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patterns, processes, and applications

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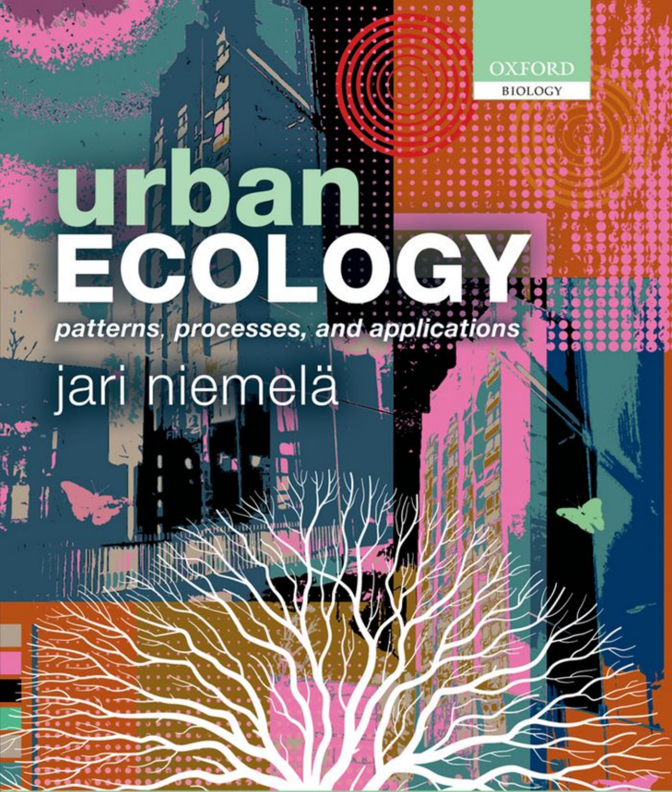

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Urban Ecology

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Patterns, Processes, and Applications

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Great Clarendon Street, Oxford OX2 6DP

Oxford University Press is a department of the University of Oxford.
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and education by publishing worldwide in

Oxford New York

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Kuala Lumpur Madrid Melbourne Mexico City Nairobi
New Delhi Shanghai Taipei Toronto

With offices in

Argentina Austria Brazil Chile Czech Republic France Greece
Guatemala Hungary Italy Japan Poland Portugal Singapore
South Korea Switzerland Thailand Turkey Ukraine Vietnam

Published in the United States
by Oxford University Press Inc., New York

First published 2011

British Library Cataloguing in Publication Data
Data available

Library of Congress Cataloging in Publication Data
Data available

Typeset by SPI Publisher Services, Pondicherry, India
Printed in Great Britain
on acid-free paper by
CPI Antony Rowe, Chippenham, Wiltshire

ISBN 978–0–19–956356–2

1 3 5 7 9 10 8 6 4 2

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Introduction

Jari Niemelä, Jürgen Breuste, Thomas Elmqvist, Glenn Guntenspergen, Philip James, and Nancy McIntyre

Most people today live in urban regions and urbanization will continue into the foreseeable future. The consequences of this phenomenon at many levels of society are still poorly understood. However, it is clear that urbanization affects the way we understand, connect with, and use natural resources in a most profound manner. The urban landscape in its diverse manifestations is becoming the most familiar environment to the majority of the human population both currently and in future generations.

Although urbanized areas cover only a small proportion of the Earth's surface, their vast human populations exert significant effects on planet Earth. However, the impacts of urbanization on biodiversity and ecosystems, especially on the global scale, remain insufficiently understood (e.g. McDonald and Marcotullio, Chapter 4.1). Thanks to a growing interest in urban ecological research, our understanding of the biotic effects of urbanization is improving, but incorporation of this scientific understanding into policy, governance, and planning is still lagging behind. This book provides insights into the complexities of urban socio-ecological systems and thereby contributes to our understanding of how such systems function and interact.

As a whole, urbanization has a myriad of effects on the environment, though not all of these effects should be perceived as negative. The variety of human impacts diversifies the urban environment by modifying existing ecosystems and by creating unique urban ones (Gilbert 1989). Consequently, biodiversity in cities may be high (Pickett *et al.* 2008). It is this diversity, the interactions between organisms and links to human activities, that are the focus of urban ecology.

Although urban dwellers have long been interested in and concerned about the future of urban plants and animals, ecologists largely ignored urban areas for the first half of the twentieth century (Grimm *et al.* 2008). Systematic urban ecological research started only after the Second World War (McDonnell, this volume). Urban ecological research has its longest traditions in Central Europe and the UK. For instance, Berlin has been intensively studied since the 1950s (Sukopp 2008). During the past decades, interest in urban ecology has increased dramatically, and we are accumulating understanding of the dynamics of urban ecosystems from various parts of the world (e.g. Alberti 2008; Breuste *et al.* 2008; McDonnell *et al.* 2009). Concerted action is now needed to provide a more comprehensive view of ecological conditions in and of cities across the world. This book includes contributions that enhance such understanding.

One result of the nineteenth and early twentieth century's neglect of the urban landscape by environmentalists and ecologists has been a view where 'wilderness' or the 'pristine' has been valued over human-dominated landscapes. In this approach, people were treated as the problem and the solution was to remove people from natural sites in order to protect or preserve them. Therefore, cities represented the worst enemies of nature. We are now moving away from this view and it is understood that cultural and biological diversity in conjunction underpin resilience and sustainability (Berkes *et al.* 2003).

The next step is to make the case that cities, precisely because of the density and diversity of their human population, are also the places where solutions can be developed. This is also the message conveyed in this book, where innovative and adaptive urban planning scenarios are explored and suggested.

The importance of an international approach is emphasized by the fact that urbanization today is being implemented as a massive, unplanned experiment in landscape change across the world. Urbanization creates a patchwork of modified land-cover types of residential, commercial, and industrial sites connected by roads and railways and interspersed with greenspaces. These mosaics have relatively similar spatial patterns throughout the world, yet we have little information on how they impact biodiversity and ecosystem processes. To improve our understanding of the ecological effects of urbanization, international, comparative research initiatives are needed. An example of such an initiative using the same taxonomic group (carabid beetles) and the same field methodology across the world to distinguish between local peculiarities and global generalities is presented by Kotze *et al.* (Chapter 3.3).

Such comparative work demonstrates generalities in biotic responses to urbanization, but purely ecological knowledge is not enough to provide the necessary scientific understanding for sustainable urban development (Elmqvist *et al.* 2008; Niemelä 1999; Grimm *et al.* 2008; Wu 2008a). Maintenance of the urban green infrastructure for residents and for biodiversity in the face of expanding cities requires that ecological knowledge be better integrated into social science research and ultimately into urban planning (Breuste 2004; Breuste *et al.* 2008; James *et al.* 2009). Urban planners, managers, and decision-makers urgently need knowledge of the socio-ecological consequences of urbanization to guide sustainable urban development (e.g. Kansanen 2004; Yli-Pelkonen & Niemelä 2006; Yli-Pelkonen 2008).

An ecosystem approach to urban planning, which includes equitable access to ecosystem services and planning at the relevant scale, is the key to sustainable cities of the future. This will not only serve cities but will help redefine our ways of interacting with nature, ecosystems, and biodiversity by making the inextricable connection between these resources and our survival as a species more evident than it is today. Investments will also have to be made in urban green infrastructure for residents and biodiversity (Breuste 2004; Breuste *et al.* 2008; James *et al.* 2009; Colding, Chapter 4.5; Pauleit *et al.*, Chapter 5.3). This book will provide insights into how ecology can be taken into consideration in urban planning.

Interdisciplinarity and collaborations between research institutions and funding agencies is essential in promoting urban ecological research. Urban ecology, if properly supported and funded, can trigger a much broader understanding of how human and ecological processes can coexist in human dominated systems and thus help societies in their efforts to become more sustainable (Marzluff *et al.* 2008a). This interdisciplinary nature is reflected in the focus of urban ecological studies that vary from plant or animal studies in the urban setting, to the integrated study of ecological and social systems and phenomena in cities (McIntyre *et al.* 2000). The interdisciplinarity of urban ecology can be operationalized by viewing urban ecology as having two components: ecology 'in' cities and ecology 'of' cities (Grimm *et al.* 2000; Wu 2008a). Ecology 'in' cities addresses basic ecological questions in urban areas, such as how ecological patterns and processes differ in cities as compared with other environments, and what is the effect of urbanization on the ecology of organisms? Ecology 'of' cities is broadly understood as the interactions between ecological and social systems in the urban setting. The understanding necessary for urban sustainability requires a combination of 'science' (ecology) and 'art' (the humanistic and holistic perspectives) into a framework for studying the links between ecology and humans in cities (Wu 2008a).

According to Wu (2008a) a major goal of urban ecology is to understand the relationship between the spatio-temporal patterns of urbanization and ecological processes. Alberti (2008) states that it is vital to understand how coupled human–ecological systems work if we are to target questions that are relevant to policy decisions more effectively. The problem in using urban ecological knowledge to enhance sustainable urban growth, however, is that there are still gaps in our knowledge of the basic ecological patterns and processes in urban landscapes, that is, we have insufficient knowledge of ecology 'in' cities (Hahs *et al.* 2009). More research is also needed on the ecology 'of' cities, particularly with a comparative approach to distinguish general urban properties from local ones. This book will contribute to filling some of our knowledge gaps.

Planners and decision-makers consider ecological information important. However, one of the problems in responding to their information requirements is that it is often a challenge for scientists to understand and appreciate the planning process. Planning is not science but a social action within a scientific, technological, and legal framework (Nilsson & Florgård 2009). If scientists want to enhance the use of their research in the planning process, it is vital to understand where inputs into the process are possible and how it should be done (Yli-Pelkonen 2008). Furthermore, it is important to understand that there are different interests involved in the decision-making and planning process, some of which are stronger than others. Although economic interests may prevail over ecological ones, it is vital that ecologists provide their values and information to the planning process. One of the aims of this book is to provide scientific understanding suitable for application in urban planning. Approaches and examples of how ecological understanding can be used in urban planning, design, and management are presented in Sections 4 and 5.

In addition to communication difficulties, the lack of a coherent theory or framework of urban ecology hampers the application of ecological knowledge in planning and design. In order to advance urban ecology as a scientific discipline and its application in planning and management, a theory or framework needs to be constructed. This book aims at providing a deeper understanding of the ecological and socio-ecological patterns and process in urban landscapes, thereby providing a foundation for a theory of urban ecology (e.g. Swan *et al.*, Chapter 3.5).

This book spans urban ecology from science to application. The book begins with a chapter on the history and development of urban ecology as a scientific discipline. The following chapters set the physical scene of urban environments (Section 1) and the book continues with chapters examining ecological patterns and processes in the urban setting (Sections 2 and 3). Chapters in Section 4 integrate ecology with human-social issues, and chapters in Section 5 conclude the book by discussing the applications of urban ecological understanding in land-use planning. These five sections are edited by leading scientists in the particular field of each section, and each section contains contributions from authors across the world to provide a global view on the issues at hand.

Section 1 provides a foundation for urban ecology by treating the physical setting as a theatre for the socio-ecological play in cities. These contributions deal with the fundaments of the urban environment: urban hydrology, climate, soils, and land-use. Chapters in Section 2 explore ecology 'in' cities by examining topics such as urban wildlife, vegetation, diversity of plants in gardens, and urban wetland diversity. Section 3 expands these themes and forms a transition from ecology 'in' cities towards ecology 'of' cities by examining ecological processes and how they are impacted by urbanization. The chapters in Section 3 address questions of the effects of humans on plants, birds, reptiles, amphibians, and insects in the urban setting. Furthermore, issues such as human–animal interactions and the selective forces they impose are addressed.

Section 4 focuses on ecology 'of' cities, that is, the integration of human social systems with ecological systems. The chapters in this section deal with issues such as social-ecological interactions in urban landscapes, and the usefulness of the ecosystem and ecosystem services approach in the urban setting. Finally, contributions in Section 5 examine how a knowledge and understanding of urban ecology is being used, and will be used, in our cities of today and tomorrow. The section includes a number of case studies from around the world to illustrate how urban ecological knowledge and understanding is being embraced into mainstream thinking. Following a contribution that sets the application of urban ecological knowledge in the widest context, there are chapters that focus on the role of providing healthy environments in which people may live and work, how green infrastructure and ecological frameworks contribute to the delivery of ecosystem benefits, and how ecological features can be integrated in to new and existing urban fabrics. The authors contributing to Section 5 also consider the major legal aspects along with schemes to provide incentives for particular actions.

It is hoped that this book will provide a compilation of information that will be useful to ecologists, planners, designers, and landscape architects. Moreover, it is hoped that the book will be a resource for ecologists in general and not just for urban ecologists because, although urban ecosystems differ in appearance from other ecosystem types, all ecosystems experience interrelationships between abiotic and biotic processes, and all ecosystems are affected by the ultimate goal of creating a sustainable world.

This book would not have been possible without the help, support, and hard work of Ian Sherman and Helen Eaton at Oxford University Press. We are very grateful to them.

The History of Urban Ecology

An Ecologist's Perspective

Mark J. McDonnell

Introduction

Urban ecology developed into a bona fide subdiscipline of ecology in the latter decades of the twentieth century from intellectual seeds sown in the late 1940s and early 1950s in Europe, North America, and Asia (McDonnell & Pickett 1993; Marzluff *et al*. 2008b; Alberti 2008; McDonnell *et al*. 2009). As with any attempt to write a history, it is especially difficult to write one in which many of the participants are still alive. Indeed, several of the pioneers of the ecological study of human-dominated landscapes have written chapters in this book. Due to the space limitations imposed by a single book chapter and the breadth of material presented in the following chapters, I will not attempt to provide a comprehensive assessment of the literature that delineates the field. Instead, this chapter will briefly discuss: 1) the early roots of urban ecology and 2) the emergence of the interdisciplinary science of urban ecology.

I have intentionally included 'an ecologist's perspective' in the title in order to explicitly expose the biases in my appraisal of this history. Over the last two decades, it has become increasingly evident that the developing discipline of urban ecology is an amalgamation of several disciplines (Alberti 2008) and it is closely aligned to the relatively new discipline of landscape ecology (Sukopp 1998, 2002; Wu 2008b; Breuste *et al*. 2008). Today, urban ecologists are trained in and utilize terminology, paradigms, and methodologies from a diversity of disciplines such as ecology, human ecology, planning, architecture, geography, economics, political science, engineering, sociology, social work, anthropology, psychology, and health sciences (McDonnell & Pickett 1993; Young & Wolf 2006; Dooling *et al*. 2007; Alberti 2008; McDonnell *et al*. 2009). Thus, urban ecology is evolving into a truly inter- and transdisciplinary science (Alberti 2008). In this chapter, I attempt to elucidate this evolution, but I am certain authors writing histories with a background in one of these other disciplines may highlight different punctuation points. However, I would hope we would all produce unique, but relatively similar depictions of the emergence of the discipline of urban ecology.

Emergence of the discipline of urban ecology

The science of ecology

The emergence of ecology as a distinct discipline occurred in Europe and North America at the end of the nineteenth century and the early twentieth century. Ecological concepts such as the 'balance of nature' have been around in different forms since Aristotle's day (Egerton 1983, 1985; McIntosh 1985). The ecological notion that nature is in a balanced state under the auspices of a creator was fundamental to the work of eighteenth century naturalists such as the father of modern taxonomy, Carl Linnaeus (Egerton 1983, 1985; McIntosh 1985). In the nineteenth century distinguished naturalists such as Malthus, De Candolle, Lyell, and Darwin articulated the role of competition for resources in controlling population growth and as a driver of extinction. These concepts form the theoretical cornerstones of modern ecology and evolutionary biology (Egerton 1983, 1985; McIntosh 1985). It is generally agreed that the publication of the first books that describe the study of ecology in the late 1800s and the early 1900s mark the establishment of the new scientific discipline of ecology (McIntosh 1985). Ecology was initially considered a fad by other biologists, but developed rapidly in the twentieth century into a legitimate and important scientific discipline (McIntosh 1985).

Early ecologists simply defined ecology as 'the study of organisms and their environment' (McIntosh 1985). There have been many modifications of this definition over the years (McIntosh 1985; Likens 1992; Krebs 2001), but I believe the definition below proposed by Likens (1992) encompasses the current scope of the discipline:

> 'The scientific study of the processes influencing the distribution and abundance of organisms, the interactions among organisms, and the interactions between organisms and the transformation and flux of energy and matter.' (Likens 1992, pg 8.)

Today, ecological studies examine complex systems at a variety of spatial scales ranging from individual organisms to our entire planet (Likens 1992). A number of subdisciplines have been identified within the discipline of ecology that focus on different proportions of the physical (abiotic) and biotic environment (Fig. 1, Likens 1992). These range from the subdisciplines of bio-geochemistry and ecosystem ecology that include elements of meteorology, geology, and hydrology (i.e. more physical sciences) to the subdisciplines of behavioural and evolutionary ecology that involve more traditional biological sciences such as systematics, genetics, and physiology (Likens 1992). To obtain a detailed understanding of the breadth of the science of ecology, I encourage readers to examine the textbooks by Odum *et al*. (2004) and Begon *et al*. (2006).

The myth of the balance of nature

Egerton (1993) describes the fundamental role that the concept of the 'balance of nature' has played in the development of the discipline of ecology.

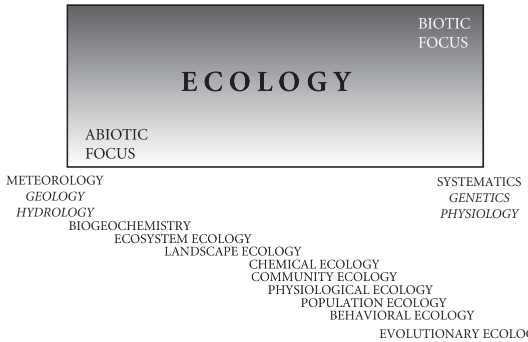

Figure 1. A diagrammatic representation of the subdisciplines of ecology arranged as a function of the different proportions of the physical (abiotic) and biotic environment they address (modified from Likens 1992)

Around the same time as proto-ecologists articulated key ecological principles, George Perkins Marsh published *Man and Nature* (Marsh 1864). In this book Marsh proposed that the world, (i.e. nature) maintained a state of stability (i.e. balance) unless disturbed by the actions of man (Lowenthal 1990; Turner & Meyer 1993). This view of the world advanced the belief that natural disturbances such as hurricanes or fires, and therefore human actions, had little or no long-term influence on the structure and function of natural ecosystems for nature was in 'balance'. This has been traditionally referred to by ecologists as the 'equilibrium paradigm' that is embodied in the cultural metaphor 'balance of nature' (Simberloff 1982; Pickett *et al*. 1992). This paradigm has had a profound effect on the development of the discipline of ecology as well as the development of conservation strategies, for it implies that to effectively study 'nature' or successfully conserve nature, ecologists and conservationists had to locate study sites far from human actions or explicitly exclude humans from conservation areas. An examination of ecology and conservation textbooks, journals, and guidelines of granting agencies that supported ecological work over the first 60 years of the twentieth century reveals a paucity of information on humans and human dominated ecosystems, especially in North America (Pickett & McDonnell 1993; Collins *et al*. 2000). In fact, Rees (1997) states that in Western industrial cultures humans are traditionally not viewed as biological beings that are part of the ecosystem. Thus, many students and practitioners of the discipline of ecology, especially those involved in basic research, were compelled to treat humans as external to the systems they studied. Consequently, for much of the twentieth century the discipline of ecology contributed relatively little information to our understanding of the ecology of human settlements. Sukopp (1998) states that some biological researchers viewed cities as 'anti-life' (i.e. without nature) for they supported few plants and animals. Those organisms that did survive had distribution patterns that were merely coincidental and were thus considered undeserving of study (Sukopp 1998).

There were exceptions to this exclusion of human-built environments as subjects of ecology, especially in Asia (Numata 1982) and Europe (Gilbert 1989; Sukopp 2002). Sukopp (2002) provides an excellent overview of the early vegetation studies conducted in European cities, while also noting the numerous applied research efforts on air pollution ('Rauchschadenforschung'), soil contamination, bioclimatology, lake rehabilitation, and on human health and welfare. Much of this early biological, physical, and sociological research conducted in cities in Europe was problem oriented and focused on improving the human condition. Indeed, similar applied research on land and water degradation and pollution effects was also conducted in North America and other cities around world over the same period, but was generally not considered under the umbrella of the emerging field of urban ecology. Instead, they formed the foundations of the new discipline of environmental science (e.g. environmental chemistry and soil science, and toxicology, etc). An important point here is that the 'balance of nature' paradigm significantly affected basic and theoretical ecology research in human-dominated landscapes, but it had little or no effect on applied research focused on solving practical problems in cities and towns.

The advent of the non-equilibrium paradigm

Over the last 30 years, there has been a mounting body of scientific evidence indicating that this old 'equilibrium paradigm' is flawed (Botkin 1990; Pickett *et al*. 1992; Fiedler *et al*. 1997). There has been an emergence of a new 'non-equilibrium paradigm' that incorporates recent knowledge of how ecosystems are structured and function (Pickett *et al*. 1992; Fiedler *et al*. 1997). This new paradigm views ecological systems as driven by process rather than end-point and as open systems potentially regulated by external forces (Pickett *et al*. 1992; Fiedler *et al*. 1997). With regard to the emergence of the subdiscipline of urban ecology, this new 'non-equilibrium paradigm' explicitly allows for the inclusion of humans as components of ecosystems studied by ecologists (Egerton 1993; Pickett *et al*. 1992; Pickett & McDonnell 1993). To make this point perfectly clear, if ecologists now accept that the structure and function of ecosystems can be regulated by external forces such as fires and floods, then human activities have to be considered as important agents of ecosystem change (Pickett & McDonnell 1993). As discussed in detail in McDonnell & Pickett (1993) and Alberti (2008), humans are components of ecosystems and human dominated ecosystems provide a new and challenging arena for inter- and transdisciplinary studies involving the physical, ecological, and social sciences.

Urban ecosystems become legitimate subjects of ecological study

While ecology was emerging as a unique discipline in the 1900s, the world population was approximately 1.4 billion people (Demeny 1990). Today, the world population is over 6.5 billion and is continuing to increase, especially in undeveloped countries (Lee 2007; UNFPA 2007; Henderson & Wang 2007). By the late 1950s and early 1960s, it was becoming apparent to everyone on the planet that humans had significantly altered local and regional ecosystems (Thomas 1956; Turner *et al*. 1990; McDonnell & Pickett 1993; Berkes & Folke 1998). In addition, the emergence of the subdiscipline of historical ecology revealed that many ecologists had indeed been working for years in areas transformed by subtle and not so subtle human actions (Cronon 2003; Botkin 1990; Russell 1993, 1997).

A decisive event in the recognition by ecologists of the inclusion of humans as components of ecosystem was the publication of the rising atmospheric CO_2 data in the early 1960s (Keeling 1998) which was soon followed by the realization that human actions were changing our global climate

(Weart 2003). Gradually, there was the recognition that no ecosystem on Earth was free from the actions of humans (Berkes & Folke 1998; Vitousek *et al*. 1997). In the early 1970s, a relatively small number of scientists in Europe, North America, and Australasia recognized the important role the science of ecology could play in mitigating the impacts of human settlements, and organized regional and national meetings to recruit multidisciplinary teams of researchers in this endeavor (Nix 1972; Duvigneaud 1974; Stearns & Montag 1974; Boyden *et al*. 1981). Around this same time, The United Nations Educational, Scientific and Cultural Organization (UNESCO) took the bold step to initiate the Man and Biosphere (MAB) program to conserve and study both natural and cultural ecosystems (Boyden *et al*. 1981; Douglas 1983; Sukopp *et al*.1990; Deelstra 1998). This UNESCO effort was critical to the establishment of the first multidisciplinary ecological studies of human settlements and thus can be credited with consolidating the emerging discipline of urban ecology (Boyden *et al*. 1981; Douglas 1983; Sukopp *et al*.1990; Deelstra 1998). The MAB program stimulated the ecological study of human settlements around the world (Dyer & Holland 1988, Song & Gao 2008) and it produced such classic urban ecology studies as the ecology of Hong Kong and its people by Boyden *et al*. (1981). Unfortunately, these early urban ecology studies did not motivate a significant number of ecologists to continue to build the discipline in the 1970s and 1980s. This was most likely due to the enduring deep-seated prejudice in the field of ecology that human-dominated ecosystems were not legitimate subjects of ecological study. A renaissance in the development of the discipline of urban ecology occurred in the late 1990s that was stimulated, in part, by the enlightened initiative of the US National Science to fund two urban long-term ecological research (LTER) programs in Baltimore, Maryland and Phoenix, Arizona, USA (Grimm *et al*. 2000).

It is now widely accepted by ecologists and others that the growth and expansion of cities worldwide are major drivers in local, regional, and global environmental change and that human actions have altered the distribution of organisms as well as the transformation and flux of energy and matter at global scales (McDonnell *et al*. 2009). The recent emergence of urban ecology and global climate research bring the science of ecology into the forefront of understanding and mitigating human impacts on ecosystems and the planet as a whole.

The role of the social sciences in the development of the discipline of urban ecology

As ecologists were busy conducting research to understand the pattern and processes of ecological systems containing relatively few humans, researchers in the fields of human ecology, planning, architecture, geography, economics, political science, sociology, social work, and health sciences were busy studying human settlements (Berkes & Folke 1998; Alberti 2008). Of particular note is the 'Chicago School' of urban sociology that pioneered the use of ecological theory and terms to describe the structure and function of cities (Hawley 1944; Park & Burgess 1967). Today, social scientists, human ecologists, and urban geographers still debate the utility of the early applications of traditional ecological theory and terms to the study of humans in cities (Alihan 1964; Catton & Dunlap 1978; Cousins & Nagpaul 1979). In contrast, their documentation of the physical features of cities coupled with investigations of how structural patterns influenced social processes is still a useful concept and is employed today to study the interactions between humans and the urban environments in which they live (Dunlap & Catton 1994). The research questions and methods developed by the Chicago School were central to the development of the discipline of human ecology (Steiner & Nauser 1993; Rees 1997).

The vast amount of information and understanding obtained by the social sciences on the structure and function of human dominated ecosystems is vital to the development of the discipline of urban ecology (Alberti 2008). But it is important to recognize that most of this work was carried out within the conceptual frameworks and paradigms of these well-established autonomous disciplines and they were typically not explicitly presented or published by their practitioners under the banner of urban ecology. Unfortunately, the explicit recognition of the role of the social sciences in the development of the discipline of urban ecology continues to be a contentious issue amongst urban ecologists (Young & Wolf 2006, 2007; Dooling *et al*. 2007).

The science of urban ecology

Defining urban ecology

It is generally accepted that the discipline of urban ecology arose in the early 1970s (Stearns & Montag 1974; Rebele 1994; Sukopp 1998, 2002; McDonnell *et al*. 2009). Historically, the term 'urban ecology' has evoked a diversity of meanings. As mentioned in the previous section, in the 1920s the Chicago School of sociologists used the term to describe their work. Deelstra (1998) reminds us that in the early 1970s the UNESCO Man and the Biosphere Program (MAB) funded the first integrated urban ecology research that brought together three sciences: 1) natural sciences, 2) engineering/planning, and 3) humanities (i.e. social sciences). Each of these sciences utilizes different terminology, paradigms, and methodologies and encompass different goals and objectives, thus resulting in an assortment of definitions and meanings for the term urban ecology. For example, natural scientists would define urban ecology in a similar manner to the definition of ecology presented earlier, but would limit their studies to urban and urbanizing landscapes (Grimm *et al*. 2000; Pickett *et al*. 2001; Alberti 2003; Niemelä *et al*. 2009). Thus, ecologists focus their work on the distribution and abundance of organisms (i.e. biodiversity) as well as the flow of nutrients and energy in urban ecosystems (i.e. ecosystem services). All of the subdisciplines of the science of ecology presented in Fig. 1 can be applied to the study of urban ecosystems. Engineers and planners, on the other hand, focus on designing facilities and services in urban environments with the goal of reducing environmental impacts and creating sustainable cities (Deelstra 1998; Pickett *et al*. 2001). Social scientists focus primarily on social structure and the social allotment of natural and institutional resources (Pickett *et al*. 2001; Alberti 2008). Today, social scientists working under the umbrella of urban ecology are exploring how to create greener more healthy and sustainable cities.

So how should we define the current discipline of urban ecology? A number of urban ecologists have endeavored to define the boundaries of the discipline (Collins *et al*. 2000; Sukopp 2002; Alberti 2008; Wu 2008b; Niemelä *et al*. 2009). I propose the following simple definition:

> Urban ecology integrates both basic (i.e. fundamental) and applied (i.e. problem oriented), natural and social science research to explore and elucidate the multiple dimensions of urban ecosystems.

