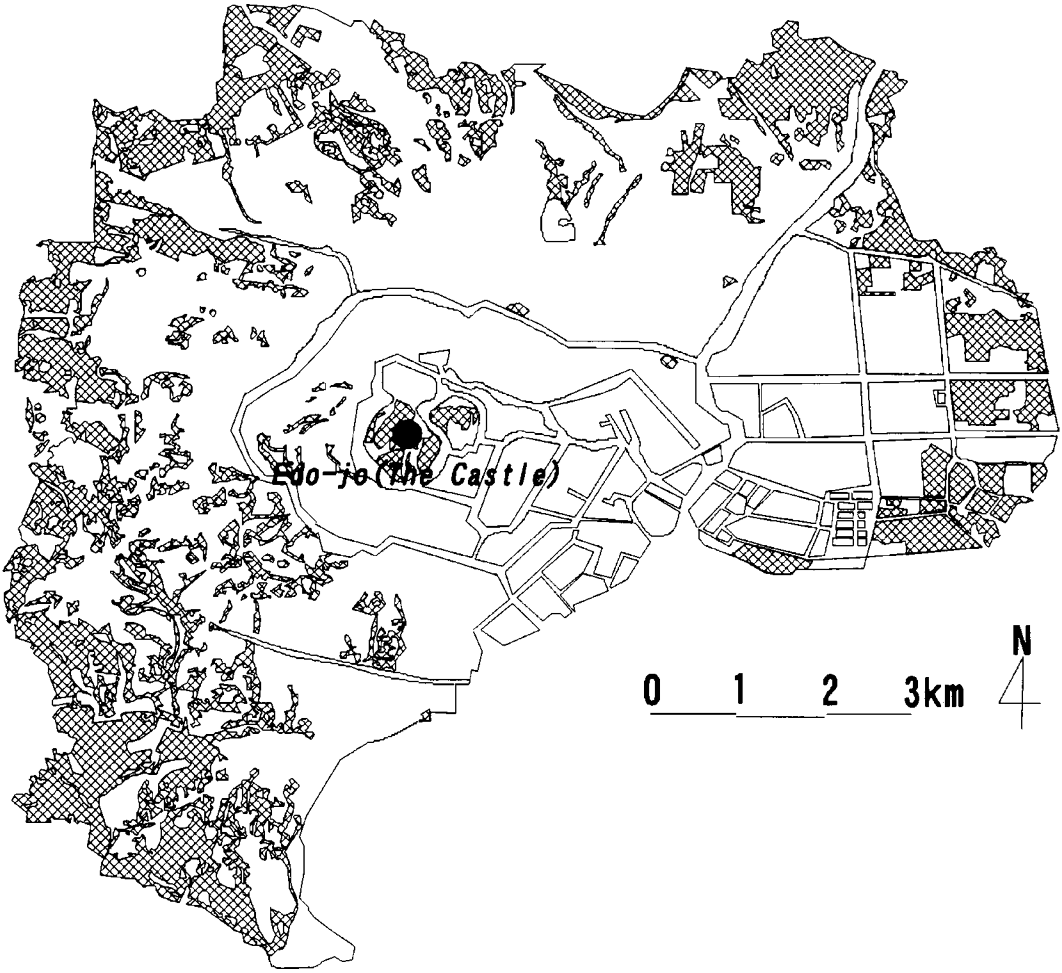

Fig. 8 Presumed open space map of Edo (re-illustrated from Tabata, 1999). Mixture of urban land use and green open space patches (hatched), mostly consist of farmland and coppice woodland, may be clearly identified

Various types of green open space were also found inside Edo. Robert Fortune, an English botanist who visited Edo in the 19th century, reported in his book "Yedo and Peking: A narrative of a journey to the capitals of Japan and China" (1863) that Edo at the time was a green city. In the book he reports pots of bonsai, gardens, nurseries, vegetable gardens, paddy fields and woods along escarpments were all found inside the city of Edo. As the result of such abundance of green open space inside the city, the boundary of the city was quite uncertain. The presumed open space map of Edo (Fig. 8) by Tabata (1999) and landscape paintings of Edo in the 19th century (Fig. 9) clearly illustrate such uncertainness of the boundary.

4.2 Rural Areas that Accepted Uncontrolled Urban Developments

When urban areas started to sprawl into surrounding rural areas, agriculture in rural areas was indeed seriously affected. However, unlike in the West, agricultural lands in the suburbs of Asian

Fig. 9 Landscape of Edo in the 19th century (Saito, 1984). The scene of residences with a glorious garden surrounded by paddy fields

mega-cities, to some extent, did survive even though they became segmented and surrounded by urban developments.