Alberti proposes '…urban ecology is the study of the ways that human and ecological systems evolve together in urbanizing regions.' (Alberti 2008, page xiv). She suggests that the science of urban ecology is emerging from the integration of several disciplines as a result of a mutual interest in understanding the ecological structure and function of cities and towns. Indeed, today the borders between these different disciplines and urban ecology are blurred. Which of these disciplines become integrated, and what degree of integration occurs, will depend on the questions being addressed. As students of urban ecology are trained more broadly and more research is conducted by interdisciplinary teams of researchers, I propose that over time, as the discipline of urban ecology matures, we will experience a dissolving of the existing boundaries between the ecological and social science disciplines. As mentioned at the beginning of the chapter, urban ecology is evolving and emerging into a truly inter- and transdisciplinary science.

Urban ecology societies, books, and journals

In general, the state of the development of a scientific discipline is strongly associated with the existence of scientific societies as well as the nature, quality, and quantity of the books and journals available that address the subject matter. Although there are urban ecology sections in some societies, the first independent international society, The Society of Urban Ecology (SURE), was established in 2009 to foster and develop knowledge and implementation of urban ecology worldwide (SURE 2010).

Several books and journals devoted to the study of the ecological, physical, and sociological conditions of urban environments appeared in the early 1970s heralding the arrival of the new discipline of urban ecology. The initial books to appear were edited compilations from multidisciplinary conferences and workshops held in the United States (Stearns & Montag 1974), Europe (Bornkamm *et al*. 1982), and Australasia (Nix 1972). Each of these volumes describe the growing ecological and social problems of cities around the world and the need for an integrated approach involving the ecological, social, and physical sciences to develop strategies to mitigate the negative impacts of human settlements. A common thread running through each of these volumes is the acknowledgment of the lack of ecological information that existed at the time for cities and the call to increase urban ecology research in order to develop appropriate solutions in the future.

The increase in knowledge and understanding of urban ecosystems, as well as the development of urban ecology as an inter- and transdisciplinary science in the 1990s and early 2000s, has been well documented in a collection of edited volumes that can be categorized into three perspectives: 1) bio-ecology, 2) planning and design, and 3) education. All of these compilations are multidisciplinary in nature and bring together the leaders in the ecological, physical, and social dimensions of the field of urban ecology, but the volumes have slightly different themes and are pitched to different audiences. It is also apparent from these books that the concept of creating sustainable cities has become a major focus of urban ecology research. The majority of these edited works have a strong bio-ecology theme with a modicum of research focused on the social dimensions of urban ecosystems. This category of books includes the works of Grodzinski *et al*. (1984), Sukopp *et al.* (1990), McDonnell and Pickett (1993), Brueste *et al*. (1998), Marzluff *et al*. (2001), Marzluff *et al*. (2008b), Carreiro *et al*. (2008) and McDonnell *et al*. (2009). Compilations of urban ecology research by Platt *et al*. (1994) and Sukopp *et al*. (1995) also include a multidisciplinary perspective, but they have a strong landscape design and planning focus. Berkowitz *et al*. (2003) provide a unique collection of chapters that explore how an understanding of urban ecosystems facilitates the development of strategies to solve future environmental problems.

Somewhat surprisingly, there are relatively few single or multi-authored books, as opposed to edited compilations, that address the subject of urban ecology. Gilbert (1989) and Wheater (1999) describe the ecology of urban habitats in the United Kingdom (UK). These volumes are very descriptive and, although comprehensive in their treatment of the subject matter, they focus primarily on the physical and bio-ecological conditions of urban habitats. They are excellent examples of what we would refer to today as the ecology 'in' cities component of the discipline of urban ecology.

As previously mentioned, Boyden *et al*. (1981) published *The Ecology of a City and its People: The Case Study of Hong Kong* which is considered by many to be the first classic book on the ecology of a city. This volume provides a comprehensive account of the Hong Kong Human Ecology Program which was composed of a multidisciplinary group of researchers from the Australia National University (ANU) in Canberra, Australia. The program started in 1972 and became the first pilot program of the UNESCO Man and the Biosphere Program (MAB) in 1974. The book is considered a classic because it 1) provided an intellectual framework on which the work was conducted, 2) utilized and integrated empirical data from many sources to describe the city and its people, 3) elucidated the ecological and societal problems facing human settlements, and 4) provided the first model for integrating the ecological and sociological dimensions of urban ecosystems. Most of the ideas, concepts, and methodologies used in this study are still very relevant to urban ecologists today.

Forman (2008), a preeminent landscape ecologist, has produced a unique book that examines urban regions. Unlike the urban ecology books cited above, *Urban Regions: Ecology and Planning Beyond the City* follows more in the footsteps of his earlier *Land Mosaics: The Ecology of Landscapes and Regions* (Forman 1995) and draws primarily on concepts from the subdiscipline of landscape ecology. This new book focuses on analysing 38 urban regions from around the world principally from a planning and management perspective, but with a strong ecological foundation. *Urban Regions* will no doubt influence the way ecologists, social scientists, conservationists, land managers, and policy-makers view and manage urban regions in the future.

Alberti (2008) has written what I would consider the premiere urban ecology text book to date. In this book, entitled *Advances in Urban Ecology: Integrating Humans and Ecological Processes in Urban Ecosystems*, Alberti explores the conceptual frameworks that underpin the science while also describing in detail the many ways ecologists and social scientists study both the ecological and human dimensions of urban ecosystems. Of particular note is her synthesis of the current state of the discipline of urban ecology at the end of the book.

Finally, I would be remiss if I did not mention books that have increased our knowledge of urban environments, but were not written under the banner of urban ecology, they are, however, now clearly perceived as important contributions to the science. These include books on urban geography (Douglas 1983), landscape design (Spirn 1984) and management (Hitchmough 1994), urban forestry (Bradley 1995), wildlife management (Adams *et al*. 2006), sustainable cities (Newman & Jennings 2008), and healthy cities (Frumkin *et al*. 2004).

The journal entitled *Urban Ecology* was first published by the International Association for Ecology (INTECOL) in 1975 with the aim of publishing original research on the ecology of urban areas. One of the primary objectives of the journal was to facilitate the exchange of ideas between the ecological science community and the practitioners of urban planning and design (LaNier 1975). The scope of its audience and its content was very broad including the social, physical, and life sciences, engineers, landscape architects, planners, and administrators of urban municipalities. *Urban Ecology* published 9 volumes between 1975 and 1986 at which time it was incorporated into *Landscape and Urban Planning* which is still one of the most highly regarded journals in the field today. To fill the growing need for outlets to publish urban ecology studies, Chapman and Hall published the first volume of *Urban Ecosystems* in 1997. This journal is committed to publishing scientific investigations of the ecology of urban environments and their policy implications. It has a new publisher (Springer) and continues to provide an important role in publishing the results of inter- and transdisciplinary urban ecology research. Over the last few decades, as interest in urban ecology research grows, new journals have been launched including: *Cities, Cities and the Environment*, *Urban Habitats*, *Urban Forestry and Urban Greening*, *Urban Environment and Urban Ecology*, *Journal of Urban and Environmental Engineering*, and *Theoretical and Empirical Research in Urban Management*.

Recent developments and emerging directions in urban ecology

As the science of urban ecology emerged, researchers borrowed concepts, terminology, approaches, methodologies, and tools from a variety of other disciplines (McDonnell & Pickett 1993; Alberti 2008; McDonnell *et al*. 2009). Over the last 30 years, the discipline has grown and now possesses a unique assortment of approaches, frameworks, study locations, and methodologies that delineate urban ecology from other disciplines. The other chapters in this book give a full account of the evolution and current status of the discipline. To wind up this short history of urban ecology, I will briefly acknowledge what I believe are the recent developments and exciting new research directions in urban ecology.

Two approaches to the study of urban ecosystems that can be expressed as 1) the ecology 'in' and 'of' cities and 2) the ecology of urbanization gradients are new concepts unique to the discipline of urban ecology. The recognition by urban ecologists of the difference between the ecology 'in' and the ecology 'of' cities has proved to be a significant conceptual leap forward for the discipline (Grimm *et al*. 2000; Pickett *et al*. 2001; Alberti 2008; Wu 2008b). Studies of the ecology 'in' cities are typically single discipline, small scale, and located within a city, while ecology 'of' cities studies are interdisciplinary and multiscale incorporating both the ecological and human dimensions of urban ecosystems (Grimm *et al*. 2000; Pickett *et al*. 2001). The majority of urban ecology research to date falls into the category of the ecology 'in' cities (Alberti 2008; Hahs *et al*. 2009). There are only a few excellent examples of ecology 'of' cities studies (Alberti 2008; Hahs *et al*. 2009). If the discipline of urban ecology is to advance and enhance our understanding of urban ecosystems, it will require the active development of more inter- and transdisciplinary ecology 'of' cities studies (Alberti 2008; McDonnell & Hahs 2009). The application of the urban–rural gradient approach to the study of urban environments (McDonnell & Pickett 1990) has proved a useful concept to ecologists around the world (Theobold 2004; McDonnell & Hahs 2008). It has also inspired the development of such useful concepts as the wildland–urban interface (Radeloff *et al*. 2005). In addition, the study of urbanization gradients has provided a foundation for the future development of the comparative ecological study of cities and towns at regional and global scales (Niemelä *et al*. 2009; McDonnell & Hahs 2009).

Urban ecologists have been building on existing, as well as creating new and unique, conceptual frameworks in which to study urban ecosystems. For the discipline to advance in the future it will need to enhance, refine, and embrace several conceptual frameworks including the human ecosystem model (Pickett *et al*. 2001, 2008), urban avoiders and adaptors model (McKinney 2002, 2008), ecosystem services (Bolund & Hunhammar 1999; Jim & Chen 2009), sustainability (Newman & Jennings 2008), and resilience (Berkes & Folke 1998; Alberti & Marzluff 2004; Pickett *et al*. 2004).

Historically, urban ecologists have focused on terrestrial environments in developed countries (Chapman & Underwood 2009), but new and exciting advances in understanding the ecological and human dimensions of urban ecosystems will come from the study of remnant vegetation (Florgård 2004), landscaped gardens (Gaston *et al*. 2005; Felson & Pickett 2005; Tratalos 2007), green roofs (Oberndorfer *et al*. 2007), marine environments (Chapman & Underwood 2009; Chapman *et al*. 2009), freshwater ecosystems (Paul and Myers 2001), and studies of human health (Frumkin *et al*. 2004, Tzoulas *et al*. 2007), aesthetics and recreation (Tyrväinen 2003), as well as urban environments in developing countries in general (Conceicao 1994; Escobedo *et al*. 2006; Cilliers *et al*. 2009; Song & Gao 2008).

A significant amount of the research conducted on urban ecosystems has been directed at the population and community level with most of these studies focused on the distribution and abundance of organisms (i.e. patterns). Shochat *et al*. (2006) propose the need to develop more mechanistic studies of urban ecosystems. Urban ecologists will discover rich veins of research opportunities in the future if they explore the study of such subjects as biogeochemistry (Kaye *et al*. 2006), biodiversity (McKinney 2002, 2008; Luck 2007), urban forestry (McPherson 1997; Nowak *et al*. 2003), and landscape ecology (Forman 2008) in urban environments. I also encourage students and practitioners of urban ecology to refine existing, and develop new, tools so that they can participate in the development of solutions to solve the many problems facing human settlements today. Examples of useful tools that need future examination and application include the UFORE model that assesses the ability of urban forests to reduce air pollution and ameliorate local climate (Nowak *et al*. 2003), the enhancement and analysis of new GIS methods (Cadenasso *et al*. 2007; McDonnell & Hahs 2009), and the application of non-parametric Bayesian statistical tools (McCarthy 2008) in the design and analysis of experiments.

Finally, to truly advance the discipline of urban ecology requires the creation of new hypotheses and the identification of confirmed generalizations (McDonnell & Hahs 2009; Pickett *et al*. 2009). Confirmed generalizations are formed when a body of tested facts results in a new, universally accepted, level of understanding of how urban ecosystems are structured and function. For example, one of the most compelling confirmed generalizations in urban ecology is the relationship between the amount of impermeable surfaces in a watershed and the health of streams (Paul & Myer 2001). When a watershed is composed of more than 20 per cent impermeable surfaces there is a significant decrease in stream biota and health (Paul & Meyer 2001). Hence, the challenge to all urban ecologists in the future is the development of a collection of confirmed generalizations or principles which will motivate basic research in the field while also informing practitioners working to design, build, and manage sustainable human settlements in the future.

Summary

Urban ecology emerged as a subdiscipline of ecology in the early 1970s due, in part, to the fact that human impacts on the planet were becoming well-documented and the growing size of human settlements was resulting in serious environmental problems that threatened the health and well-being of both urban and non-urban dwellers around the world. Influenced by these events, and coupled with the demise of the 'balance of nature' paradigm, ecologists have acknowledged that human settlements are legitimate subjects of ecological study. The creation of the UNESCO Man and the Biosphere Program (MAB) in 1974 and the establishment of two urban LTER programs by the US National Science Foundation in the late 1990s were instrumental in encouraging the study of the ecological and social components of urban ecosystems around the globe. Urban ecology has evolved as a unique field of study through the integration of several disciplines that investigate the ecological and human dimensions of urban ecosystems. Urban ecologists can be engaged in basic (i.e. fundamental) research focused on understanding the structure and function of urban environments, or they can be engaged in applied (i.e. practical) research that is focused on solving important environmental problems. The discipline of urban ecology is at the forefront of creating the knowledge base, conceptual frameworks, and tools that are crucial for building and maintaining sustainable and resilient cities and towns in the future.

Acknowledgements

This chapter was significantly improved throughout its development by the input, comments, and suggestions of Amy Hahs, Cynnamon Dobbs Brown, and Julia Stammers. Their assistance is gratefully appreciated. I would also like to thank Jari Niemelä for his helpful comments on the manuscript. The Baker Foundation provided generous support for this research.

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SECTION 1

Ecology in Cities: Man-Made Physical Conditions

SECTION EDITOR: **Jürgen H. Breuste**

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Introduction

Jürgen H. Breuste

The four chapters in Section 1 introduce the physical properties of urban areas that 'set the scene' for urban ecology. This chapter discusses such physical conditions that influence and determine biodiversity in cities as climate, soils, and hydrology. Also, the overall human steering by 'physical' intervention, through the processes of land-use which create new urban morphology, is discussed in this section.

The relation between plants and animals and their surroundings, and the establishment of ecological patterns, would not be understandable without understanding the process of urbanization, and how urban growth and urban densification proceed. It is human intervention through using the land for different purposes, and changing the frequency of interventions and disturbances both in quality and quantity, that extends the influencing area which produces the conditions of urban ecosystems.

With the replacement of the vegetation cover by paved surfaces and buildings the urban surface not only decreases the former surface but introduces new surface materials like concrete, asphalt, stone plates, glass, etc. that are not biologically active. These materials cover the soils and change their functions by having, for example, completely different thermal characteristics and influences on hydrological processes. Additionally, the utilization activities introduce new matters, nutrients, and pollutants into the natural system of the water cycle, and change the air quality. A new physical world is created—the city. However, it is not only a new physical, but also a new ecological, environment. With this section we want to improve the understanding of these preconditions for ecology in cities by analysing human interventions such as land-use, the changed climate, soil conditions, and the hydrological cycle.

The opening chapter by Stephan Pauleit and Jürgen Breuste makes it clear that the metabolism of cities is largely the result of the concentration of people and economic processes. But the physical urban form can only be interpreted by understanding land-use as a continuing process whereby land is involved in human activities and produces surface-cover patterns. Land-uses are particularly diverse, small-scaled, intensive, and influential not only on the targeted urban area but also elsewhere through noise, air pollution, etc. They consist of a mosaic of different, often very small, land-use types as basic elements which form the key tool for undertaking applied urban ecological research and urban nature conservation (Breuste 2002). Urban structural or morphological units and types can be distinguished by their characteristic pattern of built and open spaces (Pauleit & Duhme 2000). It is vegetation cover (as part of the surface-cover) which is a component of almost all urban land-use types responsible for the ecosystem services.

Eberhard Parlow's chapter gives an instructive overview of the urban climate, which is strongly modified by human activities. Most meteorological variables like temperature, heat stress, air pollution, wind, etc. are influenced by the urban conditions. Physical, biological, and chemical aspects, the urban heat island phenomenon, and the impacts of urban climate on human health are discussed. The urban climate system differs from non-urban or rural conditions in various aspects. For example, the high aerodynamic surface roughness (highly three-dimensional and therefore a very complex surface for all exchange processes with the urban boundary layer) which influences the vertical turbulence and wind field. Also, radiation and heat budgets differ due to the physical properties of construction material related to heat capacity and thermal conductivity. Cities are also significant sources of emissions from traffic and industrial sites, as well as heating and air conditioning, in terms of greenhouse gases, pollutants, and direct heat release.

The chapter by Martin Sauerwein examines how urban soils are an important part of the urban ecosystem. Soils form the life basis for many organisms. They are permeated by roots, provide support and warmth, and supply the plants with water, oxygen, and nutrients. Soils also regulate the water balance and filter, buffer, or eliminate contaminants, thus preventing these from harming organisms, entering crops, and polluting the groundwater or nearby bodies of water. These functions are severely curbed in the urban landscape. The urban emissions from different sources lead to a pollutant burden, influencing the soil functionality in the urban ecosystem. The urban soils genesis is explained. Urban soils have rule-like spatial patterns related to urban structural types and with these create urban soil landscapes (see Sauerwein, Chapter 1.3).

Marc Illgen's chapter describes the hydrological processes and phenomena of urban environments. Land-use characteristics explain the major components of the urban water cycle, together with the particular processes occurring between atmosphere, surface, and subsurface. The latest innovations on the specific infiltration and runoff patterns of urban surfaces are presented. The stormwater retention capacities of paved and unpaved surfaces are specified from an urban water management perspective. Characteristic values of the particular water balance components on an annual basis, as well as event-based runoff coefficients, are outlined for several types of surfaces, reflecting the interactions between the atmospheric impact by rainfall and the hydrological phenomena on urban areas.

The research covered in these chapters demonstrates how human interventions (focused on land-use and urban morphology) are important basic criteria for all ecosystem processes. The urban society is 'random' and not always targeted by utilization processes changing the climate, water cycle, and soils in urban ecosystems to reach socio-economic targets. The services of urban ecosystem on which our life quality in cities and towns depends, depends itself on several physical influences related to land-use, and these need much better management.

CHAPTER 1.1

Land-Use and Surface-Cover as Urban Ecological Indicators

Stephan Pauleit and Jürgen H. Breuste

1.1.1 Introduction: urban form and ecosystem processes

The ecology of urban areas is distinctive in a number of ways. High levels of consumption of energy and natural resources (water, minerals, and food) and related emissions of greenhouse gases and pollutants are of particular concern from global and regional perspectives. It has been estimated that 80 per cent of global greenhouse gas emissions stem from cities or activities which may be located elsewhere but are needed to support urban areas. The ecological footprint of the city region of London alone, that is, the theoretical area required to support it with energy and natural resources and to assimilate its waste products, has been estimated to exceed the size of Great Britain in its entirety (Girardet 2004). Clearly, these patterns of consumption are unsustainable and call for a shift towards a more resource intensive and circular metabolism.

The metabolism of cities is largely the result of the concentration of people and economic processes. It is also related to urban form. For instance, per capita consumption of energy is higher in low density cities due to increased car dependence (Newman & Kenworthy 1989). Moreover, levels of resource consumption are not only related to urban density but also to the patterning of human activities within cities. Planning has often separated urban functions, such as living, working, and recreation, and reconnected them by extensive infrastructure networks which have also increased the transport demand and, in particular, car-dependency of the city. Therefore, development of modes of mixed use in the compact city has become an important goal in current urban planning in the Western world.

The environment within cities is also distinctive (see also Alfsen *et al*. Chapter 4.3). Well known and oft described phenomena are urban climates (notably the 'urban heat island' effect, the elevated temperatures compared to the city's surroundings) (see Parlow, Chapter 1.2); altered hydrology such as negative impacts on surface waters, increased stormwater runoff, and reduced groundwater formation (see Illgen, Chapter 1.4); degraded soils (e.g. Craul 1999) (see Sauerwein, Chapter 1.3), and changes in biodiversity (e.g. Sukopp & Wittig 1993; Bridgman *et al*. 1995) (see also Section 2 and 3 in this book). These distinctive characteristics of the city are a consequence of the fundamental changes that happen to the land when urban areas are developed. Mainly, natural areas, agriculture, and forests are replaced by predominantly built surfaces. Moreover, human activities in cities are often very intense, causing the abovementioned phenomena as well as particular disturbance regimes.

Urban climates, hydrology, and biodiversity, are not uniform across cities but vary widely within cities. Ecological studies have shown the significant changes that take place along transects from rural to inner-city for a number of environmental parameters, including aspects of biodiversity, climate, soils, and hydrology (e.g. McDonnell *et al*. 1997; Alberti 2008). Changes are usually not gradual but are related to the distribution of human activities and urban physical structures in space. These relationships will be further explored in this chapter.

Attention will also be drawn towards current land-use and land-cover dynamics which cause concern because of their environmental impacts. One important caveat in this chapter is that information is almost exclusively drawn from European studies. Certainly this does not do justice to the global phenomenon of urbanization but, nevertheless, we hope that it will give some useful insights into the role of land-use and surface-cover as key indicators of the ecosystem process in the city.

1.1.2 Land-use and surface-cover patterns in urban areas

Land-use is not a state of being but rather the process whereby land is involved in human activity. This process can be driven by individuals, specific groups, or the community at large. The result is a used piece of land which can be typified as representing a land-use type. Land-use types represent the different anthropogenic activities in a given area. The typical land-use process can be identified by changes on the Earth's surface. The visible results of land-use as a process are the characteristics and elements of vegetation and/or built-up structures on the surface. This physical material at the surface of the earth is called *surface-cover (*synonymous with *land cover*).

Ecosystem services, that is 'the benefits people obtain from ecosystems' (MEA 2005; McDonald and Marcotullio, Chapter 4.1.; Alfsen *et al*. Chapter 4.3) are clearly related to land-use and land cover, both generally and specifically in urban areas (e.g. Breuste *et al*. 1998). Therefore, spatial planning and regulations that influence the spatial pattern and intensity of land-use can have huge implications for the ecology of cities (e.g. Alberti 2008). Land-use is influenced both by societal and environmental processes and patterns (Fig. 1.1.1). In turn, land-use influences ecological patterns and processes in cities which themselves lead to changes in ecological conditions and the broader environmental context, such as formation of the urban heat island. Changes in ecological conditions may affect human perceptions and attitudes and influence the formulation of policies which have an impact on land-use.

Land-uses are both particularly diverse and intensive in urban areas. They consist of a mosaic of different, often very small and fragmented land-use types (Breuste 2002, p. 410) which accommodate basic human needs for facilities in the residential, industrial and commercial, administration, education and other social institutions, and recreation sectors. Urban areas within their administrative boundaries can also harbour larger areas of agricultural and forest land as well as unused or derelict land. For analysis of the ecological conditions of urban land, a more detailed characterization of urban land-use is required. Two related approaches shall be briefly presented here:

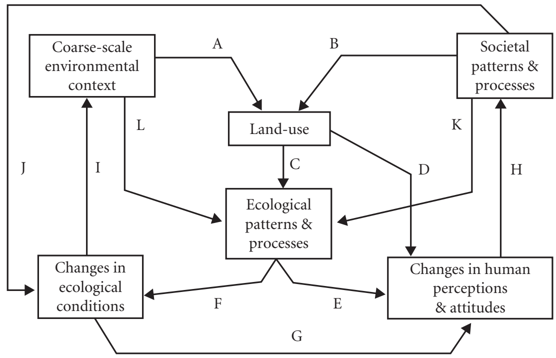

Figure 1.1.1 Conceptual scheme for integrating ecological and social systems in urban environments. Variables are in boxes; interactions and feedbacks are arrows: A, environmental context sets the range of possibilities for land-use–land cover; B, societal decisions and human behaviour (incorporating 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 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) (Grimm *et al*. 2000). Used with permission from American Institute of Biological Sciences

1.1.2.1 Biotope mapping

In Germany in the 1970s, biotope mapping had already emerged in order to provide detailed ecological information to planners. It was first applied in the rural countryside, and then this approach was transferred to cities (Lachmund 2004). Much of the early research was based on the premise that land-use is the most important and fundamental process influencing plants and animals and their communities. 'Within the settled areas there are primarily the utilization forms, which are dominating the pattern and distribution of organisms. Basis of the nature protection work in the city is therefore to analyze the most important types of land-use systematically and to describe species content and ecological characteristics. In the final result it becomes clear, which land-use forms are extraordinarily poor of species and demand management for re-implementation of nature' (Sukopp, Kunick, & Schneider 1980, p. 565).

Thus, land-use became the key tool for undertaking applied urban ecological research and urban nature conservation. An entire field of pure and applied research was developed—urban habitat mapping—and the research relied heavily on land-use as the basic element. Urban land-use types or habitat types have become the general reference units for urban ecological conditions since the 1970s. Mapping legends were developed and recommended for a broad, comparative application in urban nature conservation (Arbeitsgruppe Methodik der Biotopkartierung im besiedelten Bereich 1986, 1993). These maps were powerful tools that changed the perception of urban areas from an ecological perspective (Lachmund 2004) as the city, which was often considered to be purely artificial, was reconstructed as an ecological space. Maps showed different habitats which were not restricted to remnants of pre-urban nature but provided a differentiated picture of nature in the city. Different land-use types had characteristic assemblages of species that could be related to human influence, sometimes expressed as 'hemeroby'. In a second step, evaluations were attempted to support strategies for nature conservation in the city. However, the utilization of urban land-use types as ecological units started without the inclusion of available scientific information from landscape research (Breuste 2009, p. 359).

A further limitation of biotope mapping was that it concentrated mainly on describing urban nature, that is, the different associations of plants and animals in the city. The approach was implicitly based on an *understanding* of the city as being an ecosystem, but the *analysis* of the city as an ecosystem was not further pursued.

1.1.2.2 Urban structural types/urban morphology types

Urban structural types (Strukturtypen in German) or urban morphology units and types (UMTs) are the product of past and present human land-use activities and 'can be distinguished by their characteristic pattern of built and open spaces' (Pauleit & Duhme 2000, p.2). 'The underlying assumption is that UMTs have characteristic physical features and are distinctive according to the human activities that they accommodate (i.e. land-uses). Physical properties and human activities are assumed to be key factors that largely determine the ecological properties of urban areas' and 'Studies have suggested that the distinction of UMTs or urban structural types at a "meso"-scale (i.e. between the city level and that of individual plots) is a suitable basis for the spatial analysis of cities for urban planning' (Gill *et al*. 2008, p. 211 with reference to numerous authors) (Fig. 1.1.2).

This approach has been applied, for instance, in the German cities of Leipzig and Halle, Munich,

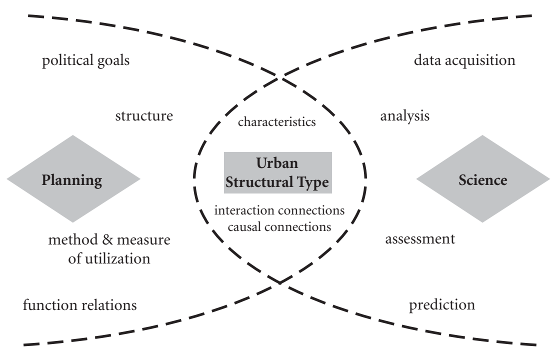

Figure 1.1.2 Urban structural types as a bridge between the science of urban ecology and urban planning (Breuste 2006)

and recently in Greater Manchester (UK) (Breuste 1994; Wickopp 1998; Böhm 1998; Pauleit & Duhme 2000; Gill *et al*. 2008). Urban morphology units were delineated from a survey of aerial photographs and grouped into a number of types. These types were derived from, or closely related to, typologies used in land-use planning and monitoring which facilitate integration of ecological information into these procedures. For instance, the typology used for Greater Manchester (Gill *et al*. 2008), which is shown in Fig. 1.1.3, was adapted to be compatible with the national land-use classification (NLUD 2003). Subdivisions of land-use classes, in particular types of open space, further differentiate this national typology.

Despite being a heavily urbanized area, the single most important morphology type in Greater Manchester is farmland (39 per cent) followed by medium density residential (23 per cent), the latter being the most important urban morphology type when farmland is discounted. While the data refer to a single case, the results are broadly the same in other English settlements, as a survey based on a similar methodology has shown (LUC 1993). In addition, data from US cities show comparable patterns (Nowak *et al*. 1996).

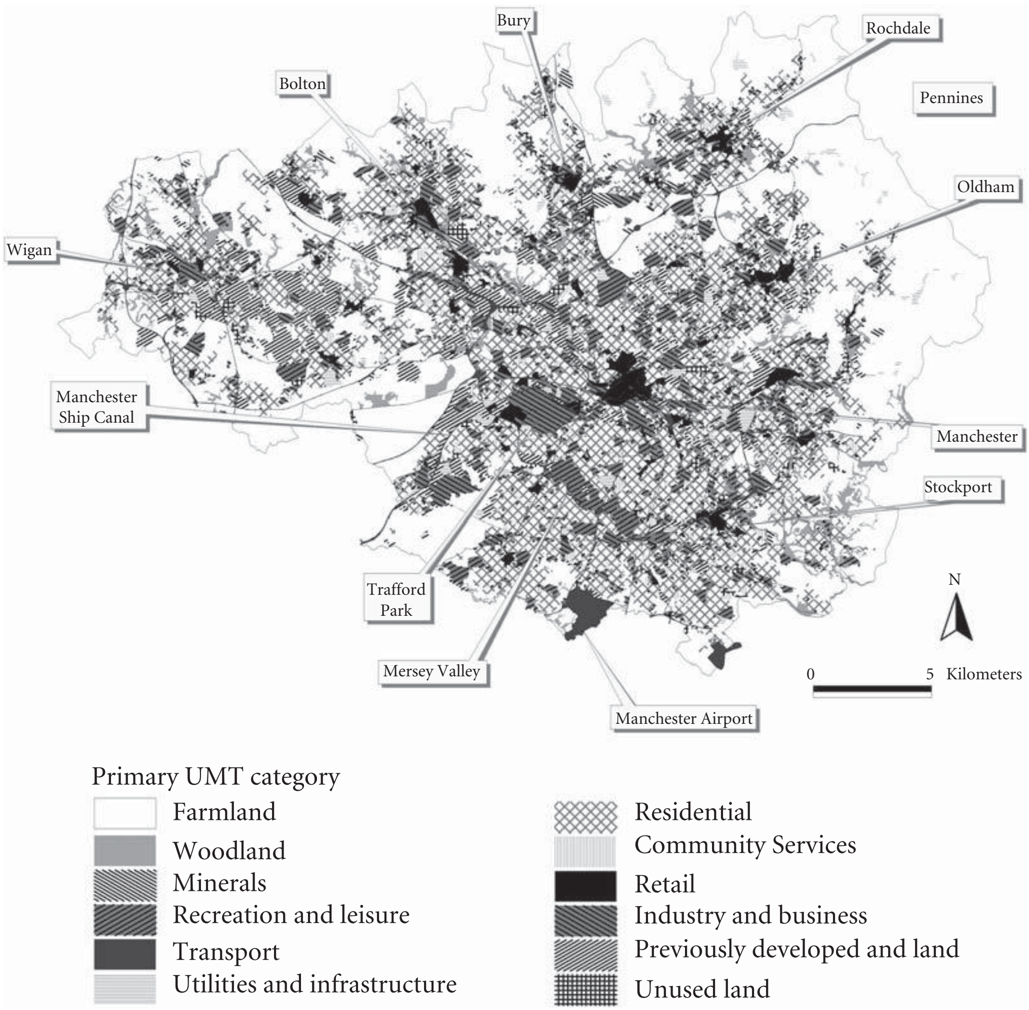

Figure 1.1.3 Urban morphology type map for Greater Manchester, UK (Gill *et al*. 2008). Used with permission from Elsevier

1.1.2.3 Surface-cover: built vs. green surfaces

For urban ecological analysis, urban morphology types need to be further characterized, in particular by surface-cover. Surface-cover is an important determinant of ecosystem process in cities, influencing elements such as climatic energy exchange and hydrology (e.g. Pauleit & Duhme 2000; Gill *et al*. 2007; Alberti 2008). Surface-cover can be easily surveyed from remote sensed data (aerial photographs or satellite imagery, for methodological discussions see Pauleit & Duhme 2000 and Gill *et al*. 2008, Fig. 5). The relationship between built and paved (i.e. 'sealed') spaces on the one hand, and vegetated surfaces on the other hand, is an important determinant of ecosystem process in urban areas. Soil sealing is usually used as general indicator of extreme physical anthropogenic influence on ecosystems. There are also attempts to connect the term with ecologically relevant functions (Sauerwein, Chapter 1.3). Table 1.1.1 summarizes some of the key impacts of replacement of vegetated surfaces by 'sealed' surfaces.

Figure 1.1.4 shows the differences in surface-cover composition between the various urban morphology types in Manchester. In this diagram, built and paved surfaces were aggregated into 'built surfaces' while different types of green surfaces were aggregated into 'evapotranspiring surfaces'. The differences are striking. Town centres, retail areas, manufacturing, major roads, but also highly dense residential areas, have a built cover of more than

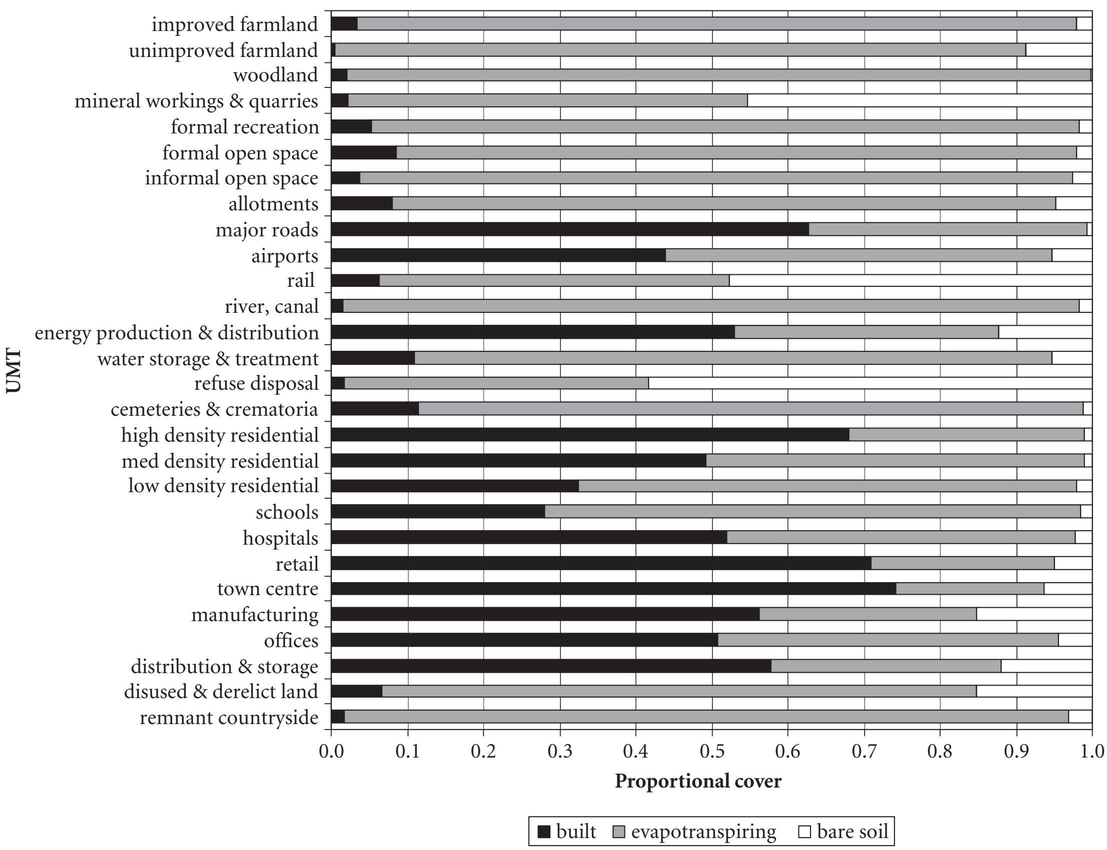

Figure 1.1.4 Proportional surface-cover grouped into three broad categories (built, evapotranspiring, and bare soil) for the UMT categories in Manchester (Gill *et al*. 2008). Used with permission from Elsevier

Table 1.1.1 Key impacts of replacement of vegetated surfaces by 'sealed' surfaces (Breuste *et al.* 1996):

Soil and water regime (through 'loss' of the vegetation cover and physical change of the soil surface and the upper soil layer)
• partial or complete removal of the upper soil layer; • decreased infiltration of precipitation water into the soil and thus reduced groundwater replenishment; • renewal; • increased evaporation; • increased and accelerated rates of stormwater runoff; • more frequent high levels in drains and streams with heavy rain and thaw.
Urban climate (through 'loss' of the vegetation cover and thermal and energetic effects due to human modifications (creation of new technical surfaces))
• increased thermal capacity and thermal conductivity of the sealing materials; • increased air temperatures; • increased particulates, and thus more frequent precipitation events; • lower volume and shorter periods of snow cover; • reduced humidity in temperate regions (not always true for arid regions).
Vegetation and fauna (through destruction of the vegetation cover and change of the local ecological conditions, intensive use by trampling and driving on)
• reduced, usually minimal colonization opportunities for plants; • lower oxygen and water supply for soil fauna, and decreased exchange of matter and gases between the soil and the near-surface air layer; • depletion of the native flora; • loss of levels of the food pyramid; • loss of habitat; • increasing isolation of populations.

50 per cent and a correspondingly low percentage cover of evapotranspiring surfaces. The percentage of evapotranspiring surfaces was two times higher in low density housing than in high density housing (66 per cent and 31 per cent, respectively). Urban greenspaces (formal and informal open spaces, formal recreation, etc.) and other green structures, such as woodlands and farmland, have a very high percentage cover of evapotranspiring surfaces (more than 90 per cent each).

The spatial distribution in Greater Manchester of built surfaces on the one side, and evapotranspiring surfaces on the other, can be seen from the map in Figure 1.1.5. Densely built areas with a very low cover of evapotranspiring surfaces (below 20 per cent) are located in town centres, in large industrial and commercial areas, and in transport infrastructures. Well-greened residential areas surround the urban cores. Urban greenspaces become visible as green patches in the urban matrix. The cover of evapotranspiring surfaces is also high on the urban fringe where farmland predominates.

The distribution of greenspaces within residential areas, and that of trees and shrubs specifically, is not uniform but instead clearly related to their socio-economic status. Other studies in Europe and the USA have shown that cover of green areas, and in particular tree cover, is higher in high status residential areas than in deprived areas (e.g. Iverson & Cook 2000; Pauleit *et al*. 2005a; Tratalos *et al*. 2007; see also Cilliers and Siebert, Chapter 3.2).

The figures from the Manchester study are broadly supported by the results from other research. Fifty-seven per cent of 'urbanized' Greater Manchester (i.e. discounting farmland) is covered by evapotranspiring surfaces. This falls within the range of values for US cities reported by Nowak *et al*. (1996) where vegetation covers between 38 and 93 per cent. Forty per cent of Greater Manchester's evapotranspiring surfaces were found in residential areas, while only approximately 19 per cent were found in the various greenspace types, once farmland is discounted. Another approximately 12 per cent of evapotranspiring surfaces can be found on the derelict land of former industrial areas in Greater Manchester, and 9 per cent in woodlands.

Thus, public greenspaces certainly play an important role in overall greenspace resource, in particular as they are publicly accessible. However, quantitatively, residential areas are still the biggest greenspace resource. A survey in five British cities and towns showed that private domestic gardens covered between 21.8 to 26.8 per cent of the urban area (Loram *et al*. 2007). Big differences in tree and shrub cover could also be observed between urban

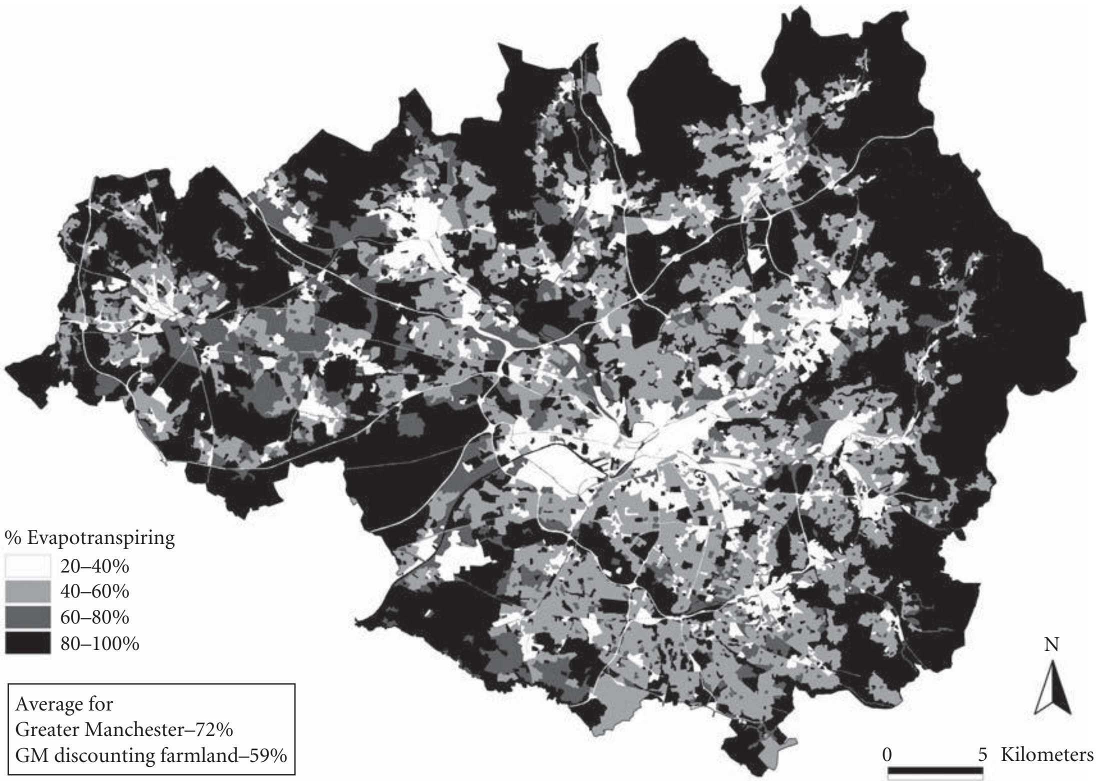

Figure 1.1.5 Proportion of evapotranspiring surfaces in Greater Manchester as estimated from UMT sampling (assuming the average from the photo interpretation) (Gill *et al.* 2008). Used with permission from Elsevier

morphology types in Greater Manchester and elsewhere (e.g. Pauleit & Duhme 2000; Leipzig: Table 1.1.2). The overall average cover of 16 per cent in 'urbanized' Greater Manchester compares with an average cover of 27 per cent in US cities (minimum 4 per cent, maximum 55 per cent) (Nowak *et al.* 1996) and 16 per cent in Munich (Pauleit & Duhme 2000).

Only data on the aggregated indicators of 'sealed surfaces' and 'evapotranspiring' surfaces was presented here. More detailed analysis is required to establish the relationships with biodiversity and ecosystem processes. The main characteristics of the vegetation cover which are ecologically relevant are the layer structure, especially the tree and bush cover, and the intensity of utilization and maintenance (e.g. Greiner & Gelbrich 1975).

The vegetation cover (as part of the surface-cover) is a component of nearly all urban land-use types (urban structural units). Some units are dominated by designed vegetation cover (e.g. parks and allotments) in others the vegetation cover is an additional decorative element (e.g. residential areas). Other vegetation covers establish spontaneously after finished or interrupted utilization of land over a longer time (derelict land).

In Central Europe, cities grew and still grow into a cultural (agricultural and only partly forested) landscape. The vegetation cover of open spaces within urban areas ranges from vegetation remnants of the original natural landscape (mainly woods and wetlands), vegetation of the cultural landscapes formed by agriculture (e.g. meadows and arable land), ornamental, horticultural, and designed urban vegetation spaces (parks and gardens), to spontaneous urban vegetation (brown fields and derelict land). These four main groups of vegetation cover are the result of different land-use

Table 1.1.2 The urban structural or land-use type of older areas of villas, compared to the city centre in Leipzig, show very different forms of land-use (modified from Breuste 2009)

	old villa areas	city centre
		
Land-use	Residential	Residential plus commercial and offices
Urban morphology type	Single houses	Compact building blocks
Building cover	20–30%	More than 70%
Open space character	Extended open spaces	Small open spaces in courtyards, some squares, streets
Vegetation character	High vegetation cover, esp. tree cover	Almost no vegetation cover
Cover of sealed surfaces	Below 40%	Above 90%

forms (functions) and intensities of utilization and maintenance (Table 1.1.3).

Further metrics have been suggested to analyse how spatial patterns in the urban landscape relate to ecosystem processes. For instance, Alberti (2008) proposed four key dimensions: form, density, heterogeneity, and connectivity. These dimensions have been shown in landscape ecological research to influence the ecosystem process, but further research is required for their validation.

1.1.3 Land-use and surface-cover dynamics in urban areas and their ecological implications

Two main processes of urban physical change are presented here to facilitate discussion on the temporal dimension of urban land-use change: (1) urban growth/sprawl, and (2) urban densification with reference to European cases.

1.1.3.1 Urban growth and sprawl

In Europe, urban areas have on average extended by 78 per cent between 1950 and 1990 whereas population has increased by 33 per cent (EEA 2006). The growth of urban areas and associated infrastructure throughout Europe consumed more than 8,000 km^2 of land in the period between 1990 and 2000 (a 5.4 per cent increase during the period). Less than 10 per cent goes the opposite way, that is, is transferred from urban land into brownfields, and only a small part of this is reclaimed for arable land-use or nature. These figures are based on the CORINE land cover survey and a study which included a sample of urban areas all over Europe. They were produced in a comparable way, based on interpretation of satellite imagery and aerial photography. The results indicated that urban areas are becoming more dispersed—a process also called urban sprawl. A north–south gradient could be observed across Europe, with cities in Northern Europe being less dense on average than Southern European cities when measured in terms of residential density (Kasanko *et al*. 2006). However, while the latter have traditionally been more compact, trends of sprawl were particular pronounced in the south, indicating that the differences are blurring.

Urban sprawl has induced change to some important characteristics of urban form and land-use structure in European cities:

Table 1.1.3 Types of urban vegetation structures—influenced or created by urban land-use (Breuste after Arbeitsgruppe Methodik der Biotopkartierung im besiedelten Bereich (1993) and Kowarik 1992, modified)

Vegetation Group	Vegetation structure type	Main utilization	Main potential functions
A) Vegetation remnants of the original natural landscape	Woods and forests	Recreation, biodiversity	Timber production
	Wetlands	Nature protection, biodiversity	Nature experience
B) Vegetation of the cultural landscapes formed by agriculture	Meadows, pastures	Agriculture	Recreation, biodiversity
	Drifts, dry grasslands	Agriculture	Recreation, biodiversity
	Arable land	Agriculture	
C) Ornamental, horticultural and designed urban vegetation spaces	Decorative green (flower beds, small lawn patches, bushes, hedges, etc.)	Decoration	Recreation, biodiversity
	Accompanied green along traffic lines or as an addition to fill up the space between apartment blocks	Decoration	Recreation, biodiversity
	Gardens/parks	Recreation, decoration	Biodiversity
	Allotment gardens (territorially organized in allotment garden estates)	Recreation	Biodiversity
	Urban trees	Decoration	Biodiversity
D) Spontaneous urban vegetation (areas)	Spontaneous herbaceous vegetation	None	Biodiversity, nature experience, recreation
	Spontaneous bush vegetation	None	Biodiversity, nature experience, recreation
	Spontaneous pre-forest vegetation	None	Biodiversity, nature experience, recreation

- Residential areas have grown faster on the urban fringe than in urban core areas.
- Commercial and industrial areas, as well as their infrastructure, have extended much faster than residential areas. Again, this growth has been stronger on the urban fringe than in urban core areas.
- Green urban areas have grown on the urban fringe but declined in urban core areas.
- Farmland and areas classified as 'natural' have declined and become more fragmented.

The phenomenon of urban sprawl has received considerable academic interest (e.g. Haase & Nuissl 2007). However, its consequences on the ecosystem process have only rarely been substantiated in a rigorous scientific way. Some of the major environmental problems associated with urban sprawl are (Gayda *et al*. 2003):

- Loss of natural areas and biodiversity, fragmentation, and degradation of remaining natural areas.
- Loss of farmland and productive soils.
- Negative impacts on hydrology: for example, deterioration of surface water quality and increased stormwater runoff.
- Increase of air pollution, in particular through more individual traffic; extension of the urban heat island.
- Increase of energy consumption from traffic.

More in-depth research into the effects of different spatial patterns of urban sprawl and specific land-use changes is still largely missing, though there are some exceptions, such as a study in the Puget Sound region, USA (Alberti 2008). Urban sprawl would be an important area for future urban ecological research.

1.1.3.2 Urban densification

The densification or compaction of urban areas has been suggested as a strategy to avoid or reduce urban sprawl. Densification means that available land within urban areas is built on, rather than farmland or natural areas on the urban fringe. Cities often offer many opportunities for densification, for instance, the redevelopment of former industrial areas which have fallen into dereliction as the old industries have gone. Derelict land can cover significant amounts of land in urban areas—in the case of Greater Manchester approximately 10 per cent of the land surface. Another process of densification is in-fill development, where further houses are built into low-density residential areas.

However, densification also means that inner urban open spaces are built over, with negative consequences for ecosystem process (e.g. Pauleit *et al*. 2005a). Moreover, wastelands which have been disused over longer periods of time can provide valuable wilderness areas for recreation in the city. In old industrialized cities they are often located in proximity to low-income residential areas with local provision of both residential and public greenspaces. Their destruction, therefore, may have serious consequences for quality of life in the city. Unfortunately, only few studies have been published that have attempted to monitor urban densification and quantify its impacts on ecosystem process.

In research in Merseyside, UK, land-use changes were studied in 11 residential areas of varying socio-economic status over a period of 25 years by interpretation of aerial photographs from 1970 and 2000, respectively (Whitford *et al*. 2001; Pauleit *et al*. 2005a). Land-use and surface-cover were recorded on a 50 by 50 m quadrant for 11 areas of 500 by 500 m. Four models were used to quantify the impacts of land-use and surface-cover change: surface temperatures, surface runoff, carbon sequestration and storage, and biodiversity. For the latter, diversity of green pace structures (trees, shrubs, rough grassland, lawns, etc.) was used as a proxy (for details see Whitford *et al*. 2001).

In all 11 residential areas over the 25-year study period, built and paved surfaces increased while green cover was lost. Land-use and surface-cover changes were caused by different processes, for instance, new housing or retail on wastelands, replacement of smaller detached houses by blocks of flats, and paving over of front gardens to provide space for parking cars. In the selected study areas, most of the changes were small when seen in isolation but amounted to significant changes over the period of 25 years. On average, 5 per cent of additional areas were built over or paved. Interestingly, all areas lost greenspace, even though

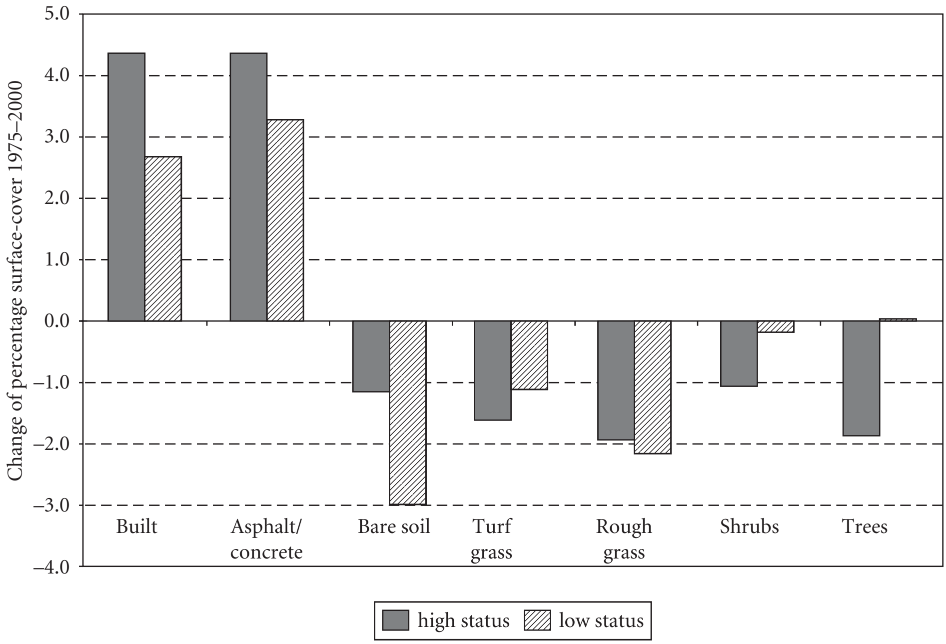

Figure 1.1.6 Land cover changes in the 11 study areas in Merseyside (Pauleit *et al.* 2005b)

the socio-economically deprived areas already had a low cover of greenspaces in the 1970s. Losses were partly caused by efforts to regenerate the more deprived areas, for instance by developing new residential areas and facilities. Obviously, little consideration had been given to landscape issues in these programmes. The larger quantitative losses, however, happened in high status areas where land prices are high. Here, in particular many mature trees were lost due to in-fill development (Fig. 1.1.6).

Impacts of these changes on ecosystem processes were marked (for details see Pauleit *et al*. 2005a):

- Maximum daily surface temperatures on a chosen hot summer day increased on average by 0.9 °C.
- During a rainstorm event of 10 millimetres rainfall in one hour, surface runoff showed a significant increase of 4 per cent.
- A Shannon index of structural diversity was calculated as a biodiversity indicator for the study areas. A significant decline of the indicator values was observed in all study areas over the 25-year period, indicating an overall loss of biodiversity.

Results from the study were limited to a smaller sample of residential areas, and therefore they cannot be considered as representative for the entire Merseyside region, and even less for other city regions. However, they do provide an impression of how urban compaction may negatively impact on the urban environment. This leaves urban planning with a dilemma: urban sprawl is clearly undesirable from an environmental perspective because it increases the urban footprint. But urban compaction, which has been proposed as a strategy to reduce urban sprawl, can have negative impacts on the quality of life and health of urban citizens.

1.1.4 Conclusions

Urban areas are a much changed environment. The ecological footprint of cities, on the one hand, and impacts on environmental quality and the ecosystem process within cities, on the other, are of particular concern for urban planners. In this chapter, it has been argued that a deep understanding of the relationships between urban form and human activity with the ecosystem process is important to find solutions for these problems. Land-use and land cover are crucial in this respect because they have a fundamental influence on the urban ecosystem process and can be controlled to some degree by urban planning.

The relationships between land-use and surface-cover with the ecosystem process were exemplified for selected ecosystem processes. This is certainly a restriction, as the cases may not be representative for all urban areas and all types of ecosystem process. A further, particular, restriction is that only cases from the European continent were chosen. Nevertheless, it is hoped that the chapter gives some interesting insights into the topic. It seems particularly important to highlight:

- The importance to the urban matrix of different types of residential, commercial, and other land-uses where most of the urban green surfaces can be found. The amount and character of these green surfaces determines ecosystem processes such as the climatic energy balance, stormwater runoff, carbon storage, and biodiversity. Great variation between urban morphology types does exist in terms of green cover and character and related ecosystem process. Socio-economic status is an important determinant. The spatial variation of urban structure and ecosystem process does not follow simple patterns, for example concentric zones, but is a more complex mosaic.
- Thus, land-use and surface-cover are important indicators of urban ecological conditions. However, land-use types or land-use zones as used in urban planning require further characterization of their ecological functionality. Furthermore, although the total amount of green area within land-uses, such as residential areas, is usually not recorded at all, it is of great importance for the ecosystem process in cities. Basic questions of how much green cover there is and what character it has often cannot be answered. Finally, land-use plans combine the current state in existing built areas and the intended outcome of future land development (e.g. areas designated for new housing, etc.). Therefore, they do not give a reliable picture of what there is in terms of land in order to assess the current status of urban areas from an ecological perspective.
- In this respect, biotope mapping and urban morphology-type survey appear to be two suitable complementary approaches to analyse the fine-grained urban spatial pattern and its relationship with the ecosystem process. This will allow urban planning to identify areas in need of protection or restoration of ecosystem processes, development of spatially explicit strategies for urban environmental planning, and assessment of alternative strategies.
- The approaches presented here promote the understanding of the consequences of land-use dynamics. Two major processes were discussed here: urban sprawl and densification. Both may have negative environmental consequences. This leaves planners with a dilemma. While urban compaction seems desirable in order to reduce consumption of land, as well as energy consumption and greenhouse gas emissions, it may compromise the ecological quality in the city and its adaptive capacity to climate change (see Gill *et al*. 2007). Further study is required to identify patterns of ecological sustainable urban development.