The area illustrated in Fig. 10, 20 km west of central Tokyo, is a typical example of rural areas in Japan that were invaded by urban developments but maintained a series of isolated farmland patches. A grid road system, first introduced when the area was reclaimed for agriculture, was suitable not only for agriculture but for installing urban infrastructure. Roads were, and still are, narrow, mostly <4 m wide, but can accommodate transportation systems and networks of water, electric and gas supply without fundamental changes. The absence of a traditional planning concept to separate urban developments from surrounding rural areas also allowed for the integration of urban developments. What is interesting in such areas is that landowners, mostly farmers of the area, gradually released their land for urban developments but have continued agriculture in the area. Some landowners kept their farmland merely as real estate. However, there were landowners who maintained segmented farmland patches active although they were surrounded by urban developments.

The same phenomenon is identified in the suburbs of Bangkok. The area included in Fig. 11, $\approx$15 km north of central Bangkok, has experienced the intrusion of urban developments mostly within the last 20 years. Arrays of newly developed housing complexes, called ‘Muban’, have been installed but the original road pattern, which was introduced to the area when land consolidation operations took place, is still maintained (Watanabe, 1991).

Farmers in the fringe of Asian mega-cities did release a part of their land to obtain capital, allowed urban developments to change their lives, and became laborers at factories or offices in their neighborhood. But they still did remain farmers and maintained their remaining green patches as active farmland.