Finally, the mapping and modelling approaches presented in this chapter are simple and easy to apply. Other research, for example by Alberti (2008), suggests further analysis with spatial metrics and transect methods, while the conceptual model by Grimm *et al*. (2000) indicates how these approaches can be embedded into a comprehensive analysis of pattern, process, and dynamics of the urban ecosystem, integrating human and natural forces. Particularly suitable are modelling approaches combined with methods of sustainability impact analysis that allow one to assess the likely consequences of different planning scenarios on ecosystem services. One such approach is currently being developed in a large European research project called PLUREL (www.plurel.net) which aims to develop strategies and tools for sustainable land-use systems in rural–urban regions. Such integrative research is urgently needed to underpin policies for urban sustainability and indeed for global sustainability.

CHAPTER 1.2

Urban Climate

Eberhard Parlow

1.2.1 Introduction

Worldwide, the urban climate has received increasing attention for research and applications during recent years. The public and political discussion on climate change and how it might influence our well-being has made people more sensitive to ecological aspects in many countries. Urban climate has also become a political issue because until now mankind lived and worked primarily in rural areas. But the world is about to leave its rural past behind: by 2008, for the first time, more than half of the world's population, 3.3 billion people, lived in towns and cities. The number and proportion of urban dwellers will continue to increase dramatically. The urban population will grow to 4.9 billion by 2030. At a global level, all future population growth will thus be in towns and cities (United Nations 2006).

The urban climate is strongly modified by human activities. Most meteorological variables, such as temperature, heat stress, air pollution, wind etc., are influenced by it. The urban climate system differs from non-urban/rural conditions in various aspects:

- The urban area has a high aerodynamic surface roughness which influences vertical turbulence and the wind field.
- It has a completely different radiation and heat budget due to the physical properties of construction material, like heat capacity and thermal conductivity.
- It is normally highly three-dimensional and therefore a very complex surface for all exchange processes with the urban boundary layer.
- It is a significant source of emissions from traffic and industrial sites as well as through heating and air conditioning in terms of greenhouse gases, pollutants, and direct heat release.

During the last decade urban planning authorities have become more aware of these factors and are increasingly integrating the urban climate aspect in their planning processes. This has led to a strong increase in both basic and application oriented research in the field of urban climatology during the last decades to meet the requirements of public authorities and political decision-makers. Climate change and the dramatic growth of urban agglomerations, especially in the Third World and Countries in Transition, are additional driving forces for the consideration of the urban climate on a global scale.

1.2.2 Physical aspects of urban climate

Analysing the urban climate needs some consideration of the urban radiation and heat budget, which directly controls the meteorological fields of temperature and wind as well as the bioclimatic situation during the day.

In general, the radiation budget of an area can be written using the following formula:

$$Q^* = E_{sd} - E_{su} + E_{ld} - E_{lu}$$

with: Q^* net all wave radiation,

E_{sd} shortwave downward solar irradiance,

E_{su} shortwave upward radiation (reflection),

E_{ld} longwave atmospheric counter radiation, and

E_{lu} longwave upward surface emission.

In urban systems the solar irradiance is strongly influenced by the three-dimensional shape of the city, for example the width of streets, height and type of buildings, type and size of urban greens etc., which produce shadowing effects or multiple reflections in the street canyons. Christen and Vogt (2004) measured the vertical variations of solar shortwave irradiance due to shadowing of surrounding houses on a flux tower in a street canyon in Basel, and could clearly show that E_{sd} is not equally distributed within individual layers of the urban canopy. Shading and exposure of the buildings and structures greatly influence its vertical distribution and, therefore, a vertical divergence of the shortwave radiation flux density could be observed.

The shortwave upward radiation depends on the surface material and the reflection characteristics. The ratio of E_{su}/E_{sd} is known as albedo, and describes the reflectivity of the surface integrated over the spectrum of solar radiation. It varies between 0 (no reflection, black) and 1 (total reflection, white). Table 1.2.1gives an overview of some albedo values for typical rural and urban surfaces (Stull 1995).

Christen and Vogt (2004) report annual totals of albedo from two urban and one rural/agricultural site measured during the BUBBLE-experiment carried out in Basel/Switzerland in 2002, during snow-free conditions, of 0.107 (urban) and 0.197 (rural) indicating lower values of urban surfaces.

The longwave downward radiation (atmospheric counter radiation) E_{ld} is an important radiative energy source. In clear sky conditions its value mostly depends on air temperature and water vapour in the lower atmosphere. It does not change very much during the daytime because air is well-mixed by wind and turbulence. Normal daily sums of atmospheric counter radiation in Central Europe are in the range of 30–35 MJ $d^{-1}m^{-2}$ during summer, which corresponds to a mean radiation flux of 350–400 Wm^{-2}. Winter values are lower at roughly 25 MJ $d^{-1}m^{-2}$, corresponding to a mean flux density of 290 Wm^{-2}.

Table 1.2.1 Albedo of different surface types

Surface type	Albedo
Soil, dark wet	0.06–0.08
Soil, light dry	0.16–0.18
Stones	0.2–0.3
Forest, coniferous	0.05–0.15
Forest, deciduous	0.10–0.25
Grass, green	0.26
Rock, granite	0.12–0.18
Road, asphalt	0.05–0.15
Buildings	0.09
Concrete	0.15–0.37
Urban, mean	0.15

The most important radiation flux, which leads to significant differences between urban and rural sites, is the upward longwave radiation flux E_{lu}. According to the law of Stefan-Boltzman it is directly dependent on surface temperature.

$$E_{lu} = \sigma\varepsilon T_0$$

With: σ: Stefan-Boltzman-constant (5.67· 10^{-8} $Wm^{-2}K^{-4}$),

ε: emissivity,

T_0: surface temperature in Kelvin.

Figure 1.2.1 shows the spatial distribution of thermal longwave emission of the city of Basel on 8 July 2002 at 10:00 UTC as seen from Landsat-ETM7 imagery with a pixel resolution of 60 m. Data are given in Wm^{-2} and correspond to a surface temperature between 293 K (418 Wm^{-2}) and 311 K (530 Wm^{-2}). The water surface of the River Rhine, bending from the south-east through the city and heading to the north, shows the coldest temperature (20 °C) while sealed surfaces of the city centre or the airport in the northwest warm up to nearly 40 °C averaged over the 60 x 60 m^2 pixel. Most of the agricultural or forested sites are represented in medium grey tones. The artificial urban surfaces warm up most because the thermal properties (heat capacity, thermal conductivity etc.) of the construction material generally used in urban systems, like concrete, bricks and asphalt, are complete different.

Table 1.2.2 gives an overview on heat capacity and thermal conductivity of urban and non-urban material. Heat capacity indicates how much energy is needed to heat a cubic metre of a certain material by 1 degree centigrade. Thermal conductivity is a property which indicates how easily heat can be transported within a material.

Next to the radiation budget, a further important aspect is the urban heat budget, which makes the urban climate special. It describes how the energy

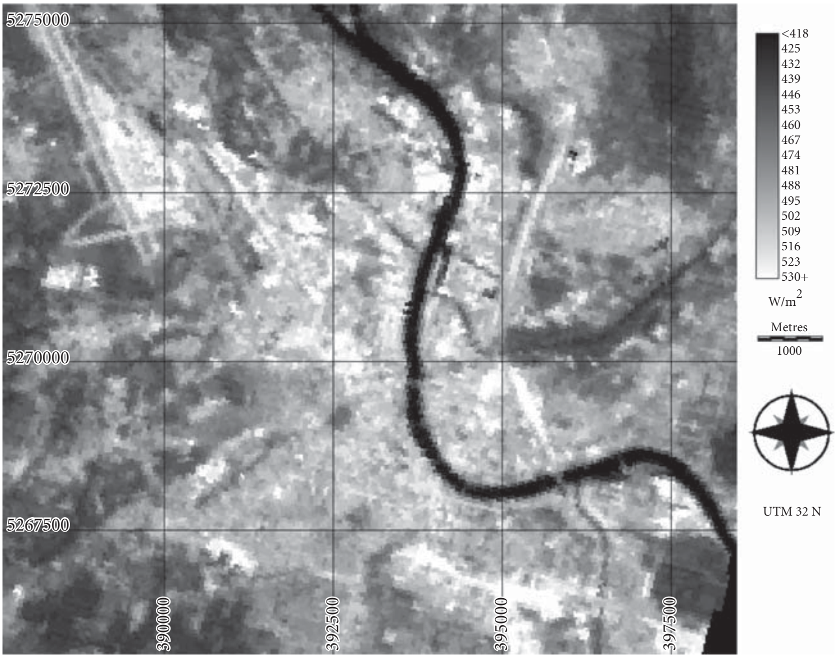

Figure 1.2.1 Longwave terrestrial emission of the city of Basel/Switzerland on 8 July 2002 (10 UTC). Landsat-ETM-7 imagery, band 6. Radiation fluxes are given in W m^{-2}

Table 1.2.2 Thermal properties of materials used in rural and urban environment (Oke 1987)

Material	Heat capacity $Jm^{-3}K^{-1}$ x 10^6	Thermal conductivity $Wm^{-1}K^{-1}$
Sandy soil, dry	1.28	0.30
Sandy soil, saturated	2.96	2.20
Clay soil, dry	1.42	0.25
Clay soil, saturated	3.10	1.58
Asphalt	1.94	0.75
Concrete, dense	2.11	1.51
Stone	2.25	2.19
Brick	1.37	0.83
Clay tiles	1.77	0.84
Wood, dense	1.52	0.19
Water, still	4.18	0.57

gain from net radiation is partitioned into the various heat fluxes.

The heat budget for a surface can be written as:

$$Q^* + Q_H + Q_E + Q_S + Q_F = 0$$

With: Q^*: net radiation,

Q_H: sensible heat flux density,
Q_E: latent heat flux density.
Q_S: storage heat flux density, and
Q_F: anthropogenic heat flux density.

Since a surface has no volume and consequently no mass, the heat budget equation sums up to 0. That means if net radiation is positive, as is normally the case during daytime, this energy is used for sensible heat flux (increase of air temperature), latent heat flux (evapotranspiration), and storage heat flux (flux into the soil and building material). The anthropogenic heat flux Q_F is normally a small source of energy and includes all additional energy input produced by human activities, such as the energy released by combustion of fuels, and electric heat from industry and traffic or through heating and air conditioning. It varies widely according to published data. Christen and Vogt (2004) report an anthropogenic heat flux density Q_F of 5 Wm^{-2} in the suburban neighbourhood and 20 Wm^{-2} in the city centre during the BUBBLE experiment in Basel/Switzerland (Rotach *et al*. 2005), corresponding to 1 to 6 per cent of net radiation. Other publications present a wider range of mean anthropogenic heat flux densities, between 32 Wm^{-2} in Lodz/Poland (Offerle *et al*. 2005), 70 W m^{-2} in winter and 15 W m^{-2} during summer in Toulouse/France (Pigeon *et al*. 2007), and 60–75 W m^{-2} in US-cities during winter and 10–55 W m^{-2} during summer (Sailor & Liu 2004).

Whether the heat flux is positive or negative indicates its direction. Positive heat fluxes are directed towards the surface meaning a gain of heat (decrease of air and soil temperatures, condensation); negative heat fluxes are directed away from the surface (heating air and soil temperatures, evapotranspiration).

Figure 1.2.2a shows the mean daytime and night-time net radiation and its partition into heat flux densities of three rural (R1–R3), one suburban (S1), and two urban (U1, U2) sites from the BUBBLE fieldwork during summer 2002. During this fieldwork, mean daily net radiation varied between 322 and 482 Wm^{-2} in daytime. At the rural sites most of this energy was used for latent heat flux (evapotranspiration). Sensible heat flux was the second largest use and the smallest use (50–70 Wm^{-2}) was invested in storage heat flux. At the suburban station (S1) the heat fluxes were much more balanced with much higher storage heat flux compared to the rural sites. Finally, during daytime the urban sites were dominated by a higher sensible heat flux and a pronounced storage heat flux with up to 184 Wm^{-2} at U1, which is nearly 4 times higher than at the rural sites and could reach up to 50 per cent of the net radiation (Oke *et al*. 1999; Rigo & Parlow 2007). Much of this energy is stored very efficiently in the urban fabrics over the day.

During night-time (Fig. 1.2.2b) rural and urban sites behave completely different. In all cases mean nocturnal net radiation is negative and is compensated for by the positive heat fluxes. At both urban sites (U1, U2) net radiation exceeds – 60 Wm^{-2}, which indicates that even during night-time the surface temperatures of urban surfaces are higher compared to the rural ones. At the rural sites negative net radiation is compensated for by all heat fluxes: sensible heat flux leads to a decrease in air temperature, latent heat flux controls condensation, and storage heat flux cools the soil temperatures. At the suburban and the urban sites the net radiation compensation is carried out predominantly by the storage heat flux. At the urban sites (U1, U2) the absolute value of storage heat flux is higher than the net radiation, indicating an over-compensation of net radiation which enables a small sensible and even latent heat flux to be kept in the boundary layer atmosphere. The source of energy for this compensation is the heat stored during daytime in the urban construction material, which acts like a battery, charged during daytime and discharged during night-time. Consequently the decrease in air temperature in the urban system during night-time is much smaller compared to that in rural conditions. These different characteristics of rural and urban energy budget are very important in explaining and understanding the urban heat island effect.

1.2.3 The urban heat island phenomenon

One very prominent feature of urban climate is the urban heat island. Daily or annual mean urban air temperatures are generally some degrees higher than in the rural surroundings. It is a characteristic feature of the urban climate and can be found in most cities of the world. It is not limited to summer conditions only. But if air temperature differences between urban and rural sites are analysed over short time periods (e.g. minutes or hours) significant differences between day and night-time conditions can be found. The heat island intensity, the magnitude of the temperature difference (ΔT), correlates positively with the size of the town, however

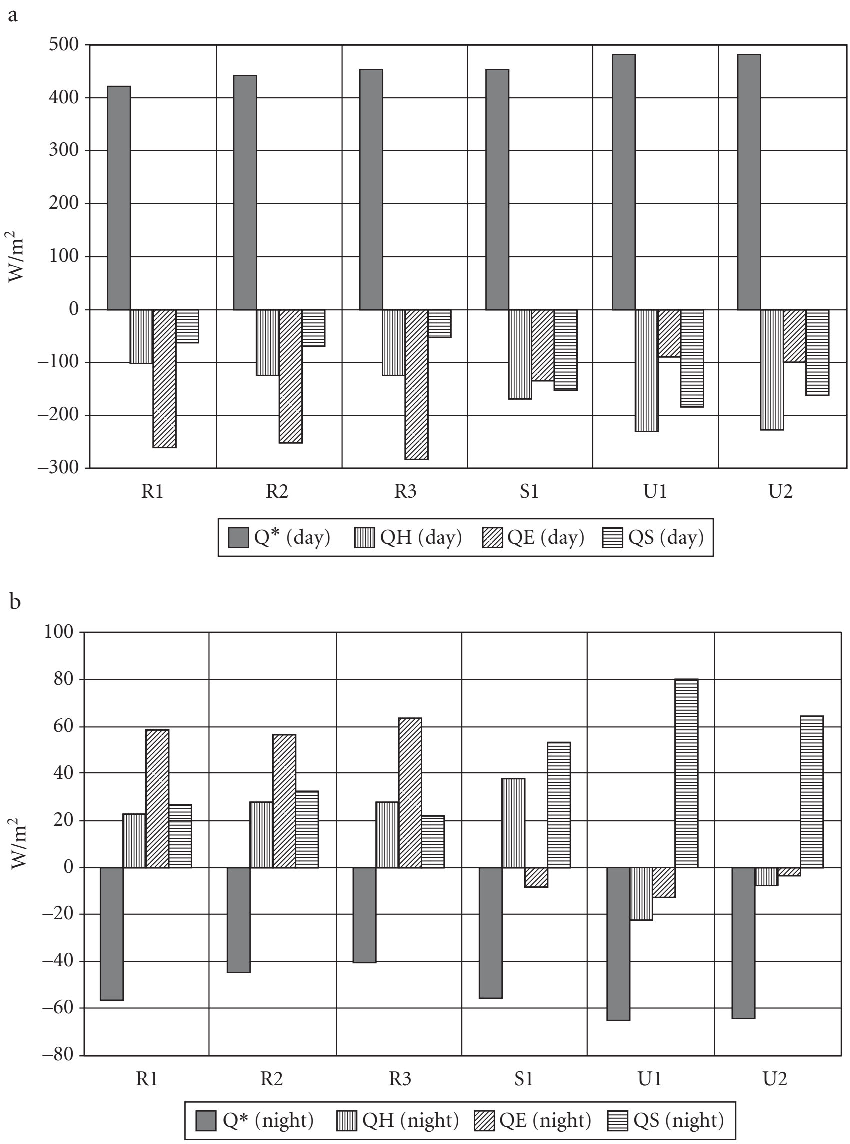

Figure 1.2.2 Mean daytime (a) and night-time (b) rural (R1, R2, and R3), suburban (S1), and urban (U1 and U2) radiation and heat fluxes during BUBBLE with Q*: net radiation, Q_H: sensible heat flux, Q_E: latent heat flux, and Q_S: storage heat flux

there are differences between European, North-American, and Asian cities (Oke 1987). These are strongly dependent on the different construction types of buildings and especially the use of air conditioning in private and public houses.

The observation of urban heat island goes back to the early nineteenth century. In 1820 Luke Howard (1772–1864) published the first urban climatology, *The Climate of London*, in which he analysed his measurements of temperature differences between the City of London and the rural surroundings over many years. He published the second edition in three volumes in 1833 (Howard 1833). Howard discovered that, during night-time, air temperatures were higher in London compared to the rural sites and that during daytime it was just the opposite. So Luke Howard was a pioneer of urban climatology.

Figure 1.2.3 gives an overview of the annual and diurnal dynamics of the urban heat island effect, measured at an urban station in the city of Basel/Switzerland and the rural station of Fischingen, which is about 5 km away, for the period from 1994–2002. The temperature differences between the urban and the rural station are shown. On the top panel the annual dynamic of the mean daily temperature differences (ΔT) are displayed. In all months of the year, mean daily temperatures of the urban site are 0.2–0.5 K higher than the rural one, indicating that the urban heat island is not dependent on seasonal effects. The right panel shows the diurnal dynamic over 24 hours as mean annual values. Here it can be clearly seen that during night-time the urban air temperatures are up to 1.4 K higher than the rural ones and during daytime the urban temperatures are up to 0.5 K lower than the rural. This phenomenon is exactly that which Luke Howard published in 1820. The urban heat island effect has a distinct diurnal characteristic. During daytime it is mostly absent, although ther-

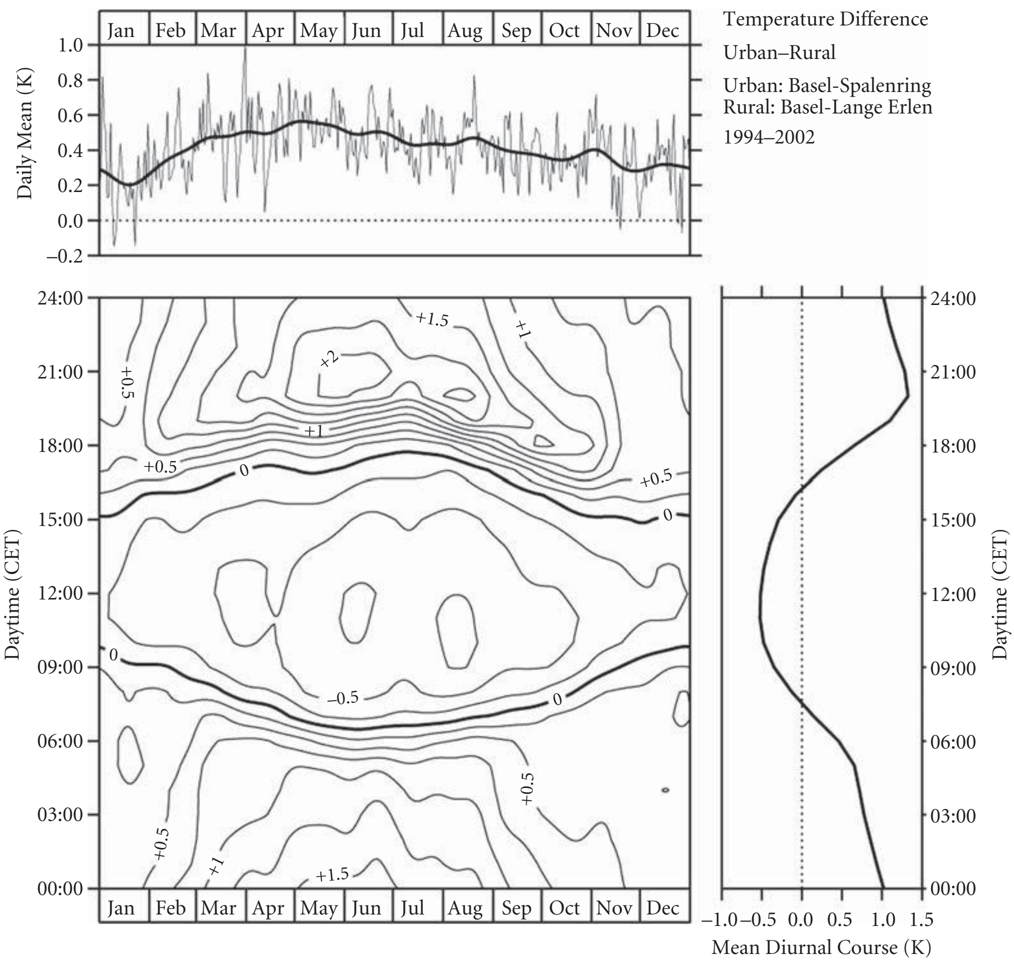

Figure 1.2.3 Thermo-isopleths of air temperature differences between an urban and a rural station in Basel/Switzerland (mean values from 1994–2002). The large panel shows the day of the year on the X-axis and time of day on the Y-axis. Contour lines indicate positive or negative temperature differences between the urban and nearby rural site with daily and annual variations

mal satellite imagery shows high surface temperatures (see Fig. 1.2.1). At first glance this is a contradiction, but surface temperatures are more closely related to the storage heat flux (which is high in towns), and air temperature is the consequence of sensible heat flux and turbulence in the urban boundary layer. The isopleths diagram (large panel) shows both the annual and diurnal variations of the temperature difference. During the summer months the heat island effect is strongest during early evening hours and exceeds +2 K. The equilibrium line ($\Delta T = 0$ K) in the morning and evening follows the annual variations of day length, giving a shorter but stronger nocturnal heat island period in summer. If data are not averaged over nine years (as in Fig. 1.2.3) but are taken from the daily measurements, the temperature difference (ΔT) can exceed 8 K or more, even during the warmest period of the European heat wave in 2003 (Parlow 2006).

The fact that sensible heat flux of an urban surface is mainly not changing direction during the day has been already mentioned in connection with Fig. 1.2.2. During the day and night the sensible heat flux is negative, which indicates that the flux remains directed upward, from the surface towards the atmosphere. In Fig. 1.2.4 the diurnal and annual variations of the sensible heat flux are shown as isopleths, averaged over a period of 9 years to smooth single weather events. During the daytime, sensible heat flux reaches up to –180 Wm^{-2} in the summer, but even during night-time a small upward flux of –10 – –20 Wm^{-2} remains over the whole year despite the negative nocturnal net radiation (Christen & Vogt 2004). This means that during the night the negative net radiation is completely compensated for by the storage heat flux and not by the sensible heat flux. Therefore, urban night-time air temperatures remain at a significantly higher level compared to rural ones: this is known as the urban heat island effect.

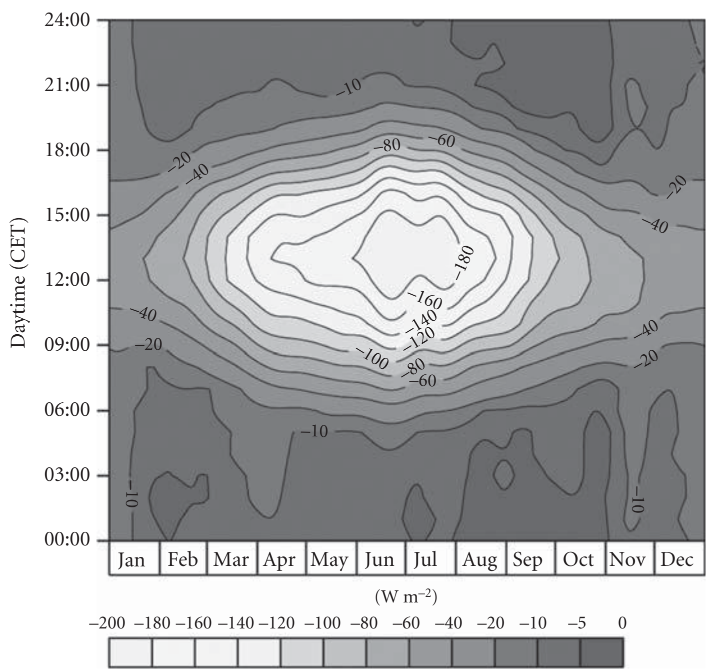

Figure 1.2.4 Isopleths of annual (X-axis) and diurnal course (Y-axis) of sensible heat flux at the urban station (Basel-Spalenring), mean 10-minute values averaged over 1994–2002. The negative sign indicates the upward direction of the sensible heat flux during night and night all over the year

1.2.4 Biological aspects of urban climate

Since the urban climate differs from the rural surroundings it is very probable that vegetation in cities is directly influenced by these differences. Air temperature, particularly, is a prime candidate directly related to plant phenology. Phenology is a highly sensitive indicator of any modification in air temperature on a year-to-year basis. But temporal shifts in phenophases (beginning of pre-spring or full-spring flowering) are influenced by both the general global climate change and the local urban climate effect. It is therefore necessary to analyse the start dates of phenophases in urban and nearby rural situations. Many studies show a shift in start dates of phenophases and growth lengths for urban agglomerations around the world using data from phenological gardens, measurements, or remotely sensed data from satellites or aircrafts (Lu *et al*. 2006; Menzel & Fabian 1999; Neil & Wu 2006; Luo *et al*. 2007). Rötzer *et al*. (2000) report from ten central European cities that, in nearly all cases studied, flowering in urbanized areas started earlier than in corresponding rural areas. They studied the flowering dates of snowdrop (*Galanthus nivalis*), forsythia (*Forsythia* sp.), sweet cherry (*Prunus avium*), and apple (*Malus domestica*). During the period 1951 to 1995 the mean difference in urban areas versus rural locations was about 4 days for the pre-spring and 2 days for the full-spring phenophases with a general shift in both areas towards an earlier flowering by up to 15 days per decade as a consequence of general global warming since the late 1970s and early 1980s. Rötzer *et al*. (2000) demonstrated that this earlier start of flowering in urban areas has been seen in many urban areas for decades and the phenomena is not limited to recent years. Data from Berlin/Germany go back to the decade 1965–74 and show a shift in urban areas of up to 5 days, Vienna/Austria can be traced back until 1961–1970 and also show a shift of 3 to 6 days compared to nearby rural conditions.

Having already discussed the urban heat island effect and its physical basis, the response of urban vegetation to higher air temperatures is not surprising. Especially during night-time, the increase of urban air temperature is significant and minimizes frost risk in spring.

The general influence of global warming is demonstrated in Fig. 1.2.5, which shows the decrease in the number of days with frost (minimum temperature < 0 °C) for the city of Basel/Switzerland for the period from 1901–2007. The mean trend over this century is a decrease of 1.3 days with frost per decade, resulting in a reduction of 25 per cent of days with frost over the period 1901–2007. It can be pointed out that the special situation of urban climate with pronounced local urban heat island effect is amplifying the general trend of global warming and prolongs the vegetation period in urban areas significantly.

1.2.5 Chemical aspects of urban climate

Urban areas are the centres for population and industry and are therefore an important source for emissions and pollutants. The urban atmosphere has a mostly unique mixture of specific concentrations of gasses compared to a non-urban atmosphere. Urban areas contribute significantly to global carbon dioxide and nitrogen oxide emissions. Vehicle emissions and other anthropogenic and industrial activities are a source of important trace gases and pollutants. During recent years the urban aerosol has become an important political issue. Particulate matter (PM) describes the mass of all aerosol particles with an aerodynamic diameter of less than 10 µm (PM10, thoracic fraction) or less than 2.5 µm (PM 2.5, respirable fraction). Even smaller particles (ultrafine particles) are important, having a diameter of less than 0.1 µm. Sources of PM can be natural (volcanoes, dust storms, forest fires etc.) or anthropogenic (burning of fossil fuel, burning of waste, power plants, industrial processes, tire abrasion etc.). Since these particles can penetrate into the lungs, and the ultrafine particles into the blood system or even the lymphatic system, they can affect other organs and are related to health hazards such as heart disease, disorders of the respiratory system, and lung cancer.

Many of the pollutants tend to have increased concentrations in the urban atmosphere. Table 1.2.3 gives a short overview of mean values of air pollutants for Germany (Helbig *et al*. 1999).

EU legislation has set limits for many air pollutants. From 1 January 2010 the yearly average of

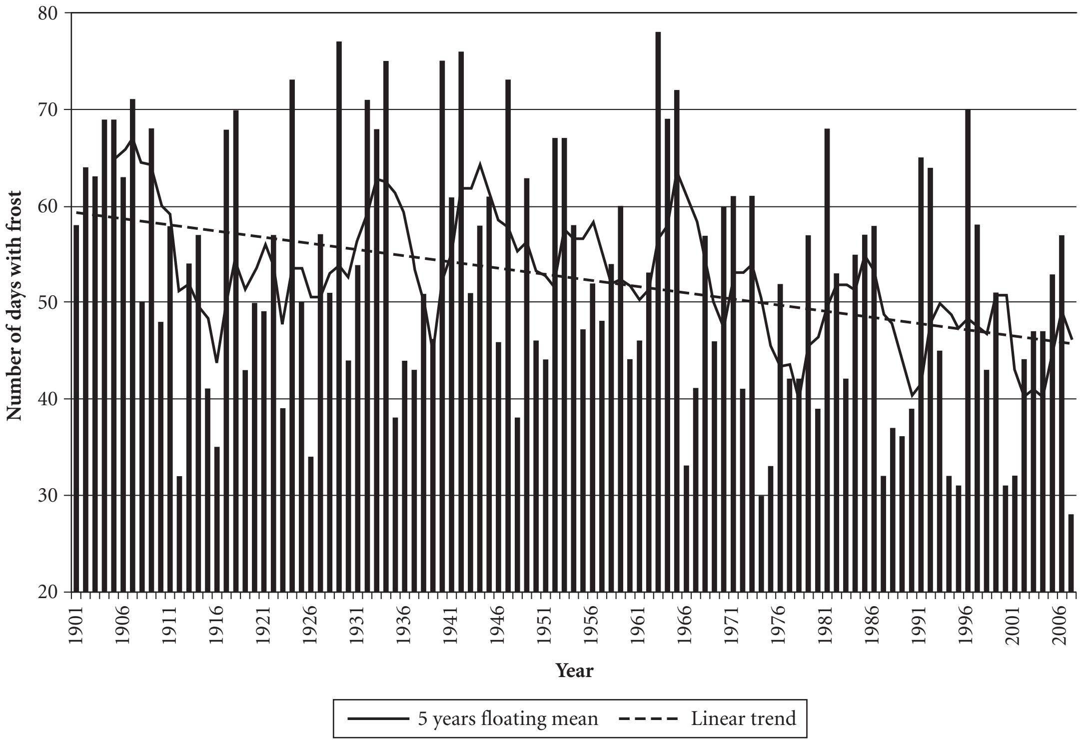

Figure 1.2.5 Number of days with frost in Basel/Switzerland during 1901–2007

Table 1.2.3 Mean concentrations of air pollutants in rural and urban areas for Germany

Pollutant	Rural areas	Urban areas	Maxima (urban)
CO	< 0.5 mg m^{-3}	0.5–5 mg m^{-3}	30 mg m^{-3}
SO_2	5–15 µg m^{-3}	10–75 µg m^{-3}	1600 µg m^{-3}
Total suspended matter (TSP)	25–30 µg m^{-3}	75 µg m^{-3}	1200 µg m^{-3}
NO	5–15 µg m^{-3}	40–120 µg m^{-3}	1200 µg m^{-3}
NO_2	10 µg m^{-3}	40–80 µg m^{-3}	450 µg m^{-3}
CH	20 ppb C	200–600 ppb	up to 30 000 ppb C
O_3	50–75 µg m^{-3}	25–50 µg m^{-3}	300 µg m^{-3}

PM10 is limited to 20 µg m^{-3}. Due to heavy anthropogenic emissions, these strict directives are currently mostly unachievable in urban areas. For nitrogen oxide (NO_2) the yearly threshold is 40 µg m^{-3}, and for carbon monoxide (CO) the maximum daily threshold averaged over eight hours is limited to 10 mg m^{-3}.

The mean carbon dioxide (CO_2) concentrations in urban and rural areas are very much influenced by the global background concentration and the interaction between vegetation and atmosphere through photosynthesis and the respiration process. Due to the long lifetime of carbon dioxide, the global concentrations are well mixed by atmospheric circulation and the global background value of 385 ppm for 2008 is still growing by roughly 2 ppm per year (data from the Mauna Loa Observatory on Hawaii). This corresponds with data from the German Environmental

Protection Agency (Umweltbundesamt) for the station Schauinsland in the Black Forest of 387 ppm. This is the longest record for CO_2-measurements in Europe. Most of the monitoring stations for global carbon dioxide concentrations are located in non-urban areas and it is imperative that we improve our knowledge on carbon dioxide concentrations, fluxes, and dynamics in the urban environment.

Quantifications of CO_2-emissions are mainly estimates of fossil fuel consumption. Little is known about measured concentrations in urban areas. The daily variations are a result of anthropogenic, biogenic, and meteorological influences. Vogt *et al.* (2006) measured CO_2 fluxes and profiles in several levels in the urban canopy layer of a European city during summer 2002. The mean diurnal course can be classified into four stages: (1) low concentrations in the afternoon (375 ppm); (2) rising concentrations during late evening and during night (390 ppm); (3) maximum values in the early morning between 5:00 and 7:00 hours (425 ppm); and (4) rapid reduction during the morning until noon (410 ppm). The diurnal amplitude was in the range of 50 to 100 ppm and was not correlated to urban traffic load. It was rather dominated by the atmospheric stability, with shallow inversions during the night and reduced vertical mixing which led to maximum concentrations. On the other hand, the carbon dioxide fluxes measured in a street canyon were higher during daytime, with a mean maximum of 20 µmol m^{-2} s^{-1} at the late afternoon rush hour. A similar diurnal pattern and level of CO_2 concentration is also reported from a suburban locality in Tokyo/Japan, with concentrations between 406 and 444 ppm during a two-month period in winter 2004 (Moriwaki *et al*. 2006).

Another important pollutant which is greatly increased in urban areas is nitrogen dioxide (NO_2), which is mainly traffic-related. Emissions are generally highest in urban rather than rural areas. Annual mean concentrations of nitrogen dioxide in urban areas are generally in the range of 10–45 ppb, and lower in rural areas. Levels vary significantly throughout the day, with peaks generally occurring twice daily as a consequence of rush hour traffic. Concentrations can be as high as 200 ppb. The problem is not necessarily concentrated in the inner cities, because many major roads and motorways are situated in semi-rural areas. Today, concentrations of key air quality components and trace gases such as NO_2, SO_2 (sulphur dioxide), BrO (brome oxide), O_3 (ozone), OClO (chlorine dioxide), and aerosol characteristics can be measured from Earth-orbiting satellites like the NASA Aura-OMI (Ozone monitoring instrument) in high temporal and spatial resolutions. Figure 1.2.6 shows the monthly total tropospheric NO_2-mass over East Asia, in 10^{15} molecules per cm^2 air column, for January 2007, measured by NASA-OMI. The highest concentrations are clearly evident around Hong Kong, Shanghai, Beijing, Seoul, Osaka, and Tokyo with more than 18×10^{15} molecules NO_2 cm^{-2}, contrasting with less than 2×10^{15} molecules NO_2 cm^{-2} in the western part of China, in Mongolia, or over the Pacific Ocean.

These very high concentrations are related to traffic and industry emissions in urbanized areas in combination with the stable stratification of a winter atmosphere. These few examples all clearly show that the urban atmosphere has a significantly higher level of carbon dioxide and also nitrogen dioxide concentrations compared to mean global conditions, due to the increased local emissions by traffic, industry, and heating.

1.2.6 Impacts of urban climate on human health

There are two important issues of urban climate which influence the health of people: heat stress during summer conditions and air pollution. Due to the urban heat island effect, heat stress for human beings is enhanced and air pollutants also show higher concentrations both in summer and in winter. Since the summer of 2003 (which was probably the hottest in Europe since at latest AD 1500 and resulted in unusually large numbers of heat-related deaths reported in France, Germany, and Italy) this has become a particularly important topic in science and policy (Stott *et al*. 2004).

Bio-climate is an important factor for human well-being, and integrates all meteorological variables such as air temperature, air humidity, wind speed, shortwave solar radiation, and longwave terrestrial emission. All these variables influence the human heat budget. Humans have to keep a body temperature of 37 °C under all climatic conditions, and this

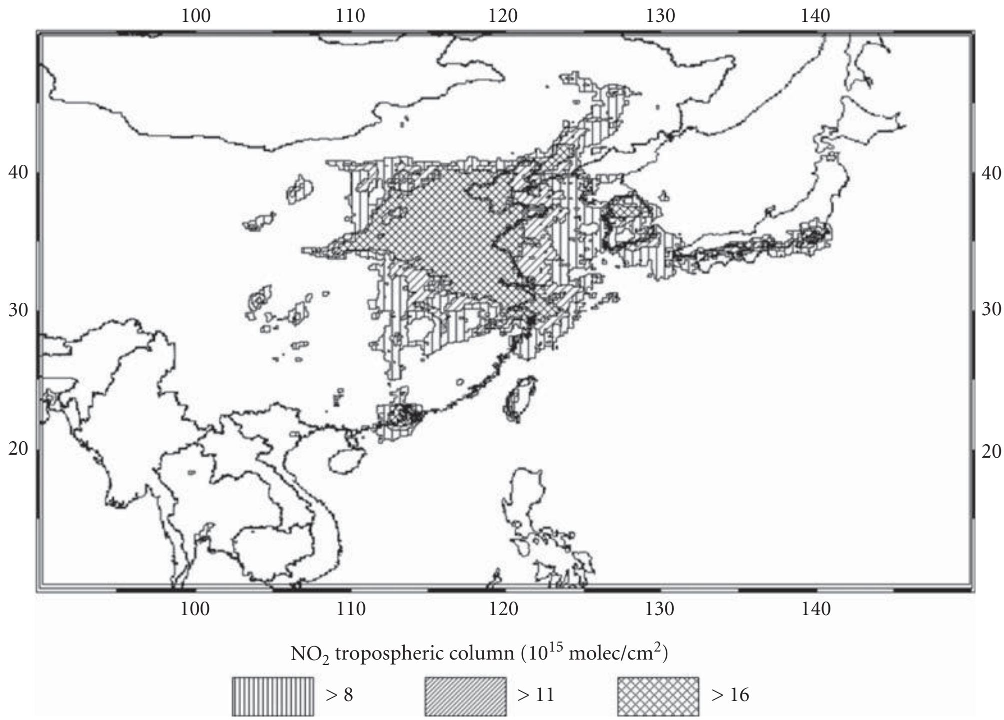

Figure 1.2.6 Total tropospheric nitrogen dioxide column over East Asia for January 2007, measured from the NASA-OMI sensor

value only allows very small deviations. Therefore thermal stress can lead to adverse effects for human health. This issue has already been addressed in numerous epidemiological studies, in which the thermal environment has been related to thermal related mortality (Basu & Samet 2002; Curriero *et al.* 2002; Koppe & Jendritzky 2005). It is very probable that, especially in the urban environment, the frequency and intensity of summer heat waves will increase due to global warming, and that heat waves such as that experienced in Europe in 2003 will not remain an exception (Meehl & Tebaldi 2004). Since heat stress is a combination of several meteorological variables it cannot be measured directly but has to be computed or modelled. Therefore, the human heat budget has to be formulated and the heat budget equations for humans can be written as follows (VDI 1998):

$$M + W + Q^* + Q_H + Q_E + Q_{Sw} + Q_{Re} = 0$$

With: M: total energy conversion (metabolic rate),

W: human activity,

Q^*: net radiation,

Q_H: sensible heat flux density,

Q_E: latent heat flux density by diffusion of water vapour through pores,

Q_{Sw}: latent heat flux density by sweat transpiration, and

Q_{Re}: heat transport by respiration.

Through food digestion and activity rate, people always produce energy, which has to be balanced to keep the body temperature constant. The possibility of releasing heat is strongly dependent on environmental conditions. The gradient between air temperature and skin temperature regulates the loss or gain of energy. In cool air we lose energy through sensible heat flux and vice versa. A very effective way for humans to release energy is through the latent heat flux by transpiration, sweating, or respiration. By breathing deeply we nearly saturate the air with humidity (relative air humidity ≈ 100% at 37 °C body core temperature corresponds to a vapour pressure of 62 hPa or a mixing ratio of 43 g H_2O kg^{-1}, which normally is not possible under natural conditions). This humidity has to be transpirated within the respiratory system, which leads to a significant loss of energy from the human body. Even under tropical conditions this process of latent heat release is possible, but as air humidity increases, this heat flux becomes less effective and people increasingly suffer from heat stress. Any wind speed will help to keep the sensible and latent heat fluxes working. Exposure to solar radiation, as on the sunny side of the street, implies an additional energetic input to the human body, which makes walking in the shade of a street more comfortable even though air temperature does not differ substantially between the sunny and the shady side.

Humans absorb nearly all longwave radiation (black body radiators) and, according to the law of Stefan-Boltzmann, surfaces with high surface temperatures emit longwave radiative energy, which is also a significant heat gain for the body.

According to these physical principles of the human heat budget, heat stress is related to summer fair weather conditions without high wind speed, high air temperatures, high air humidity, and being surrounded by warm surfaces of walls and asphalt. In the urban environment these conditions are met much more frequently compared to rural areas, and, due to the urban heat island effect, nights in cities are much warmer than in non-urban situations. During hot summer nights the air temperature does not necessarily drop below 20 °C, which is the temperature threshold for a ‘tropical night’. Under these conditions the thermophysiological recreation phase during sleep is not sufficient for most people. In many cases medical care and treatment is requisite and, in extreme situations like during the European summer heat wave of 2003, a significant increase in mortality rate occurs (Schär & Jendritzky 2004).

The second problem of urban climate for human health is air pollution. Air pollution in cities is the result of complex interactions between natural and anthropogenic environmental conditions. Poor air quality in cities is a serious environmental problem and a seriously growing one in developing countries. Emissions from motor traffic are a very important source of air pollution throughout the world. During transmission, air pollutants are dispersed, diluted, and subjected to photochemical reactions. But in many cases there are additional sources of air pollutants such as burning fossil fuel, burning waste, industrial processes, tire abrasion, and many more. This is an increasing problem in most of the world’s mega-cities that have more than 10 million inhabitants. The booming urban population is congregating in the mega-cities of today, particularly in the developing countries of Asia, South America, and, increasingly, Africa. Most cities worldwide suffer from serious air-quality problems, which has received increasing attention over the past decade.

Urban population growth is caused by migration into cities and a surplus of births in the cities themselves, particularly the high birth rates in the developing countries. A deep structural change is mainly responsible for the migration to cities, especially in non-industrialized countries. This structural change is the consequence of economic opening-up, new trading partners, and a change of political conditions. Structural change takes a rapid course in some countries. It is not surprising that the expected urban population growth from 1992 until 2010 is much higher for Lagos, Bombay, or Dhaka than for Tokyo or New York, which were the first mega-cities of the world.

Urban population growth has many consequences. One of them is higher emission of air pollutants. Even though for most air pollutants, the emission rate per inhabitant is at present higher in industrialized countries, it is obvious that this rate will in future be higher in developing countries. Most of the present mega-cities are in developing countries, where urban air-pollution problems are the most severe. These include Bangkok, Beijing, Bombay, Calcutta, Delhi, Jakarta, Karachi, Manila, Seoul, Shanghai, Buenos Aires, Cairo, Mexico City, Moscow, Rio de Janeiro, and Sao Paulo.

Time series of air pollutants indicate trends in air quality. In mega-cities of non-industrialized countries, the time series of air pollutants are often too short for statistical trend analysis. In many cities of industrialized countries, however, air pollutant time series are sufficiently long to permit trend analysis. WHO and UNEP created an air pollution monitoring network as part of the Global Environment Monitoring System for time series analysis of air pollution. A study of air pollution in 20 of the 24 mega-cities of the world shows that ambient air pollution concentrations are at levels where serious health effects are reported. The study shows that each of the 20 mega-cities has at least one major air pollutant which occurs at levels that exceed WHO health protection guidelines (Baldasano *et al*. 2003). Beijing, Cairo, Jakarta, Los Angeles, Mexico City, Moscow, and Sao Paulo are facing a variety of air pollution problems requiring comprehensive solutions. The study shows that the ambient air quality in the majority of the mega-cities is getting worse as population, traffic, industrialization, and energy use are increasing. In terms of the degree of severity, the high levels of particulate matter (PM) are the major problem affecting the mega-cities. Particulate matter presents a very serious problem in 12 of the mega-cities surveyed, the majority of which are located in the Pacific Basin. The concentrations of PM in these cities are persistently above the WHO guidelines by a factor of as much as two or three. From a review of trends in air quality in different cities made by Mage *et al*. (1996), it is quite evident that the experience of the current mega-cities in the developed countries is being repeated in the developing countries.

One of the most polluted cities of the world is Cairo (Mosalam Shaltout *et al*. 2001; Zakay *et al*. 2008; Mahmoud *et al*. 2008). Measurements of particulate matter over a four-month period at several stations in Cairo showed dramatic results. A passive sampler, collecting particulate matter on an adhesive plate over several days, was exposed in the city centre of Cairo for four days and, as a comparison, at a traffic site in Berlin for seven days. Figure 1.2.7 shows how dramatically the air in Cairo is contaminated with black particles compared to a heavily frequented traffic road in Berlin/Germany. The scale bar in the bottom left corner corresponds to a size of 40 μm. This high degree of air pollution in Cairo is related to enormous volumes of traffic and waste burning all around the city. Particles can be analysed in relation to size, transparency, and shape of depositions.

Air pollution in mega-cities is a challenging problem of the future. Since the urban population is growing dramatically, especially in developing countries, this is an issue of global importance and many more studies must be carried out to gain more insight into the impacts and feedback of air pollutants to human health.

1.2.7 Conclusions

Worldwide, the urban climate has become an important issue and it will remain on the agenda of politics and research for a long time. Today, more than half of the world's population is living in an urban

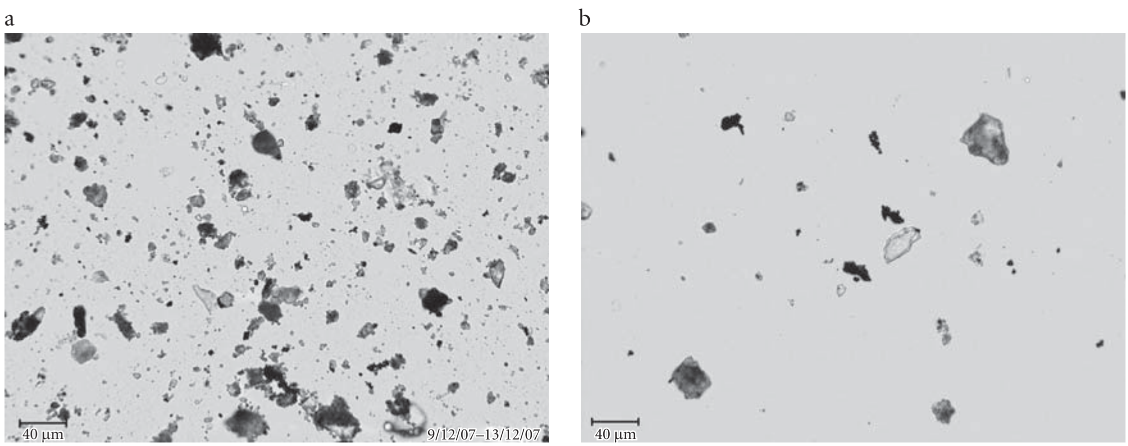

Figure 1.2.7 Passive sampling of particulate matter (PM10) in Cairo (city centre, left, exposed over four days), and Berlin (traffic site, right, exposed over seven days)

environment which modifies the local climatic conditions compared to rural surroundings. The urban heat island effect, which is a characteristic feature of the urban climate, has to be discussed by analysing the urban heat budget in detail. It is a normal feature for night-time conditions but during daytime city air temperatures are usually cooler than rural ones. The accelerating growth of mega-cities, especially in third world countries, is a challenge for urban climate research and applications since these cities are major contributors to air pollution and the emission of greenhouse gases. Urban agglomerations already modify the present global climate conditions, and this can be demonstrated with numerous examples. The global warming expected over the next decades will again significantly influence the climate of cities with respect to air temperatures, heat stress, and so on.

CHAPTER 1.3

Urban Soils—Characterization, Pollution, and Relevance in Urban Ecosystems

Martin Sauerwein

1.3.1 Introduction—what are urban soils?

Anthropogenic disturbances of any kind will have an effect on the geo-ecosystem, including the soil cover. The outcome depends on the type, intensity, and the point in time or duration of the human activity. Anthropogenic interferences in (former) natural landscapes are difficult to evaluate. The complexity of these activities normally precludes a simple explanation (compare with Haber 1999) of the impact on the geo-ecosystem.

In Europe, for example, the natural landscape has been almost completely converted into a cultural landscape. However, this transformation has not taken place equally in all areas—the duration and intensity of the process varies considerably. Consequently, many landscapes are in a transient state—they are still in the process of responding to anthropogenic interference (Bork *et al*. 1998).

Human activities and practices have produced new geo-ecosystems such as the agricultural, managed forest, and urban ecosystems. One characteristic feature of the latter is that the environmental conditions of the urban ecosystem, described by the energy, material, and water balance, are completely different from other geo-ecosystems (Bullock & Gregory 1991; Burghardt 1994). Typically, the urban landscape is distinguished by the particular qualities of the urban hydrosphere, atmosphere, and biosphere as well as by the specific modifications of the pedosphere as a result of land-use (Pietsch & Kamieth 1991). One of the most noteworthy activities in this context is surface sealing. This complex ecological parameter can provide important information about intra-system material flows and balances (Brown 1999).

The ecological importance of urban soils was underestimated for a long time. Soils were seen as nothing more than a support for buildings (Craul 1992). The first urban soil surveys made in the 1970s (Runge 1975) initiated methodical discussions on surveying procedures and urban soil classification which led to new insights and innovative approaches (Burghardt 2002; Schraps *et al*. 2000). To date, however, there is still no concept that adequately integrates urban soils into the urban ecosystem (Ad-hoc-AG Boden 2005).

Consequently, many soil science textbooks usually characterize the urban pedosphere only by virtue of its extreme lateral and vertical heterogeneity (Craul 1999; Blume *et al*. 2009). However, there is an underlying and recognizable pattern to this intense variability, which is not the result of anthropogenic 'chaos' (Sauerwein 2006). These patterns are apparent in the soil profile and the contaminant contents and form the ecological basis for numerous other ecosystem functions.

Urban soil research is still a relatively young scientific discipline. The first Central European urban soil surveys date back to 1975 in Berlin (Blume 1982) and 1979 in Halle (Billwitz & Breuste 1980). There is still no internationally accepted survey concept. This has led to a situation where numerous approaches co-exist for classifying and typifying soils in urban areas. The international standard

Table 1.3.1 Characterization of urban soils

Source	Designation	Characterization
Blume 1998	Soils of the urban-industrial agglomerations	Distinguishes three groups: • modified soils on natural deposits • soils on anthropogenic deposits of natural substrates or mixtures thereof • sealed soils.
Burghardt 2002, 1994	Urban soils	• Categorization according to the type of deposit, pedogenetic processes, and soil characteristics. • Substrates differentiated according to modified properties due to substrate treatment. • Many soils are in the initial stages of development. • The soil environment has been altered (many soils are relics). • Soils composed of moved horizons retain properties that have not developed in situ (phenotype).
Craul 1999	Urban soils	• Primary differentiation into natural and 'urban made'. • Secondary differentiation according to site conditions and soil conditions (fill, cut, mound, or berm).
FAO Soil Map of the World 1985 (www.fao.org)	Anthrosols	• Soils that have been formed or profoundly modified through human activities. • Four second level classes: aric, fimic, cumulic, and urbic anthrosol.
Fiedler 2001	Urban soils	• Division of anthropogenic soils, classified into terrestrial cultosols, moor-cultosols, mining soils, deposited soils (deposols), sealed soils, irrigation soils, and reductosols.
Kuntze, Roeschmann, & Schwerdtfeger 1994	Division 'Anthropogenic Soils' (= cultosols)	• Divided into three classes in addition to non-classifiable soils: 'anthromorphic'soils (deposols, truncated soils/denusols, infiltrated soils/intrusols).
Pietsch & Kamieth 1991	Urban soils	• Soils as a component of urban-industrial ecosystems. • Land-use type as reference sites.
Sauerwein 2006	Urban soils	• Characterization of urban soil landscapes.
Schwerdtfeger 1997, Schwerdtfeger & Urban 1997	Division 'Anthropogenic soils'	• 7 classes. • Main distinguishing principle: objectives of human activities.

literature, as well as the main attributes, nomenclature, and characterization of urban soils, are summarized in Table 1.3.1. Siem (2002), Meuser, and Blume (2001), as well as Bullock *et al*. (1999), Effland and Pouyat (1997), Tan (1994), and Pierzynski *et al*. (1994) all point out that the international classification is very unsatisfactory.

A definition of 'urban soils' (synonyms: settlement soils, urban-industrial soils, soils of urban and industrial agglomerations) could be formulated as follows:

> 'All soils occurring in urban areas. These soils are formed on natural as well as anthropogenic deposits of natural and technical substrates and are often characterized by a complex small-scale distribution pattern. Urban soil associations are subject to human interventions (such as sealing) and intense use which modify soil properties and contaminant concentrations.' (Sauerwein 2006)

Most classification approaches listed in Table 1.3.1 do not take into account the function and importance of the soils as an integral component of the urban geo-ecosystem, or do so only inadequately. This compilation illustrates the variety of currently available schemes for categorizing soils in the urban landscape. However, from the ecosystem theoretical point of view, these approaches fail to provide a reproducible ecosystem-based and spatially relevant classification system.

Urban soils are an integral component of the urban-industrial ecosystem and an important ecological asset. Anthropogenic activities involving the urban terrain deliberately or arbitrarily modify the functionalities of the unsealed soil. Consequently, the specific pedogenic processes in the urban environment lead to the development of distinct soils. Thus urban soils are at least in part the result of the incriminating influences of human activities.

Unsealed urban soils are principally subjected to two categories of incriminating influences:

(1) Soils are substance sinks (Fellenberg 1994): most substances emitted into the atmosphere and hydrosphere, as well as those from the circular flow, are deposited in the geosphere, especially on terrestrial soils. Industrial and trade activities draw large quantities of substances into the urban area where they are transformed and re-distributed (emitted). These short-distance emmissions accumulate on surfaces with only a restricted ecological capacity for containing substance deposits. Management practices and accidents can also introduce use-specific (contaminating) substances to the soil.

(2) Soil structure is determined by the spatial arrangement of the solid particles and has an important effect on soil properties and pedogenesis. The mechanical impact on soil structure in urban-industrial areas is restricted both in time and spatially.

These effects can change the properties of the soil (Christopherson 2008). In some cases the soil functions only as a 'carrier' for contaminants with detrimental effects on their function and potential uses. The evaluation of soil and groundwater contamination is based on the results of sample analyses.

The substances and their chemical transformations, as well as mechanical impacts, affect the following properties (Rowell 1994): soil chemistry (pH-value, substance mobility), soil physics (new substrates, water budget, and aeration), soil biology (modification of the living conditions for flora and fauna), and site ecology (for example soils as the basis for biotopes).

In summary, urban soils can be described as former natural or artificial soils with a specifically urban pattern of contamination, whose natural properties have been strongly modified by a wide range of human activities of varying intensities.