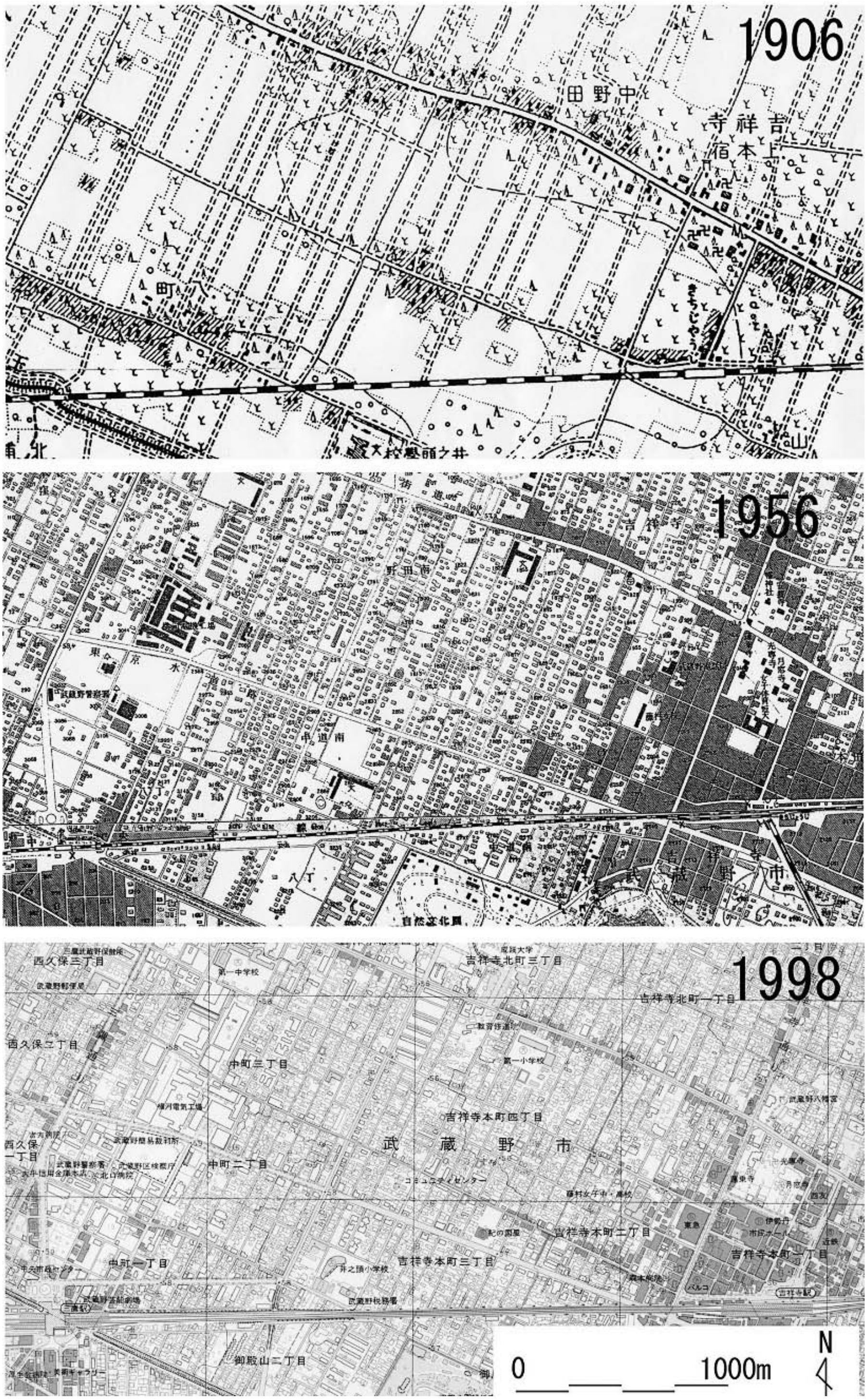

Fig. 10 Land use changes in the fringe of Tokyo: The original grid road systems in the area, Kichijoji and Mitaka city, 20 km west of the central Tokyo, accepted micro-scaled urban developments but maintained agriculture

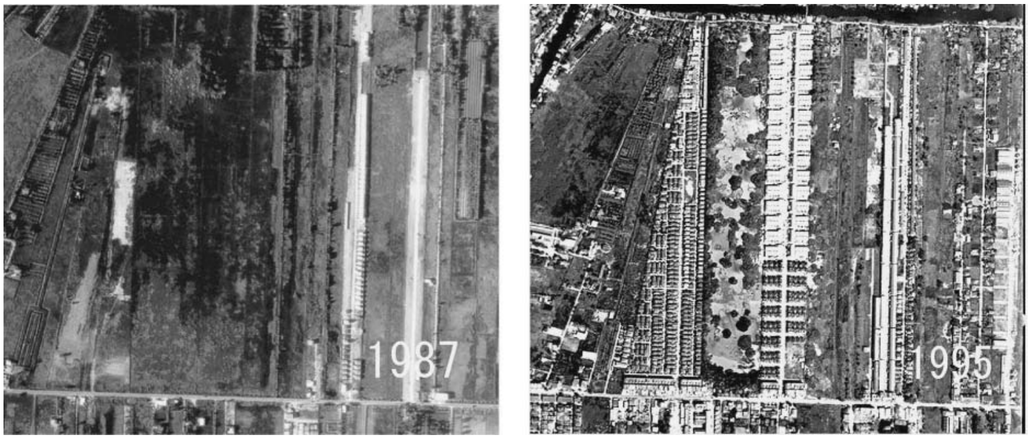

Fig. 11 Land use changes in the fringe of Bangkok: Muban developments. Arrays of linear housing complexes are developed by following the original rectangular land use pattern of the area

5 Controlled Mixture of Urban and Rural Landscapes: A New Ecological Planning Concept for the Future of Asian Mega-Cities

History shows and current trends support that the mixture of urban and rural land uses in the fringe of Asian mega-cities forms a true vernacular landscape. McGee (1991) defines areas in Indonesia with such land use mixture as Desakota, an Indonesian term that expresses the mixture of country (=desa) and city (=kota). The Landscape of Desakota is not the visual expression of a transitional stage of urbanization but is a vernacular landscape that characterizes Asian cities (Fig. 12). Desakota is the result of sustainable social systems in the fringe of Asian mega-cities (Takeuchi, 1998).

An unbalanced mixture of urban and rural landscapes must be controlled. However, the application of western urban planning concepts represented by greenbelts and zoning may not be the best solution in the Asian context. Controlled mixture of urban and rural landscapes should be nominated as one of target concepts that reflect the characteristics of Asian cities.

Contemporary restoration of the relationships between urban and rural landscapes is perceived to be a valid approach. Asihara and Lynne (1992) explains that the spatial order identified in Japanese cities is not obvious as those in European cities, and thus names it as the ‘hidden’ order. Functional relationships, which may bring order to the mixture of urban and rural landscapes, are also invisible