1.3.2 Pollution of urban soils

All substances or compounds that enter the soil are potential contaminants. The importance of a contaminant is determined by the dose-response relationship (Biasoli *et al*. 2007). Soils can be affected by natural substances (e.g. chloride, nitrate) as well as over 100,000 artificial substances (Ellis & Mellor 1995; Thuy *et al*. 2000). The quantity of individual substances released into the environment varies between several 100,000 t/a to a few kg/a. Table 1.3.2 presents an overview of important general and specifically urban soil contaminants.

In addition to these, there are numerous other substances whose toxicological effects and distribution

Table 1.3.2 Important soil contaminants

Common and/or especially harmful confirmed potentially hazardous substances	Confirmed potentially hazardous substances, but only locally important
Arsenic, cadmium, lead, zinc, nickel, aluminium, copper	Chrome, thallium, beryllium, cobalt, uranium
Nitric acid/nitrates	Hydrofluoric acid/fluorides, cyanides
Sulphuric acid/sulphates	Ammonium
Hydrochloric acid/chloride	Mineral oils
PCB/PCT/PCN (polychlorinated biphenyls, polychlorinated terphenyl and Naphthalene), HCB, DDT, PCP, HCH, PAH, volatile chlorinated hydrocarbons (trichloroethylene, perchloroethylene), PCDD/PCDF	Aromatic nitro-compounds, aromatic hydrocarbons (especially benzene, toluene, naphthalene), Paraquat
Durable radionuclides	Phenols

in the environment are not yet fully understood, such as antimony, selenium, vanadium, borate, bromide, phthalates, octachlorostyrene, other organochlorines, constituents of washing and cleaning agents such as tensides, and phosphate replacements (Pietsch & Kamieth 1991; Blume 1998; Bädjer 2000).

In any consideration of substance accumulations it is very important to realize that the possibilities for removing contaminants from soils are extremely limited. The removal of salts, heavy metals, or accumulations of certain organic compounds requires an enormous technological or chemical effort (Blume *et al*. 2009). Some organic compounds are decomposed within a few days to years by biological processes. Substances (e.g. chlorides) may also be removed from the profile by leaching, leading to a displacement of the contaminants from the soil to the groundwater or surface waters (see Illgen, Chapter 1.4).

The group of persistent problematic substances, which resist degradation in the soil, are a growing potential threat (Wilcke *et al*. 1999; Sörme *et al*. 2001). These contaminants will continue to accumulate in the soil as long as they are being emitted. These accumulations can lead to latent and, once specific threshold limits have been exceeded, distinct adverse effects on soil flora and fauna (Schnoor 1996; Zhang *et al*. 2001). In addition to this they also pose an acute threat to human health either by direct contact or via the food chain and groundwater. There are five different exposure pathways for contaminants from the soil to the human receptor (see Fig. 1.3.1):

(1) Exposure pathway soil – air – humans (pulmonary/direct uptake).
(2) Exposure pathway soil – humans (oral/direct uptake).
(3) Exposure pathway soil – humans (dermal/indirect uptake).
(4) Exposure pathway soil – groundwater – potable water – humans (oral/indirect uptake).
(5) Exposure pathway soil – plants – food – humans (oral uptake via food chain).

Contaminant concentrations are generally much higher in urban areas as compared to rural areas. There are two fundamental sources of soil contamination (Sauerwein 2006; Pouyat *et al*. 2007):

(1) Diffuse, extensive, and relatively continuous low-level inputs. This includes mass contaminants as well as organic compounds and heavy metals. This type of input mainly affects the upper soil layers.
(2) Short-term, concentrated, spatially limited, local inputs (point sources). This contaminant source tends to be erratic, use-specific, and can affect all soil layers or horizons (Table 1.3.3).

Soil contaminant impact can be divided into qualitative effects (e.g. due to substance toxicity or persistence) and quantitative effects (e.g. due to acidification or leaching). These effects follow the various diffuse or concentrated, direct and

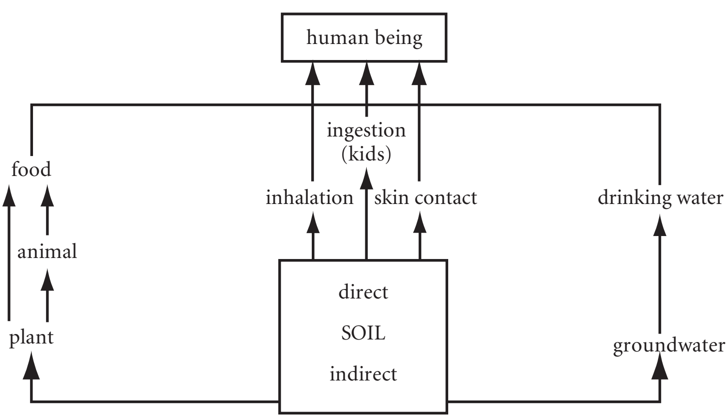

Figure 1.3.1 Soils as a source of contamination for humans (Sauerwein 2006, modified)

Table 1.3.3 Urban soil contaminant sources and input forms

Source	Duration	Input form	Concentration	Contamination
Immissions	very long	diffuse	very low	surface
Deposition	medium-term	local	high	surface/deep
Sewage system	very long	linear	high	deep
Defective tanks, conduits	medium-term	local	very low	deep
Accidents	short	local	very low	surface
Management measures	medium-term	local	low	surface

indirect input pathways (Yaron *et al*. 1996; Sauerwein 2002):

First, deposition of atmospheric pollutants from trade, industry, and house fire (diffuse, usually extensive input) aerosols may be introduced to the soil through two processes: wet and dry deposition. The most important deposited mass soil contaminants in Central Europe are: sulphur dioxide (approx. 2.5 million t/a, mostly from power stations), nitrogen oxides (approx. 3.1 million t/a, a large proportion emitted by traffic), and particulate matter containing inorganic and organic substances.

While emissions of sulphur dioxide and particulates have been clearly decreasing for years, the trend for nitrogen oxide emissions is slightly increasing. Trace pollutants, especially organic compounds, are also introduced to the soil via the atmosphere. Just like heavy metals, they are mostly bound to particulates and may also be washed out of the atmosphere by rain. Approximately 1.9 million tons of these organic compounds are released in to the environment in Germany every year. This group consists of thousands of substances, which are released in different quantities. Many of these compounds are broken down in the atmosphere, more persistent substances are introduced to the soil together with particulates or precipitation. This group of trace pollutants includes the environmentally important and comprehensive group of hydrocarbons, especially the halogenated hydrocarbons.

The second soil contamination pathway is traffic-related emmissions on road surfaces and railway facilities (diffuse as well as concentrated inputs on traffic surfaces and adjacent areas).

For instance, in 2007 traffic-related emissions in Germany included approx. 5.5 million tons carbon monoxide, approx. 0.6 million tons hydrocarbons, approx. 1.9 million tons nitric oxides, approx. 0.1 million tons sulphur dioxide, 68,000 tons of particulate matter and 1200 tons of lead. About 0.8 million tons of de-icing compounds were distributed on the roads in 2000. The environmental burden also includes unspecified amounts of herbicides used on the traffic facilities.

Third, hazardous substances from conduits and infrastructure such as sewage systems and gas pipelines (concentrated and linear inputs to deeper soil layers) are a pathway for soil contamination, but urban soil and groundwater contamination from leaking sewer lines has been underestimated in the past. Several million cubic metres of household and industrial (indirect dischargers) sewage seep into the soil beneath major cities every year.

Fourth, industrial deposits and contamination of industrial areas occurs on existing and abandoned industrial premises also known as brownfields, and embraces a wide range of industry-specific substances. This type of contamination may consist of a heterogeneous mixture of inorganic and organic substances depending on the production process. Concentrated inputs affect the whole soil profile to a depth of several metres and deposits include the disposal of production-related waste on tips, tanks, internal infrastructure etc.

Fifth, waste tips and 'wild' deposits (concentrated inputs) are a source of soil contamination. In addition to the contaminated ground of the brownfields, industrial areas often have operating waste tips and numerous small-scale deposits, landfills, and heaps. In some old industrial zones the area covered by these waste deposits may exceed that of the remaining natural soils.

Sixth, soil replacement with contaminated substrates (concentrated inputs) is a pathway for soil contamination.

Excavated soil, building rubble, and other (waste) substrates have been and are sometimes still used today for back-filling as well as constructing earth mounds during foundation work and safeguarding procedures for buildings, infrastructure constructions etc. These welcome (recycling) materials are quite often contaminated with hazardous substances (Pouyat *et al*. 2006).

Finally, sedimentation in standing and running waters (concentrated input) is a pathway for soil contamination.

The sediments of standing and running waters contain natural levels of contaminants, especially heavy metals. However, a much greater part of the contamination can be traced back to anthropogenic origins. Water draining an urban area concentrates diffuse contaminations to a relatively small contaminant sink. Studies, for example in the city of Halle, show that a large proportion of urban river and lake sediments are contaminated by heavy metals. These contaminants can be introduced to the soil, especially in the riparian zones, during flood events (Winde & Frühauf 2001).

In contrast to air and water pollution, soil contamination is not immediately apparent to the human senses. The effects of the accumulated substances are often only noticed when normal soil functions are seriously disrupted (Blume *et al*. 2009). The ecological potential under these circumstances is problematical, because natural 'decontamination', for example of heavy metal accumulations, is usually impeded by high pH-values and technological remediation options are both limited and very expensive.

The volatile nature of the 'soil problem' in urban ecosystems is not limited to the immediate effects on the (in situ) site properties. Soils are an integral part of the urban landscape. Therefore any reduction or even inhibition of the natural soil regulatory, control, and reservoir functions will decrease the ecological potential at the contamination source as well as have a (negative) effect on the substance and energy balance of the whole urban landscape (Leser 2008). The following examples illustrate the wide-ranging consequences of soil contamination: adverse effects on percolation and groundwater recharge potential (see Illgen, Chapter 1.4; Wessolek & Renger 1998), impairment of the micro-climatic effects (see Parlow, Chapter 1.2; Kuttler 1995; Kuttler & Weber 2006), or negative effects on the habitats of urban flora (Wittig 1998) and fauna (Klausnitzer 1998).

1.3.3 Properties of urban soils

In addition to water and air, soil is the third 'environmental medium' which closely interacts with the other ecological compartments. This interrelationship does not depend on the degree of anthropogenic modification or interference. As mentioned above, the soil plays a key role in the geo-ecosystem because it functions as a sink as well as a source and has the capacity to transform substances (see Fig. 1.3.4) (Hirner *et al*. 2000; Pavao-Zuckerman & Byrne 2009). Placing humans at the centre of the urban ecosystem, the previous section illustrated that human beings are confronted with a wide variety of exposure pathways and effects that begin in the soil (see Fig. 1.3.1).

On the basis of the above definition of urban soils, the following sections describe how natural soils, soils from artificial substrates, and sealed soils may be modified in the urban environment.

1.3.3.1 Urbanization challenges: natural soils

Settlement activities and expansion can have various effects on the pedosphere, for example by significantly increasing stoniness (Stasch & Stahr 2002), raising humus contents and promoting compaction (Lehmann 2002), or creating 'new' soils by mixing or covering with foreign materials (Fetzer 2002; Pickett & Cadenasso 2009). Initially, only natural soils were or are affected by these changes (Blume 1998), especially during the early phase of settlement. However, settlement activities may also affect previously modified soils. This occurs (or occurred) for example when mostly agriculturally used land is consumed as settlements expand. In this case these soils are called 'pre-urban soils' in which the historic land-use plays a significant role in terms of the substance balance.

Natural soils can be modified in many ways in the urban setting (Brady 2004; New York City Soil Survey Staff 2005; Norra & Stübben 2003), for example by:

- building over and paving (soil sealing);
- filling (back-filling, filling, cultural debris; fill materials can consist of: ash, refuse, building rubble, slag);

- raising (addition of soil materials, relevant in gardens and parks, for example);
- excavation: removed or capped (excavated soil);
- compaction (mechanical soil compaction in response to loads: machines, vehicles, levelling, tread);
- dessication (cause: anthropogenic lowering of groundwater levels);
- relocation (construction work);
- mixing (soil cultivation, construction work);
- contamination (accidents, leakages, immissions, contaminated sites, construction work, de-icing salts).

These modifications can have a profound impact on the following urban soil attributes:

1. Substance concentration:
   - solid addition—natural and technical substrates, or mixtures of these;
   - substance inputs—gasses, solutions, or solids, from the atmosphere, industrial and settlement areas, traffic, infrastructure facilities;
   - contaminant transfer;
   - humus formation and lowering of the groundwater table.
2. Exchange of substances between soils and the other spheres (atmosphere, biosphere, hydrosphere, lithosphere):
   - climate change;
   - soil compaction and sealing;
   - changes in the catchment area;
   - changes in the distance between soil surface and groundwater table.
3. Alteration of natural processes and characteristics:
   - anthropogenic spatial patterns;
   - vertical and horizontal heterogenization;
   - anthropogenic controlled relief modifications.
4. Period of development and the frequency of land-use change.
5. Modifications of the contaminant storage and transfer functions.

To sum up the previous sections, one can characterize the properties of urban soils as follows.

The typical distribution pattern in the urban settlements is a small-scale mosaic, which can change dramatically from metre to metre. As urbanization progresses, so too does the impact on soil structure caused by construction work, mechanical load, as well as foreign substance and contaminant input. This is accompanied by a decrease of bare soil surface area. Where present (in gardens, allotments, parks) these open areas are associated with a very wide range of soils from humus-deficient soils on tipped substrates to those that are dark, humus-rich, and fertile (e.g. as a result of intensive, artificial fertilization). Most urban soils have a neutral pH-value because of the presence of carbonaceous building rubble and wind-blown dust. A thin humus cover, low soil organism populations, and mechanical soil compaction all have an adverse effect on soil aeration. Reducing the pore volume also decreases the soil water storage capacity and the infiltration rate—the soil becomes more impervious to water. This results in increased runoff, as stormwater or water flowing from sealed surfaces is only partially absorbed by these soils (see Illgen, Chapter 1.4). Mud transported by these overland flows clogs the pores of the topsoil, further aggravating the situation. Urban soils may be contaminated by inputs from the atmosphere, rainfall and dew, floods (especially soils of the riparian zone), contaminated sites, de-icing salts, leakages, accidents, improper storage, and over-fertilization. Other forms of contamination are acidification from acid rain and accumulation of typically urban substances such as heavy metals (lead, copper, zinc, nickel, manganese, cadmium) or organic compounds (PCBs, PAHs) (Wilcke 2000; Linde *et al*. 2001; Mirsal 2008).

1.3.3.2 Soils of technical substrates

In addition to the re-located natural solid and unconsolidated materials, many anthropogenic urban soils also contain a wide range of artificial substrates. These technical materials are products of a variety of technological processes (Blume 1998). This provides a basis for categorizing the main components for example: building rubble, slag, ash, tailings, refuse, and mud (Hiller & Meuser 1998).

Technical substrates are difficult to identify in the field and therefore pose a problem for the urban soil surveyor. As a result, many soil descriptions contain imprecise classifications of the materials (e.g. slag/ash), which fail to take into account the different properties, especially concerning the contaminant potential. A number of different identification keys are available at present for differentiating manmade and natural substrates in urban-industrial soils (Meuser 1996; Schwerdtfeger 1997). Typical properties of these soils can be:

- conspicuous odour;
- dominant substrate colour;
- hardness;
- surface structure and internal structure of the fragments (substrates > 2mm);
- particle size; and
- carbonate content.

Technical substrates (also called 'anthropogenic stones'—Meuser 2002) do not often occur as monosubstrates. Usually they form heterogeneous mixtures, which often vary between horizons or layers. An analysis of approximately 900 soil horizons sampled from depths between 0–100 cm in the Ruhr Region revealed that a third of all horizons were free from technical substrates (naturally developed horizons, re-located natural substrates; Hiller & Meuser 1998). Similar results were also obtained in a study of the urban environment in Halle, where about 40 per cent of the upper soil horizons (sampling depth < 1 m) contained no artificial substrates. An analysis of the anthropogenic relief modifications in the city of Halle showed that many areas were filled with materials containing a large proportion of artificial substrates (mainly tips and dumps). Purely technical mono-substrates were found in less than 2 per cent of the sites. Other studies have confirmed that anthropogenic modified soils primarily contain a mixture of artificial substrates (Holland 1996; Pluquet & Lenz 1997; Tietböhl *et al* 1997; Siem 2002).

1.3.3.3 Sealed soils

According to Wessolek (2001), soil sealing involves compacting the bare soil and/or covering it with more or less permeable substances, such as asphalt, concrete, or buildings. These practices effectively block exchange processes between atmosphere and soil, which has a profound effect on the abiotic properties (water infiltration and evaporation, aeration) and the biotic components (see Illgen, Chapter 1.4).

Wessolek (2001) identifies three forms of sealing found in urban soils:

- full sealing (horizontal and vertical), for example roads, squares, buildings, channels, and so on;
- partial sealing, for example paving, paving stones, grass pavers, and so on; and
- subsurface sealing, for example underground car parks, tunnels, conduits, and so on.

Sealed soils are usually removed before construction and are condensed. Under full sealing, the soils are fossilized. However, in full sealing using 'porous' materials or in partial sealing, contact between the soil and the atmosphere remains and in this case the soils may retain roots (for example of street trees). Overgrown joints between covering materials are more permeable than vegetation-free, condensed joint fillings. Such soils could, however, be difficult locations for plants and their roots as pollutants (heavy metals, pesticides, bedding salt) also enter these soils, in addition to air, water, and dissolved nutrients. Contamination from defective storage tanks and pipes (for example gasoline) may also occur. On totally sealed surfaces, soil substrates and man-made substances can also accumulate as dust particles across the surface. On flat roofs, for example, this dry dust can accumulate loosely, creating a soil with a high heavy metal content.

1.3.4 Genesis of urban soils and soil functions in urban ecosystems

The previous sections have illustrated that the genesis of soil forming substrates in urban ecosystems may be autochthonous (created at the location) as well as allochthonous (created away from the location, both natural and artificial material). These substrates not only determine the type, intensity, and speed of pedogenesis but also affect the ecological potential of these locations. The effects of this correlation become apparent in the water and nutrient balances of these soils. These properties are the key to many other material and energetic soil processes in the urban ecosystem (such as potential infiltration and groundwater recharge—see Wessolek 2001).

In some cases, these (direct or indirect) effects on the urban soil, and the resulting modification of the soil ecological properties and functions, are intended. In other cases, they may be unanticipated and appear as 'negative side effects'. The main properties affected by sealing are the soil energy and especially the water balance.

Sealing not only modifies the substrate but can also affect natural soil forming processes. Therefore sealing triggers a shift from natural to urban soil formation involving modified processes such as:

- humus enrichment: production of Regosole (carbonate free) and Pararendzinen (carbonaceous);
- carbonate enrichment, mainly from building rubble: production of weak to strong alkaline soils—these are classified as Pararendzinas just like their natural counterparts;
- manmade substrate mixtures with natural soil: production of Phyrolithe (Phyro: mixture of natural and artificial substrates);
- manmade substrate deposits (building rubble, ash, etc.): production of Technoliths;
- waterlogging above impervious layers: production of Pseudogleye;
- reductomorphic processes as a result of anaerobic conditions, for example methane generation, production of Methanosole (belong to the group of reductosols).

Soils form the life basis for many organisms. They are permeated by roots, provide support and warmth, and supply the plant with water, oxygen, and nutrients. Soils also regulate the water balance and filter, buffer, or eliminate contaminants, thus preventing these from harming organisms, entering crops, and polluting the groundwater or nearby bodies of water. These functions are severely curbed in the urban landscape. In this setting, soils are primarily seen as an infrastructural element, a foundation for buildings, industrial and business premises, roads and railway lines. In most cases, they have been sealed and thus lack vitality. Urban soils are also used for landfills for the disposal of solid and liquid wastes. Examples of this type of use are mining spoil heaps, building rubble and refuse tips, and sewage irrigation fields. The remaining areas under near natural management are usually only small and used as parks, cemeteries, gardens, playing fields, and playgrounds as well as allotment gardens and agricultural fields. This differentiation concurs with the urban structural unit approach (see Pauleit and Breuste, Chapter 1.1). To a large extent, these remaining areas determine the life quality of the town population, because only these soils provide the appropriate conditions to sustain greenspaces, which offer opportunities for recreation and stimulation and promote health (see Tzoulas and Greening, Chapter 5.2). In addition to this, these soils regulate and filter water and, therefore, play a vital role in recharging the groundwater.

Many soils of the urban open spaces bear little resemblance to those of the pre-urban landscape. They have been altered to such an extent that their ability to function in the landscape or urban ecosystem is severely restricted. The Soil Protection Act of the Federal Republic of Germany (BBodSchG 1998) contains a categorization of soil functions (Table 1.3.4) and acknowledges that soil conservation must be understood in terms of protecting soil functions. However, it quickly transpires in practice that protecting one soil function can quite often adversely affect another function and can only be achieved by ignoring the effects of land-use. For example, the consequence of protecting soil locations as an integral component of the ecosystem is that these areas cannot be used for building settlements or roads. From the urban ecological point of view, it is therefore exceedingly difficult to evaluate the relative importance of the various soil functions, because human activity and impact is the prime characteristic of the urban ecosystem.

A possible means of soil protection at a local level is the application of soil concept maps, in which

Table 1.3.4 Soil functions according to the Soil Protection Act of the Federal Republic of Germany (BBodSchG 1998, modified)

1. Natural function as	a) natural resource and habitat for humans, animals, plants, and soil organisms
	b) component of the ecosystem and water and nutrient cycles in particular
	c) medium for decomposing, regulating, and synthesizing materials as a result of its capacity to filter, buffer, and transform substances—key functions for protecting the groundwater.
2. Archival function as	record of cultural and natural history
3. Land-use functions as	a) raw material
	b) space for settlements and recreational activities
	c) location for forestry and agriculture
	d) location for other industrial and communal activities, traffic, supplies, and disposal.

suggestions are made for priority areas and soil protection measures. Three main goals should be pursued in such maps:

1. Conservation of particular soil and area qualities, including multi-functionality, mono-functionality, proximity to nature, and the possibility of restoration. Connectivity and the degree of network formation are important spatial issues.
2. Avoidance of degradation of soils and sites. Degradation of soil properties should be particularly avoided and restoration encouraged.
3. Mitigation of risks of soil degradation.

The value of protection is reinforced, in the case of qualitative and spatial peculiarities in particular, by the need for protection from degradation and the scale of previous land-use impacts.

1.3.5 Urban soil landscapes

The distribution of soils in urban areas is only marginally comparable to that of non-urban landscapes—each construction alters the former natural 'pre-urban' soil conditions. In an attempt to describe the systematic nature of these modifications, Urban Structural Units (see Pauleit and Breuste, Chapter 1.1) were applied to the pedosphere. Besides age and type of land-use, each structural unit is also characterized by specific distribution patterns, building densities, degree of sealing, as well as a particular vegetation structure. The presumption is that these properties should have a characteristic effect on the underlying soils. Therefore, two basic factors determine the character of an urban soil: history and land-use (Jäger 1997). In combination, these factors can be identified from the urban structural unit as well as by analysing the mass balance, whereby it is important to identify when and for how long specific substances have been introduced to the soil.

In order to evaluate the degree of soil modification, it is useful to augment soil profile descriptions with an analysis of specific mass balance parameters, which are indicative of the type and intensity of the modification. Particularly helpful indicators over and above classic soil parameters are heavy metal and organic compound contents, because their presence correlates with specific activities and times. From the viewpoint of a geo-systematic (space–time) approach, these indicators can be utilized to assess the extent and quality of the soil modification. However, any evaluation based on mass balance assessments must take into account that soils can function as a material sink, source, and converter depending on the parameters and time (Fig. 1.3.2).

Soils can function as both a material sink and material source. The entry of material into the soil can take place though emissions from the atmosphere, for example as fine dust, as erosion or, as is frequent in cities, through direct entry (section 1.3.2 in this chapter). In the soil, transformation processes or translocation processes can take place according to local conditions (Fig. 1.3.3). Substances can be transformed in the soils through translocation processes both horizontally and vertically.

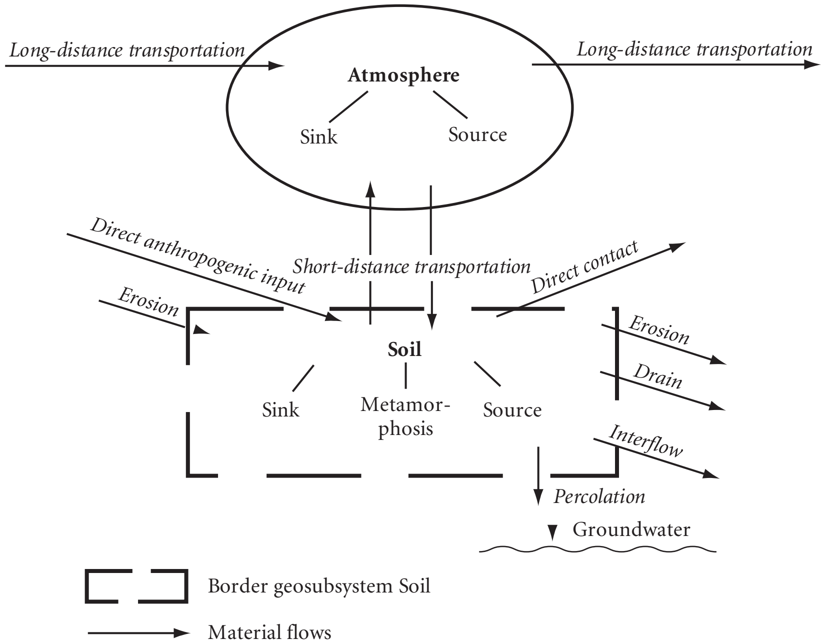

Figure 1.3.2 Soil as an ecosystem compartment with relation and interdependences to the other compartments (Sauerwein 2006, modified)

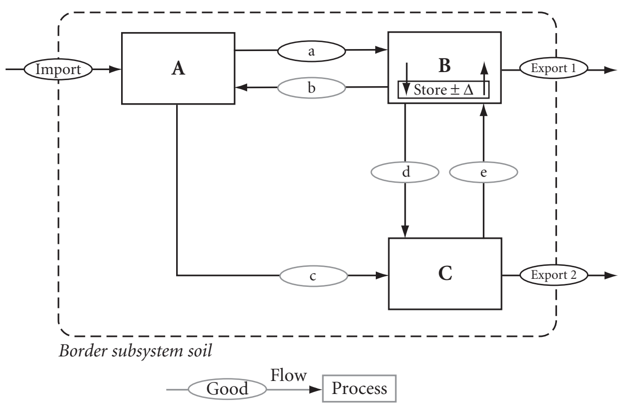

Figure 1.3.3 Material flow in the geo-eco-subsystem soil: pedogenetical processes can transform and translocate materials within the soils (Sauerwein 2006, modified)

Depending on the local situation, individual substances can concentrate and, if pedological conditions permit, substances can move elsewhere in the soil or end in up in the aquatic system. Thus, urban soils are parts of the urban ecosystem and interact with other parts of the ecosystem, such as aquatic systems.

These considerations form the basis for differentiating urban zones in which soil characteristics follow a quantifiable pattern. In accordance with the geographic-geo-ecological landscape concept, these 'urban soil landscapes' are defined as follows: 'Urban SOIL LANDSCAPES are anthropogenic influenced segments of the pedosphere characterised by similar parameter distribution patterns that are the product of urban landscaping activities, their pre-urban properties, age and current as well as historic use.' (Sauerwein 2006)

1.3.6 Balancing the soil substance budget in settlements

As shown above, it is necessary to measure input, soil pool (as a sink and source), and output in relation to temporal processes in order to balance the soil substance budget.

Potential inputs include wet and dry deposition, flooding, direct input (waste disposal), and input via percolation/interflow (in landscaped areas). Potential outputs include extraction by plants, leaching, and runoff/erosion. However, it is next to impossible to measure all inputs and outputs empirically. On the one hand, additional research is required to ascertain the 'correct' measurement techniques for quantifying the parameters mentioned above. On the other hand, it is especially difficult to find and use representative surfaces in an urban setting due to difficulties in getting permits, and vandalism. The measurement of dynamic processes such as leaching is extraordinarily difficult (see Illgen, Chapter 1.4; Koch *et al* 2004). Substance balances must be based on continuous long-term measurement series in order to take into account the variability of the climatic effects. This is not a viable option financially or technologically in a major city. One practical approach for balancing substance losses by leaching from the soil takes into account urban structure units. An alternative method involves identifying substance sinks, which retain a chronological record of the substance balance. There are two such sinks in the urban area: the soils themselves and the groundwater. The latter has been the object of intense research (see Illgen, Chapter 1.4). A study carried out in the municipality of Halle showed that substance concentrations measured in selected groundwater monitoring wells varied according to the urban structure unit. These results were also confirmed in continuous long-term studies (Koch *et al* 2004). These findings also indicate that current urban soils function both as a sink as well as a source for substances.

Achieving a 'real' balance would require setting up a monitoring system for quantifying the most important input, transfer, and output parameters. Control plots are a useful monitoring element for assessing quality and quantity of atmospheric substance inputs. Additional discussions of this topic can be found in Hertling and Raschke 1995; Clemens *et al*. 1997; Kiene and Miehlich 1997; Huinink 1998; Bartsch *et al*. 1999; Gröngröft *et al*. 2000; and among others.

1.3.7 Classification of soils in settlements

There are several international soil classification systems which are, however, only partially comparable (Eswaran *et al*. 2003). The Soil Taxonomy (Soil Survey Staff 1999), and the World Reference Base for Soil Resources of the FAO (http://www.fao.org), are at the most commonly used. In neither classification, however, are urban soils explained. In Germany, there has been intensive research into urban soils for 30 years, but there is still no final consensus on soil classification (Burghardt 2002; Ad-hoc-AG Boden 2005). A possible classification emerges through the change of formerly natural locations and soil qualities to manmade ones (see section 1.3.4 of this chapter). Such classifications have to be constructed by soil scientists, and landscape ecological conditions and ecosystem properties also need to be considered.

The following suggested procedure is not a 'new' classification system for urban soils. The objectives of this geo-ecological soil classification are different from the systems available for soil mapping. However, this new approach is not intended to compete with current concepts—it is proposed as an additional tool for characterizing urban areas. At the same time this approach incorporates results from precise soil surveys. The proposed scheme for classifying urban soil landscapes is outlined in Fig. 1.3.4. Accordingly, an urban landscape is a (three-dimensional) segment of the pedosphere characterized by a single urban soil unit.

The scheme outlined in Fig. 1.3.4 shows the basic conditions and parameters required for determining an urban soil landscape. The classification is based on the three principle criteria: 'urban constraints', 'location quality', and 'local burden'. Each of these criteria can be characterized by means of specific indicators. The basic urban conditions are influenced by history and historical use. The urban structure unit approach is an instrument for characterizing these indicators. Local site conditions are determined by the soil profile (substrate type, bedding, and thickness). The level of naturalness is an indicator which describes the degree of anthropogenic disturbance of the natural soil. Local contamination

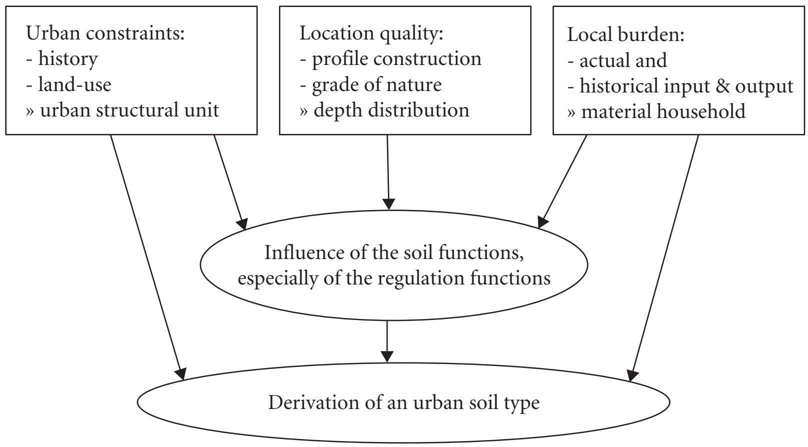

Figure 1.3.4 An approach for classifying urban soil landscapes (Sauerwein 2006, modified)

depends on historical and current contamination. Although the urban structure unit approach can help to describe 'relative' contaminations, exact characterizations must be based on measurements of the most important substance budget parameters. Contamination levels can be described using the following parameters: humus content, lead, copper, and zinc, as well as PAH concentrations.

An 'urban soil type' in the sense of an urban soil landscape, is therefore characterized by the following properties:

1. urban structure unit
2. anthropogenic landscaping
3. degree of naturalness
4. pre-urban soil type
5. vertical distribution of substance budget parameters
6. soil unit (e.g. of soil taxonomy or world reference base).

This list clearly shows that this characterization is not a 'rigid' classification but can be used to differentiate a wide range of urban soil landscapes. The system makes it possible to distinguish several urban soil types within a single urban structure unit, which has been subjected to a variety of landscaping activities. Similarly, different pre-urban soil types within a single urban structure unit will be classified as different urban soil types.

When surveying urban soils it is necessary to have access to the following spatial information before commencing with the fieldwork:

- urban structural unit survey;
- pre-urban soil map; and
- survey of anthropogenic landscaping activities.

Geographic Information Systems (GIS) allow the user to superimpose this spatial information and thus to identify individual potential survey areas. The next step involves a detailed survey of these 'disjointed' areas by digging pits or by auger to establish the remaining properties, degree of naturalness, and soil unit. Ideally the soils should be sampled and analysed in the laboratory to examine the vertical distribution of substance budget parameters.

In practice, it is next to impossible to carry out a comprehensive survey of this type in a city. However databases and GIS are useful tools that enable the surveyor to collect extensive spatial data to turn a 'mosaic' into a 'patchwork rug' containing a high level of information about anthropogenic soil disturbances.

1.3.8 Urban soil protection concepts

The current approaches for distinguishing, recording, and evaluating soil quality in the urban landscape are generally unsatisfactory, because there is no universally accepted scientific base. In most approaches the soil is seen as an interface between the abiotic (geology) and biotic components (living organisms) of the element cycle. From the urban-ecological and system-theoretical point of view, these basic functions must be preserved. Humans use the soil and some of its functions for various purposes. An approach that further distinguishes between 'soil functions' would compare the natural soil system with the properties due to anthropogenic land-use. The basic problem, especially in the urban setting, is the competition between individual soil functions. In the end, it is a political decision to resolve for each area, or even location, which soil function should be prioritized. From the urban-ecological point of view, these decisions should be based on the spatial distribution and quantitative potential of the soil function.

The following example illustrates the problem. A well-known phenomenon in towns is the lowering of the groundwater table in comparison to the surrounding region (see Illgen, Chapter 1.4). This change is determined by the degree of soil sealing. Infiltration and therefore groundwater recharge decreases as the degree of sealing increases. Consequently, decreasing the sealed area within an urban structural unit would result in a local increased recharge. To achieve the same effect on the overall urban water budget requires that at least 5 per cent of the town area is unsealed. However, unsealing previously sealed areas can have an adverse effect on the underlying soil. The substances found in the sealed soil are usually immobilized (retained) and isolated from the substance cycle. As these surfaces are unsealed, highly mobile substances/compounds may enter the substance cycle and develop into a potential source of contamination. This example shows that urban soils possess important regulatory and buffer functions and that any interference with the current soil substance budget can lead to a wide range of qualitative and quantitative consequences (Kunst *et al*. 2002).

Soils play a central role within the ecosystem: they are the basis for human, animal, and plant life and a medium with important functions in the nutrient and water cycles. Soil is an easily destroyed, non-renewable resource. Intense land-use is often accompanied by an irreversible loss of important soil functions for future generations. New soil conservation concepts are aimed at counteracting these problems and provide the basis for taking soil properties into account in overall spatial planning (ELSA 2007; ICLEI 2007).

The soil resource is not literally used up by anthropogenic activities. Some proportion remains in the original position while other parts are excavated and transported to other locations. Depending on the type of activity, disturbances usually have an adverse effect on soil properties by causing erosion or compaction and changing substance inputs and outputs, which in turn can lead to a restriction or complete loss of important soil functions. The interference is not limited to the surface but affects the whole three-dimensional functional space. This results in a reduction of soil quality, which, in contrast to soil quantity, cannot be restored. The problem thus described is usually ignored in the daily management of the soil resource. In many cases, the soils are privately owned. Therefore, they are an integral part of the economic value chain, in which ecological aspects of the soil play only a minor role: their function is limited to providing the ground for settlements, industry, and infrastructure. Despite their ecological importance, soils do not receive the appropriate attention and esteem generally given to the other compartments: water and air. The central problem is that the urban community no longer depends on the soil for their everyday life. Prior to the transformation of Central Europe from agricultural to industrial states at the beginning of the twentieth century, soils were an important regional resource, providing food and livelihood for large parts of the population. However, rapid technological and scientific progress in the following years led to a severe rationalization of the primary sector. Today, only a small proportion of the population works in the highly productive agricultural sector. Particularly in cities, soils are mostly only used as a resource for construction activities. The traditional inhibitions associated with the soil resource have all but disappeared. This trend is observable on a global scale.

Sustainable soil conservation is not feasible without broad acceptance and support from the community. When compared to the public attention given to water and air, soils have a lot of catching up to do, especially in urban areas. 'Soil awareness' should not be underestimated and must be taken into account when developing soil conservation concepts. Conceptual soil conservation based on soil function evaluation is still in the developmental stage. Ultimately, the success of communal soil conservation concepts depends on communication and an exchange of experiences between the different institutions and political instances. After all, the objective is to preserve a vital resource.

CHAPTER 1.4

Hydrology of Urban Environments

Marc Illgen

1.4.1 Introduction

Water is the most vital element on Earth since it is crucial for the survival of all living organisms. Besides its fundamental life giving relevance, water also facilitates and considerably determines the development of human societies. Urban development as well as economy, public health, and modern living are closely related to the availability and quality of regional water resources.

The natural occurrence, distribution, movement, and quality of water on or near the land surface are the subject of hydrology as an earth science discipline. Hydrology mainly deals with the circulation of water and its constituents through the so-called hydrologic cycle. It covers the phenomena of precipitation, evaporation, infiltration, subsurface flow, surface runoff, and stream flow including the transport of dissolved or suspended substances.

This chapter describes the general hydrological processes and phenomena of urban environments. The major components of the urban water cycle, together with the particular processes occurring between atmosphere, surface, and subsurface are explained in the context of land-use characteristics. With regard to the specific infiltration and runoff performance of urban surfaces fresh knowledge and latest innovations are presented. The later portion of the chapter addresses the water balance of urban areas. The stormwater retention capacities of paved and unpaved surfaces, which ultimately control the surface runoff process, are specified from an urban water management perspective. Characteristic values of the particular water balance components on an annual basis as well as event-based runoff coefficients are outlined for several types of surfaces, reflecting the interactions between the atmospheric impact by rainfall and the hydrological phenomena on urban areas.

1.4.2 Urban water cycle

Water is always in movement. It circulates continuously between the atmosphere, land and water bodies, thereby changing state among liquid, vapour, and ice. This movement is described by the water cycle—the most fundamental principle of hydrology (Fig. 1.4.1).

The global water cycle is driven by solar energy and gravity. Water in the oceans and on the land surface is heated by the sun and can evaporate or be transpired as vapour into the atmosphere. Even though this process is generally not visible, its effect is obvious through the formation of clouds and discharge of precipitation in the form of rain or snow. Through several pathways on and under the land surface the water re-enters the oceans or reaches the atmosphere again. The terrestrial pathways comprise infiltration and surface runoff, groundwater and stream flow, as well as evaporation and transpiration from the surface.

The water cycle is a vivid model to conceptualize and explain the process chain between precipitation and terrestrial water resources. Nonetheless, it also comprises a multitude of extremely complex processes and mutual interactions of the media of water, air, and soil. Understanding these processes requires knowledge of climatology and meteorology, of geology and soil science, of fluid mechanics and stream flow hydraulics.

The various components of the water cycle are influenced by an infinite number of factors. The

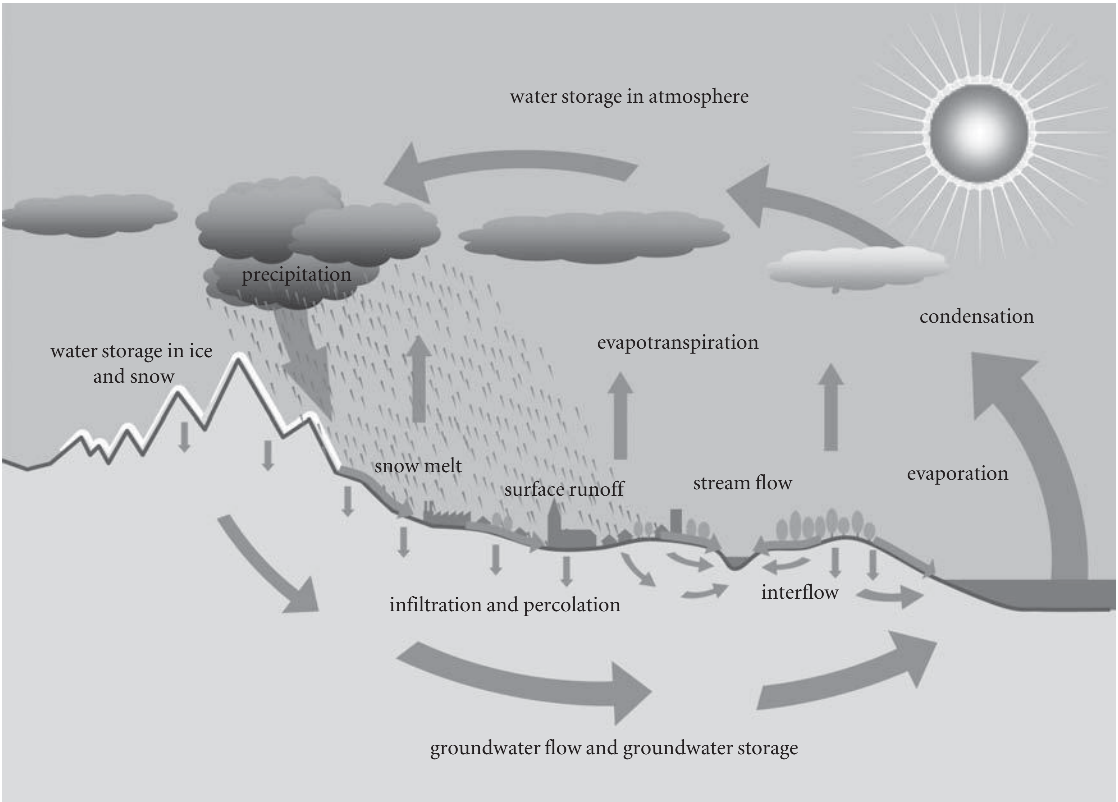

Figure 1.4.1 Simplified scheme of the hydrologic cycle

hydrological response of a particular watershed on a certain rain event, for instance, depends not only on its geomorphology, topography, and land-use, but also on present and antecedent meteorological events.

The landscape and land-use of a catchment area exhibit the primary influence on the observed hydrological processes. Hence, the hydrologic cycle of urban environments significantly differs from natural watersheds. In urban environments natural land surfaces have been replaced by paved and impervious surfaces to allow convenient living and working (see Pauleit and Breuste, Chapter 1.1). Roads, roofs, and pavements reduce the retention and infiltration capacity formerly provided by soils or vegetation. They increase the generation of surface runoff and effect a modified evapotranspiration regime. Furthermore, urbanization can contribute to a deterioration of water quality due to air pollution and leaching of contaminants from and beneath the ground surface.

The magnitude of these impacts on the water cycle depends on the extent of surface sealing. For instance, the infiltration and percolation ratio of natural ground cover, typically around 50 per cent, can be reduced to around 15 per cent in a city setting where the ground surface is intensively paved. Stormwater runoff in an urban environment can increase from around 10 per cent to 50 per cent of the precipitation, while a similar reduction in the evapotranspiration ratio can be expected.

The increase of surface runoff, together with the necessity to collect and treat wastewater, has led to a sophisticated drainage infrastructure in our cities. The basic functions of an urban drainage system are the preservation of hygienic conditions and the mitigation of flood damage. For these purposes, underground channel systems and wastewater treatment facilities have been installed since the middle of the nineteenth century. Today, urban hydrology as a specific discipline of hydrology incorporates the management of stormwater runoff in urban areas and the design of the drainage infrastructure.

1.4.3 Hydrological processes in urban areas

1.4.3.1 Precipitation

The atmospheric process of precipitation is one major component of the hydrologic cycle. Even though the hydrologic cycle is a true cycle with no beginning or end, precipitation is often seen as the initiating and driving process of hydrology.

Precipitation describes the deposition of condensed atmospheric water vapour and occurs when the content of vapour in clouds has reached a point of saturation. Precipitation is commonly divided into frontal, orographic, convective, or cyclonic, according to the mechanism by which the air masses are lifted from lower to higher altitudes, and can occur in various forms including snow, hail, or rain. In urban hydrology, rainfall is usually the most prevalent form of precipitation and therefore the dominating factor in the design of urban storm drainage facilities to prevent flooding (Akan & Houghtalen 2003).

The precipitation process is a complex interaction of many regionally changing variables such as temperature, pressure, and density. Consequently, precipitation exhibits an enormous variability in space and time. Local pluvial conditions are a result of regional climate and weather systems. The annual amount of precipitation, as well as its distribution over the year, can vary significantly between different regions and locations (see Parlow, Chapter 1.2). Furthermore, precipitation patterns are not uniform among particular regions of similar climate and may significantly vary within a range of less than 25 km. In coastal environments and at higher altitudes, annual precipitation depths tend to be higher, reflecting the geographically and topographically related variation of local pluvial conditions (Mansell 2003).

From a less meteorological but more hydrological point of view, precipitation represents a given or expected input to a hydrological system. Once precipitation reaches the land surface it is transformed into a water flow on the surface, into the ground, or back to the atmosphere. Rain events are generally characterized in terms of the total rain depth, rain intensity, and rain duration, all of which vary from one event to another. Moreover, the rain intensity can vary significantly during a rain event. To cope with this enormous spatial and temporal variability and the unpredictability of future rain events, hydrology generally applies local precipitation data, either in the form of measured data series or in form of statistical values. In particular, statistical values that correspond to a specific time scale and return period are very common; for example average annual precipitation depths or average rain intensity of a 50-year storm event with a particular duration. With regard to the design of urban drainage systems, primarily statistical values relating to rain durations of less than 12 hours (mainly 15 minutes to 4 hours) are of particular importance to reflect rain intensities of short-term storm events. Statistical precipitation data are widely adopted to represent the principle characteristics of the local or regional pluvial regime and are applied to evaluate present and future hydrological systems by assuming that average pluvial conditions will not change in future.

The probabilistic correlation between average rain intensity, rain duration, and return period, for instance, can be represented by intensity–duration–frequency (IDF) curves extracted by frequent analyses of historical data series for a particular location. An example relating to a Central European city is given in Fig. 1.4.2.

The curves represent the typical precipitation phenomenon that the average rain intensity of a particular return period decreases with ongoing duration, especially within the first 30–60 minutes. Short-term storm events resulting from convective cloud formations are usually associated with much higher rain intensities than long-lasting orographic induced rain events. The average rain intensity of a particular rain duration increases with the return period; the more so the shorter the particular rain interval is. In regions of a temperate climate, storm events providing high rain intensities larger than

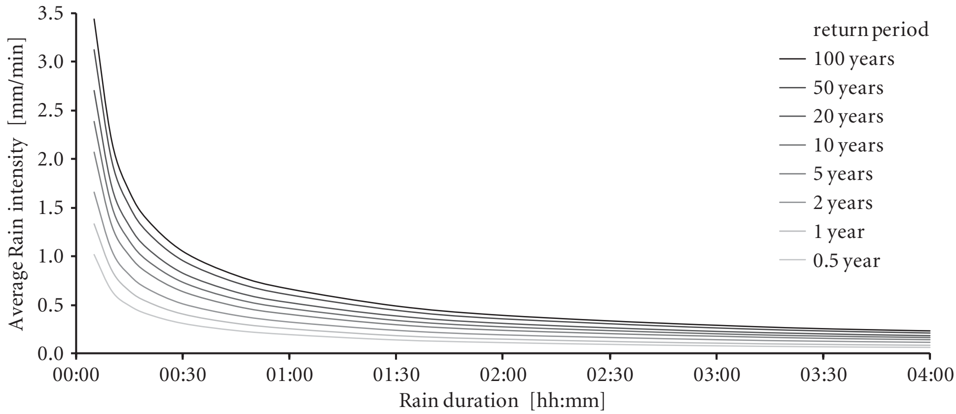

Figure 1.4.2 Example of intensity–duration–frequency (IDF) curves (based on DWD, 1997)

0.2 mm/min only account for 10–25 per cent of the annual rainfall depth. Most of the annual precipitation falls at much lower intensities. This is of particular importance when considering an average annual or an event-based water balance of a particular watershed, as the rain intensity considerably determines whether the fallen rain can infiltrate, evaporate, or run off.

The aforementioned characteristics and relationships can be used to classify local or regional rainfall characteristics for hydrological purposes. They are also used to define a design storm out of a historical data series or even to generate a synthetically designed rainfall in order to evaluate the hydrological processes of an urban watershed.

1.4.3.2 Evaporation and transpiration

Evaporation describes the transformation process of a liquid water phase into water vapour. This transition process is driven by solar energy and occurs when water molecules in a liquid phase have achieved sufficient kinetic energy to overcome attractive intermolecular forces and to leave the water surface. Transpiration describes the evaporation of water from aerial parts of plants through diminutive pores, leaves, stems, or flowers. For practical reasons, evaporation and transpiration from water bodies, soils, vegetation, and other surfaces are referred to together by the term 'evapotranspiration' (Dingman 2002).

From a hydrologist's perspective, evapotranspiration represents another major component of the hydrologic cycle. Both water contained within the oceans and water on or near the land surface evaporates into the atmosphere where clouds are formed and subsequently precipitate liquid water. Evapotranspiration also contributes to a significant water loss from natural or urban watersheds, as it may account for up to 70 per cent of annual precipitation.

The governing factors controlling the evaporation process are temperature, humidity, air movement, and solar radiation. The kinetic energy of a molecule is proportional to its temperature. Hence, the latent heat required to initiate the phase transition will be attained more quickly for water at higher temperatures, resulting in higher evaporation rates. A limiting factor of the evaporation process is the number of water molecules that are located close enough to the water–air interface. The evaporation rate, in particular, reflects the difference between a vaporization rate controlled by water temperature and a condensation rate controlled by vapour pressure of the surrounding air. Higher rates of evaporation will occur when the air above a water table is at a lower state of saturation. In the case of full saturation of the surrounding air, both rates are equal and evaporation will cease. The vapour pressure that indicates this state of saturation is related to air temperature. For this reason, the humidity of the surrounding air masses as well as the wind speed play a key role with regard to the current evaporation rate (Shuttleworth 1992). To exemplify these processes and interactions with a real life example, imagine laundry hanging outside on a clothesline; due the processes explained above, it will dry faster on windy days and at higher temperatures.

During evaporation energy is adsorbed due to the work against intermolecular forces. Molecules of higher energy leave the liquid phase and reduce the kinetic energy of the remaining molecules as well as the temperature of the liquid. This phenomenon is called 'evaporative cooling' and is the same process that cools our body when we sweat. Evaporative cooling plays an important role with regard to the optimization of urban climates.

It is difficult to directly measure the rate of evapotranspiration over a large area (see Parlow, Chapter 1.2). Commonly, evapotranspiration is measured by point measurements using evaporation pans or lysimeters. An up-scaling of point data on a meso-scale over an entire natural or urban watershed is problematic and associated with enormous uncertainties (Grimmond & Oke 1998). To describe the intensity of the evaporation process, two standard evaporation rates are commonly defined: the potential evaporation rate and the reference crop evaporation rate. Even though both rates represent idealizations, they are widely used to estimate evaporation in a particular environment. Potential evaporation is commonly defined as the quantity of water evaporated per unit area and unit time, from an idealized spacious free water surface, under existing atmospheric conditions. It represents the amount of evaporation that would occur from a given surface assuming an infinite supply of water to be evaporated. Reference crop evaporation can be defined as the rate of evaporation from an idealized grass crop with particular crop height, albedo, and surface resistance, and serves as reference for estimating evaporation of vegetated surfaces (Shuttleworth 1992).

Since evaporation is driven by solar energy and closely related to current temperature and humidity, evaporation rates vary in diurnal and seasonal patterns. An example of a seasonal pattern related to Central Europe and examples of diurnal patterns of suburban districts in three North American cities are given in Fig. 1.4.3 (based on Grimmond & Oke 1999; Sieker *et al*. 2006).

In urban environments, evaporation and transpiration are significantly affected by the sealing of the surface and the restricted availability of potentially evaporating water. Within a city area, both the average annual and daily evaporation rates vary locally depending on the particular urban development. In the city of Berlin, for instance, annual evaporation depths can vary between two adjacent city blocks from below 50 mm to above 300 mm, depending on the impact of corporal elements such as buildings, streets, or trees that influence the prevailing

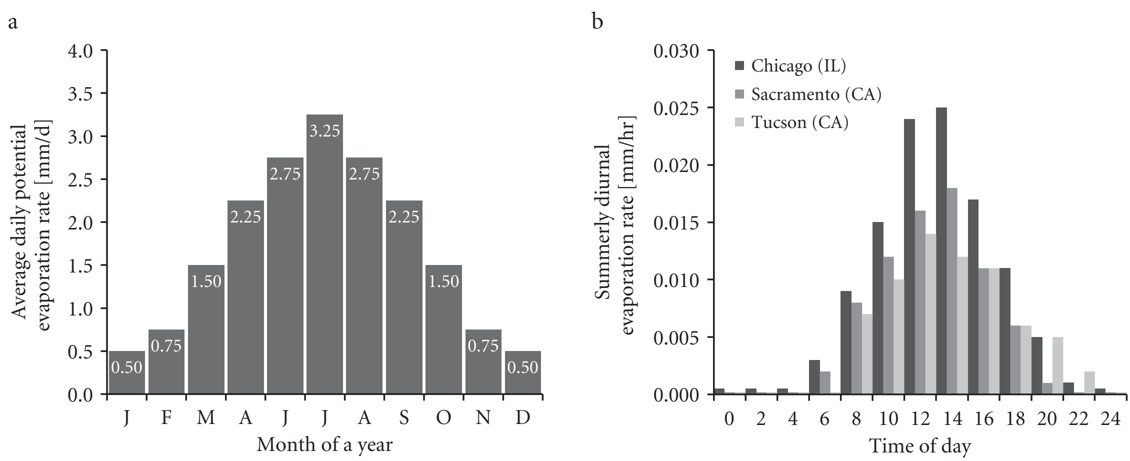

Figure 1.4.3 Seasonal (a) and diurnal (b) patterns of local evaporation rates

microclimate (City of Berlin 2009). At downtown, commercial, and industrial urban sites, evapotranspiration is commonly lower than in suburban areas due to the limited areal coverage of vegetation and the reduced amount of potentially evaporating water retained on paved surfaces (Harlass & Herz 2006). Compared to nearby rural areas, evapotranspiration rates are typically lower in the city, reflecting the general relationship between the evapotranspiration rate and the ratio of vegetated surfaces. Nevertheless, intra-urban evaporation rates can also be higher than in rural or residential neighbourhoods, underlining the local variability of evaporation processes. In addition, built-up areas are typically 1–5 °C warmer than surrounding rural or natural sites due to heat produced within the city area and the thermal properties of the surface materials. Urban construction materials absorb and store solar heat whereas evaporative cooling is repressed in lack of water. The stored heat can then be re-emitted to the air, conserving high temperatures during night-time. This phenomenon is often referred to as the urban heat island effect (Akbari *et al*. 1992; Parlow, Chapter 1.2).

1.4.3.3 Infiltration and soil water transport

Soil properties and physical processes

The process of infiltration can be considered as a rail switch which controls whether rainfall is transformed into a water flux into the soil or along the earth's surface. Accordingly, the infiltration capacity of a surface and the underlying soil layers govern important components of the urban water cycle, such as surface runoff, groundwater recharge, and evapotranspiration. In natural watersheds, most precipitation infiltrates into the soil and subsequently evaporates to the atmosphere or moves within the soil matrix. Here, the processes of infiltration and soil water movement are closely interrelated. On the one hand, infiltration is restricted by the water transport capacity of the subsurface. On the other hand, infiltration to the subsurface determines the water movement within the subsurface.

The infiltration and movement of water through a particular substrate are strongly affected by its hydraulic properties, particularly hydraulic conductivity and the water retention characteristic. The hydraulic conductivity of a soil describes its capacity to drain water through the porous matrix and can vary depending on the water content. The highest values are reached for a saturated soil. In an unsaturated soil, parts of the pores within the soil structure are filled by air and do not participate in the water transport process, resulting in a reduction of the overall water transport capacity. The hydraulic conductivity is closely related to soil properties such as pore-size distribution, porosity, and pore continuity, but also on physical properties of the water such as density and viscosity. Fine soils comprising large percentages of silt or clay typically provide much lower hydraulic conductivities than coarse soils containing large percentages of sand or gravel.