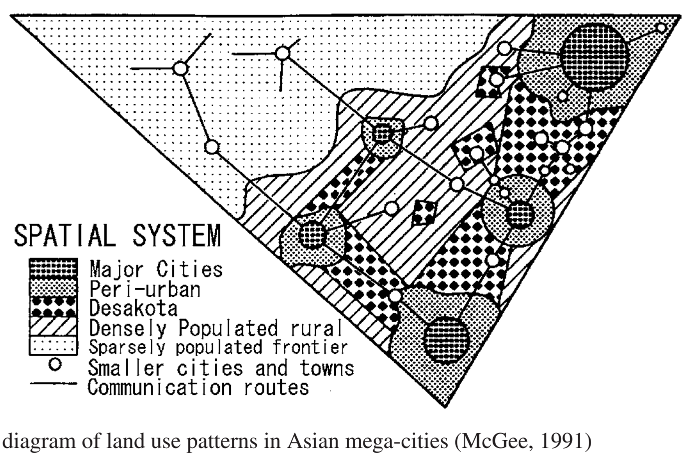

Fig. 12 Conceptual diagram of land use patterns in Asian mega-cities (McGee, 1991)

and thus may not be obvious. However, we should understand that such uncertainty on the surface is the identity of Asian cities.

The 21st century is projected to be the century of the environment. The future of human beings is dependent on the actions we will take on behalf of the environment at the beginning of the next century. Responding to such concern on the environment, the new concept for Asian mega-cities, ‘controlled mixture of urban and rural landscapes’ should be considered as a workable concept for the future.

Vegetated open spaces in urban fringe areas, including agricultural lands, are known to have many ecological functions (Yokohari et al., 1994). They may provide habitats for wildlife, become recreational spaces such as allotment gardens and aesthetically pleasuring gardens, and maintain comfortable living environment. From the time of Olmsted the effect of greenspace on the visual quality of urban areas has been understood as most important to people’s health. Yokohari et al. (1997) report that paddy fields remaining in urban fringe areas have a significant effect on controlling summer heat for surrounding residential areas. Kato et al. (1997) report the ecological functions provided by farmland and woodland yield water retention capability, landslide prevention capability and air pollution control. This suggests the need for a conservation strategy for farmland and woodland based on the ecological functions they afford.

Growing social concern for safe food is another issue that should be recognized. Modern agricultural technologies rely on the excessive use of agricultural chemicals including pesticides, insecticides and chemical fertilizers. These practices produced significant growth in agricultural production but, at the same time, caused serious pollution. Farmland located in urban fringe areas may provide safe and fresh food to consumers in the neighborhood by conducting organic and ecological farming. There may also be possibilities for consumers, urban citizens, to support such farming not only by buying products but providing labor as ‘Sunday’ farmers.

Segmented farmland patches that remain in urban fringe areas provide many services including water retention capability, micro-climate control, conservation of visual quality, and the supply of safe, fresh food. Citizens in urban areas surrounding those farmland patches support them both by means of subsidies and labor provision. Such services are perceived to be the key issues for the restoration of successful relationships between urban and rural landscapes.

Ebenezer Howard proposed a garden city concept in his book ‘Garden Cities of To-morrow’ (1902). His concept was based on the idea of utilizing surrounding rural areas to restrict urban

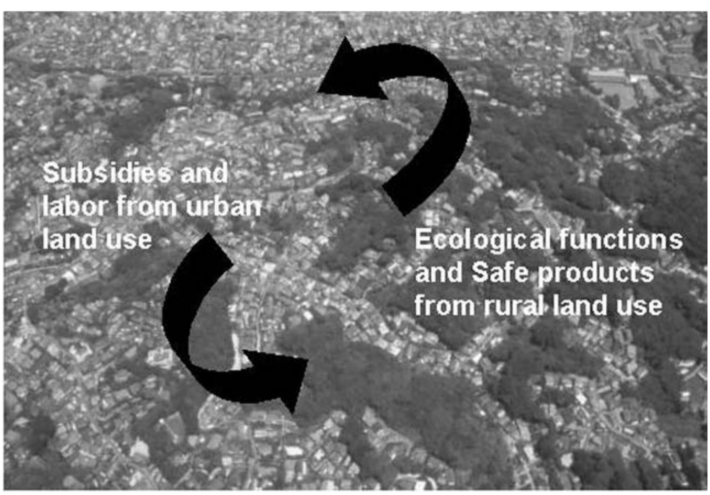

Fig. 13 Contemporary functional relationships between urban and rural areas

growth, and control residential, commercial, and industrial zones in the city with a population no larger than 30,000. His plan, in this sense, was based on the state-of-the-art urban planning concept at the time in UK. Yet, his plan was not only a physical plan but a social plan that aimed to realize a self-sufficient community free from disadvantages of major cities.

At the end of the century, by having vernacular Asian landscapes in mind, we must develop a new planning concept truly applicable to Asian cities, which supports the integration of urban and rural landscapes (Fig. 13). The reestablishment of newly conceived urban and rural landscapes, 'garden city system concept', is deemed to be the key for realizing an Asian urban environment in the 21st century.

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