The water retention characteristic of a soil describes the relationship between volumetric water content and capillary forces of the soil matrix. Capillary suction, also called capillary pressure or matric potential, is affected by hydrostatic forces that reduce the drainage capacity of the pores. The lower the water content of the soil matrix, the stronger the soil water is adsorbed to the solid phase (Fig. 1.4.4). Therefore a soil close to saturation has a higher capacity to drain, whereas some soil water will not drain under natural conditions (so-called residual water content).

The water retention characteristic is determined by several physical soil properties and therefore will also vary with soil texture. For the same volumetric water content, the matric potential head of a clay is much higher compared to silt, sand, or coarse road construction material. For this reason a sandy soil with a particular water content (e.g. 20 per cent) feels wet to touch, whereas a silt only feels slightly humid and a clay even feels dry. These highly nonlinear relationships between water content and matric potential can be described mathematically by a number of analytical functions. The most popular equations may be those developed by van Genuchten (1980) and by Brooks and Corey (1966).

The movement of water through a porous medium is driven by variations in the water potential resulting from evaporation, precipitation, and other atmospheric influences. This leads to an imbalance in the water potential over a defined

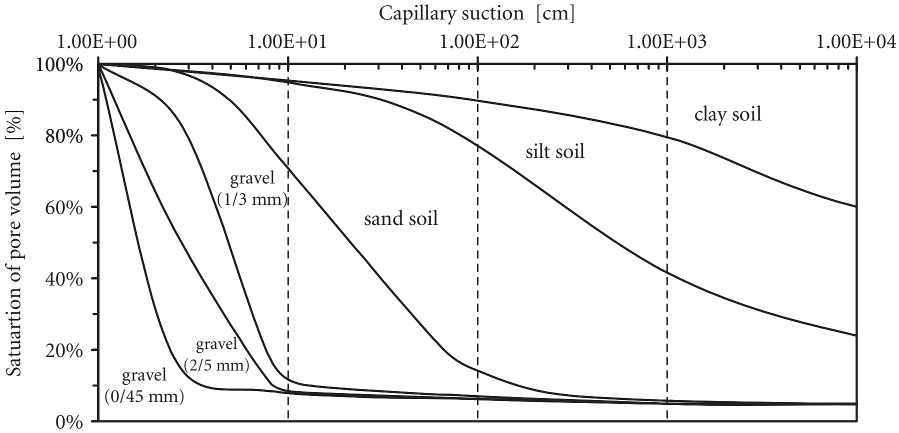

Figure 1.4.4 Exemplary water retention curves of natural soils and urban construction materials (Illgen 2010)

space and initiates water movement from regions of higher to lower potentials. The flow velocity primarily depends upon the potential gradient, the hydraulic conductivity, and the water content. The simplest mathematical description of the flow velocity in saturated porous media was developed by Darcy, a French engineer, in 1856. Darcy's flow equation, also called Darcy's Law, is a cornerstone of soil hydrology and is still widely applied today. For one-dimensional water flow, Darcy's Law may be expressed in differential form as:

$$q = \kappa - \frac{\delta\psi}{\delta z}$$

where q is the steady flow rate of water passing through the soil per unit cross-sectional area (m s^{-1}), K is the (saturated) hydraulic conductivity (m s^{-1}), Ψ is the pressure head (m), and z is the distance in direction of flow (m).

The transport of water into a soil usually occurs under unsaturated conditions. As the hydraulic conductivity is a function of volumetric water content, the water transport into and through an unsaturated soil is an unsteady process. Buckingham (1907) adapted Darcy's equation on unsaturated flow conditions. Richards (1931) finally combined the modified Darcy equation with the law of mass conservation (also called the continuity equation) to a non-linear partial differential equation that describes three-dimensional water transport in an unsaturated soil. The Richards' equation is the basis of many numerical models for detailed simulation of water movement in unsaturated and partly saturated porous media. It is the governing equation adopted when there is a requirement to accurately simulate infiltration, whereas Darcy's equation is mainly applied when calculating groundwater flow through a saturated media. The sophisticated Richard's equation is, however, rarely applied when modelling large-scale natural or urban watersheds, as a large number of soil parameters and substantial computational effort are required.

Commonly, hydrological models are developed to evaluate stormwater runoff and water transport processes on the surface, in streams, and in drainage systems. They use simplified mathematical descriptions to simulate the infiltration processes and to compute the resulting surface runoff. Some models are empirical, others are physically based. Some of the most widely applied are the approaches of Horton, Holtan, Green, and Ampt, and the US SCS runoff curve number (CN) method (Akan 1993).

Infiltration in urban areas

Large parts of urban environments are covered by impervious materials. Therefore the water balance of urban catchments significantly differs from natural watersheds or those used for agriculture or silviculture. However, a considerable portion of urban areas do encompass permeable paved surfaces and unpaved or vegetated surfaces (Pauleit and Breuste, Chapter 1.1). Pavements—whether permeable or impervious—are one of the most ubiquitous structures built by mankind and can be found in almost all urban areas. In the US they occupy twice the area of buildings (Ferguson 2003). Due to the extensive coverage of pavements in our cities, it is not surprising that they exhibit substantial influence over the infiltration of large volumes of water. Hence, cobblestone pavements and other more or less permeable land surfaces significantly govern the hydrology of urban environments.

As impervious surfaces such as rooftops or asphalt paved surfaces inhibit infiltration, fallen rain will either run off under the influence of gravity or will be stored on the surface as depressions are filled. In contrast, unpaved or vegetated areas, such as gardens, lawns or parks, commonly provide high infiltration capacities. Their hydrological performance is similar to that of natural soils with comparable vegetation cover and can be described by the infiltration characteristics and equations explained above.

The hydrology of semi-permeable surfaces such as pavements, porous asphalt, or green roofs is more complex. Pavements may be impervious if cement is used for bedding or grouting. However, in the majority of cases, pavements are at least moderately permeable. Depending on the type of pavement and the particular rain intensity they allow some infiltration but may also contribute to considerable surface runoff at the same time. In the past the infiltration capacity of pavements was rarely considered and conventional block or flag pavements were seen as being impervious. Today a huge variety of structures with varying permeability are available and widely used on private and public properties. Consequently, the specific infiltration performance of pavements has gained in importance and has been subject of intensive research work in recent years (e.g. Pratt *et al*. 1995; James 2000; Brattebo & Booth 2003; Bean *et al*. 2007; Illgen 2010; Illgen *et al*. 2007). Since they allow both preservation of the natural water cycle, as well as usability for traffic and modern living demands, permeable pavements have become a cornerstone in regard to sustainable urban development.

Pavements allow water to percolate through joints or voids into the subgrade layers and the subsurface beneath. The joints are commonly filled with coarse aggregates such as sand or crushed stone with relatively high hydraulic conductivities. The pavers are usually lying on a thin bedding layer which overlies a thicker layer of sub-base. Bedding and sub-base layers are commonly constructed of coarse and compacted mineral aggregates, mainly of gravel or crushed stone. Due to the physical properties of the coarse grained construction material used, the technical structure and hydrological performance of a permeable pavement will significantly vary from natural soils (Fig. 1.4.4).

The infiltration capacity of a pavement is strongly affected by a multitude of site-specific constraints. Accordingly, infiltration rates can vary significantly from one site to another or even on a site-scale, irrespective of the particular type of pavement. Commonly the infiltration capacity of a pavement will reduce significantly within a number of years of its installation (Borgwardt 2006). This occurs when the hydraulic conductivity of the pavement construction is reduced as a result of fine material accumulating into the slots or voids, the so-called clogging or colmation effect. The infiltration capacity may further be reduced over a longer timeframe by continuous mechanical compaction of the aggregates by vehicles and other heavy machinery.

Beside clogging and mechanical impacts, the structural condition of a pavement, such as opening ratio, surface slope, and grain size distribution of the joint aggregates, has a major effect on the infiltration capacity of a pavement. Altogether, the infiltration performance of a particular pavement on a certain site is controlled by many site-specific boundary conditions and offers notably stochastic attributes. Figure 1.4.5 shows cumulative frequency curves of the infiltration rate for common types of pavement (Illgen 2010).

This figure illustrates the enormous variability of the infiltration capacity and reflects its distinctive stochastic character. In a logarithmic representation, the range of value between the 10th and 90th percentiles of the infiltration rate covers more than two decades for almost any type of pavement. Median infiltration rates amount to between 55 l/(s ha) or 22 mm/hr for conventional block and interlocking pavement with small sand-filled joints, and 750 l/(s ha) or 270 mm/hr for more pervious types of pavement. For grassed grid pavers a median of 200 l/

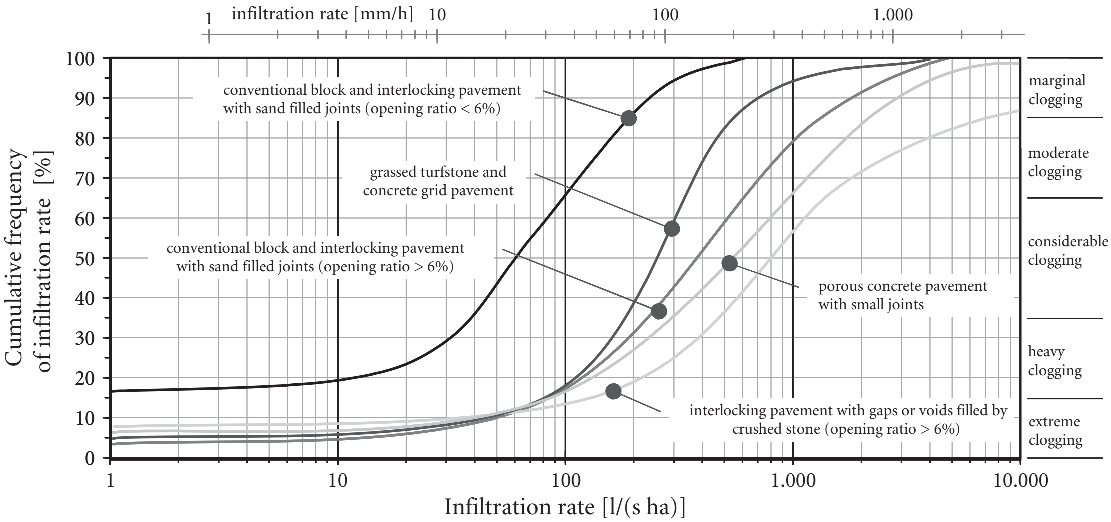

Figure 1.4.5 Cumulative frequency of infiltration capacities of common pavement types

(s ha) or 72 mm/hr can be stated. Moreover, on 5–15 per cent of all pavements or of a particular pavement area, respectively, clogging is at such an extreme that infiltration is almost completely inhibited.

Compared to clogging effects, the surface slope has a much lesser effect on the infiltration capacity. An increase of surface slope from 2.5 to 5.0 per cent, for instance, entails a decline of the infiltration rate of 5–20 per cent. Depending on the type of pavement and the particular grade of clogging, the rain intensity may be more crucial as well. Commonly, the infiltration capacity of a pavement rises considerably with increasing rain intensity as a result of the inhomogeneities of the hydraulic conductivity of the joint aggregates (Illgen *et al*. 2008). The initial soil water content within the joints may also influence the runoff and infiltration performance of the pavement construction, but on a much lower level than for natural soils. Relatively high soil water contents resulting from antecedent rain events repress the higher infiltration capacities commonly observed at the beginning of a rain event. Under dry conditions a pavement offers extremely high infiltration rates within the first few minutes of a storm due to the filling of wide pores in the mineral aggregates in the joints. These initial infiltration rates largely exceed the hydraulic conductivity of the mineral aggregates. When the pores have filled with water, infiltration settles down to much lower and almost constant values.

In contrast to previous expectations, the hydraulic conductivity of the underlying soil subgrade has only a minor impact on the infiltration performance of urban surfaces. The water contents within the pavement structure only rise moderately during a storm event due to the much higher pore volume of the coarse base layer material. Even a subgrade of extremely low hydraulic conductivity does not generally induce a full saturation of the overlying base layer due to its enormous retention volume, generally far above 50 mm.

All in all, the infiltration performance of the entire pavement structure is predominantly determined by the infiltration capacity of the topping layer, which is mainly dependent on the opening ratio of the joints, the aggregates used therefore, the surface slope, and in particular the extend of clogging. Sandy aggregates and small joints provide a much lower infiltration rate than aggregates of crushed gravel and wider joints. Furthermore, they allow a faster and more distinct clogging by fine particles. The extent of clogging does vary widely and randomly and causes the enormous variability of the infiltration capacity illustrated in Fig. 1.4.5.

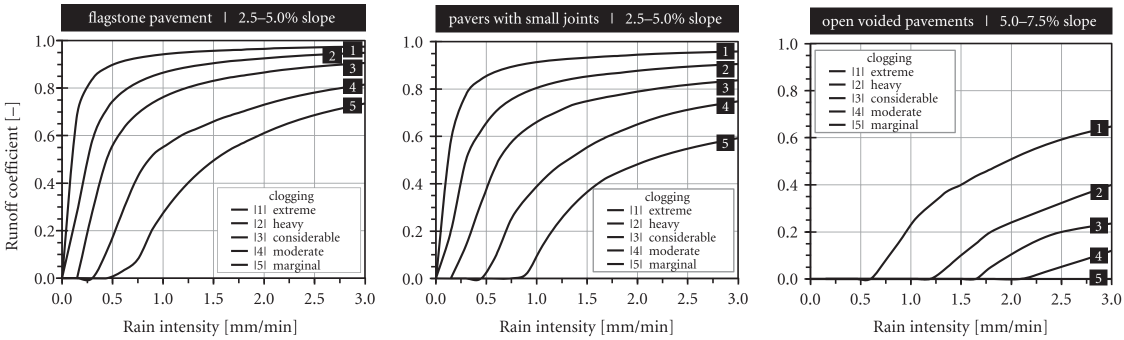

Figure 1.4.6 Runoff coefficients of pavements related to rain intensity and degree of clogging

Even though the specific infiltration performance of semi-permeable urban surfaces significantly differs from subnatural soils, common models of urban hydrology are still mainly applying similar mathematical approaches as for natural or subnatural soils, and simply adapting infiltration parameter values (James *et al*. 2002; Ferguson 2003). Consequently, it is not surprising that simulation results often exhibit large differences compared to observed runoff volume or peak discharge. Advanced approaches have been developed recently by Illgen (2008) and by Kipkie and James (2000).

1.4.3.4 Stormwater runoff

Surface runoff occurs when the present rain intensity exceeds the retention capacity of a surface. Even on a completely impervious surface some water is stored on the surface and represents a so-called rainfall loss. Rainfall losses mainly result from interception, depression storage, and infiltration. Evaporation and transpiration during a rain event play minor roles and are commonly neglected in event-based considerations.

Interception describes the wetting process of a surface due to precipitation or irrigation. In urban hydrology, interception covers that proportion of rainfall intercepted by aerial parts of plants and by urban surfaces such as rooftops, streets, or yards. Intercepted rainfall is retained from runoff and subsequently evaporates. The interception loss depends on the particular surface characteristic. Roughness and disposable surface are important factors that determine the retention capacity through interception. Urban surfaces commonly offer interception losses between 0.1 and 1.0 mm of rainfall depth. Plants or vegetated areas provide in general much higher interception losses than urban surfaces due to the larger specific surface area of leaves, branches, and flowers.

Additional volumes of fallen rain are retained in an infinite number of depressions in the ground surface. These depressions may range in size from lakes, through puddles, down to microscopic asperities. From there the water either evaporates directly into the atmosphere or infiltrates into the subsurface (Pilgrim & Cordery 1992). Even though the contained water may be exchanged during a rain event, the captured volume represents another rainfall loss to surface runoff. The depression storage capacity is mainly determined by planarity, roughness, and slope of the particular surface and commonly varies in a range of 0.1–6.0 mm of rainfall depth. Since interception and depression storage mainly occur at the beginning of a rain event, both are often lumped together as an initial loss, whereas infiltration represents a continuous loss.

The surface runoff volume, also called effective rainfall, finally results as the difference between the precipitation depth and the sum of hydrological losses. The sum of losses can be represented by a runoff coefficient, the ratio of surface runoff to rainfall that may range from 0 to 1.0. The runoff coefficient depends upon both characteristics of the surface, such as infiltration capacity or topography, as well as characteristics of the rainfall, such as intensity and duration. For an intensive rainfall on an impervious surface like an asphalt street, for instance, the runoff coefficient may range close to 1.0. For the same asphalt surface and a rainfall of much lower rain depth of 1–4 mm a much lower runoff coefficient of more than 0.5 may result due to interception and depression storage that, in this case, may cover a high percentage of rainfall.

The temporal distribution of the surface runoff (hydrograph) follows the temporal distribution of the rainfall (hyetograph) superposed with the temporal distribution of the losses. Due to initial losses, runoff starts with a lag time which depends upon the rain intensity and the retention capacity of the surface. In addition, surface runoff undergoes retention and translation processes along its flow path on the surface that effect a reduction of the peak flow and a stretching of the runoff curve. For a given rainfall, stormwater runoff can be estimated by multiplication of rain depth or rain intensity, size of the contributing area, and the according runoff coefficient. In addition to this simple, but quite popular, approach (called the Rational Method), a lot of related or more complex approaches have been developed over the years and are adopted for hydrological evaluations of urban watersheds (Akan 1993).

1.4.4 Water balance characteristics of urban areas

The numerical water balance of an urban watershed is closely related to the considered temporal frame. The division of a rainfall volume into surface runoff and infiltration depends upon rainfall depth, rain intensity, and rain duration. These rainfall characteristics vary from one rain event to another and cause related variations between water balances of different timescales that cover different rain events. Long-term water balances comprise a sum of rainfall events and have to account for the processes occurring during dry periods. Since rainfall events with comparatively low intensities of less than 0.2 mm/min provide more than 75 per cent of the annual rainfall depth in many climate regions,

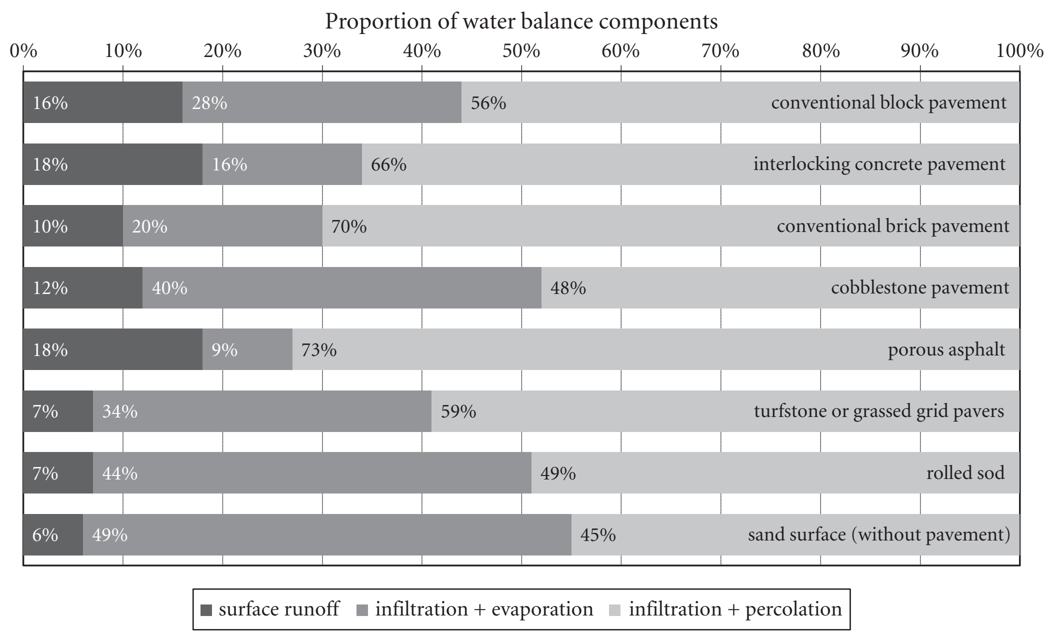

Figure 1.4.7 Annual water balances of common urban surface materials (BWW 1984)

annual water balances differ markedly from water balances of a single storm event. In addition, evapotranspiration is a large component in the long-term water balance due to continuous evaporation and transpiration of previously intercepted, stored, or infiltrated rainwater, whereas event-based considerations may neglect evapotranspiration during an event.

With respect to single storm events with return periods of 0.5–10 years that may provide rain intensities of up to 3.0 mm/min, comparatively high runoff coefficients are to be expected for impervious as well as for permeably paved urban surfaces. Examples of runoff coefficients relating to minor and more permeable surface materials are illustrated in Fig. 1.4.6 (Illgen 2010). The single nomograms reflect the proportion of generated surface runoff and stormwater retention depending upon the rain intensity and degree of clogging. They therefore correlate with the cumulative frequencies shown in Fig. 1.4.5 and express the high local variability of the retention capacity of urban surfaces.

In contrast to single storm events, a large proportion of fallen rain is retained from surface runoff when a long-term period is considered. Even on a completely sealed asphalt surface more than 40 per cent of annual rain is commonly stored on the surface and evaporates. Pavements that allow at least some infiltration reduce the runoff ratio even further. Commonly, more than 75 per cent of annual rainfall does infiltrate and recharge groundwater or evaporate into the atmosphere, whereas less than 20 per cent actually generate surface runoff. Major components of the urban water cycle for various pervious surfaces, obtained from measurements in Berlin, are shown in Fig. 1.4.7 (based on BWW 1984).

This figure indicates that on common pavements more than 45 per cent of annual rainfall infiltrates and percolates to deeper horizons of the subsurface. Evapotranspiration only accounts for 10–40 per cent of the annual rainfall volume. Compared to sub-natural soils, such as rolled sod or a mere sand surface, conventional or exceptionally permeable pavements may thus allow more water to percolate and to recharge groundwater, whereas evaporation may be accordingly reduced.

The water balance of an entire urban catchment or a particular city district finally depends on the proportion of the different surface materials and the extent of surface sealing (Pauleit and Breuste, Chapter 1.1). It can be approximately estimated by the summation of the water balance components of individual surface types as described above, thereby taking into account local climate conditions and the considered temporal frame. An ambitious goal of urban planning and development is to accomplish near-natural water balances in our cities in the future. In this regard a large-scale arrangement of greenspaces, together with permeably constructed pavements, significantly improves the hydrological conditions of urban environments and supports a sustainable urban development.

Summary

Jürgen H. Breuste

The chapters of Section 1 give not only an introduction into the basic physical elements of the urban ecosystem, climate, water, and soil, as changed by humans, but also show how these physical elements are connected to the idea of land-use as a steering factor for ecologically relevant change. It results in visible changes on the Earth's surface and changes in the land's physical condition through land-use processes. These new physical conditions are responsible for new conditions of life, including human life, in cities. We recognize that we are still unable to analyse all ecologically relevant aspects of the complex matter of urban land-use. Land-use includes several aspects related to urban ecosystems, caused by a huge number of different human activities that differ in frequency, intensity, special extension and so on.

Urban areas are tremendously modified environments. Human impacts on environmental quality and ecosystem processes within cities are of particular concern for urban planners. In these chapters it has been argued that a deeper understanding of the relationships between land-use and urban form and the ecosystem process is important. Land-use and land cover are crucial in this respect because they have a fundamental influence on urban ecosystem processes and they can be influenced by urban planning (Pauleit and Breuste, Chapter 1.1).

There is interaction between urban grey and urban green. The urban matrix or urban form consisting of different types of built structures (residential, commercial, and others) is responsible for the physical and biotic functionality of urban ecosystems. This matrix is land-use based. It contains 'the urban green' (all forms of urban vegetation cover in mostly all land-use types) and the 'urban grey' (all forms of paved surfaces and buildings). The quality and proportion of both parts of the matrix influence the climatic, hydrological, and soil-related functionality of urban ecosystems. Adequate planning of land-use can influence this balance of urban grey and urban green and steer ecological functionality.

While many studies have investigated the urban green, chapters in this section show that it is urgent matter to adequately investigate and evaluate the 'urban grey' in its ecological relevance and physical functionality. There seems to be a lack of basic and applied research on this issue. Paved and built surfaces are strongly influencing urban climate, urban heat island effects, and urban hydrology. They are partly responsible for increased flooding risks, unhealthy urban climate, and disturbed urban biodiversity. Planning of urban grey does not exist or is fragmentary at best. These chapters show that adequate planning is urgently needed.

The stormwater retention capacities of paved and unpaved surfaces, which ultimately regulate the surface runoff process, can be specified from an urban water management perspective. Characteristic values of the particular water balance components can be outlined for several types of surfaces, reflecting the interactions between the atmospheric impact by rainfall and the hydrological phenomena on urban areas. Urban land-use creates land surfaces with new physical and biological land conditions that are possible to typify. The paved, impervious or sealed surfaces especially, the urban 'grey', are the key elements of physical change of the urban environment and they are based on types and locations of land-use activities.

The ecological importance of urban soils has been underestimated for a long time. Urban soils were seen as nothing more than an easy way to replace a physical element without having their own value. Although the first urban soil surveys were made in the early 1980s, there is still no concept that adequately integrates urban soils into the urban ecosystem.

In addition to water and air, soil is the third environmental medium, which closely interacts with the other ecological components in cities. This interrelationship depends on the degree of anthropogenic modifications or interference. Urban soil plays a key role in the urban geo-ecosystem because it functions as a sink as well as a source and has the capacity to transform substances (see Sauerwein, Chapter 1.3.).

During the last decade urban authorities have become more aware of the lack of ecological functionality in urban areas. This has led to a strong increase in research and application in selected fields such as urban climatology to meet the requirements of public authorities and political decision-makers. Worldwide, urban climate has become an important issue and it will remain on the agenda of politics and research for a long time. Today, more than half of the world's population lives in an urban environment which modifies the local climatic conditions compared to rural surroundings. The expected climate change will have tremendous influence, especially in cities (on health, vegetation etc.), and ecological planning is needed to mitigate these effects. The accelerating growth of mega-cities, especially in Third World countries, is a challenge for urban climate research and applications as these cities are major contributors to air pollution and emissions of greenhouse gases. Urban agglomerations modify the present global climate conditions. The global warming expected in the next decades will again influence the climate of cities significantly with respect to air temperatures, heat stress etc. (see Parlow, Chapter 1.2.).

Conceptual models (e. g. Grimm *et al*. 2000) may indicate how these approaches can be embedded into a comprehensive analysis of the pattern, process, and dynamics of urban ecosystems, integrating human and natural forces. Modelling approaches combined with methods of sustainability impact analysis that allow us to assess the likely consequences of different planning scenarios on ecosystem services are particularly suitable. Such integrative research is urgently needed to underpin policies for urban sustainability and, indeed, for global sustainability (see Pauleit and Breuste, Chapter 1.1.).

SECTION 2

Ecology in Cities: Patterns of Urban Biodiversity

SECTION EDITOR: **Glenn R. Guntenspergen**

Introduction

Glenn R. Guntenspergen

Ecology in cities refers to the patterns of species occurrence, distribution of habitats, and biodiversity. The literature is replete with numerous floristic and faunal surveys in urban ecosystems worldwide. But making sense of the patterns of biodiversity found in this broad literature has proved problematic. What is now required is a broader view that includes an understanding of the role that interactions between ecological and social systems play in determining the patterns of biodiversity found in urban settings. The five chapters in this section build on the insights of the physical context of Section 1 and attempt to draw a series of generalizations about the adaptability of species to the abiotic conditions and built structures in urban environments, and the impact of human subsidies. Three of the chapters investigate unique habitat types to develop insights about plant biodiversity patterns and drivers. The last two overview chapters explore these common themes of the earlier chapters and provide larger scale insights about patterns of plant and animal biodiversity in urban environments.

Andrew Baldwin uses wetland habitats and a worldwide comparative approach as a framework to describe the impacts of urbanization on the composition and diversity of plant communities. He emphasizes the direct impact of the built urban environment on these plant communities resulting in a reduction in habitat area, and how changes in abiotic and biotic processes affect biological diversity. He points out that urbanization creates different environmental settings that alter the fundamental controls affecting vegetation structure. Despite these impacts, urban wetlands still have considerable impact on ecosystem services for urban dwellers.

The chapter by Martin Quigley examines the impact of human decision-making on the plant biodiversity of designed landscapes by emphasizing the altered abiotic and biotic characteristics associated with these habitats. These habitats are often designed for aesthetic and visual purposes that have a profound effect on their composition and structure. Although high plant biodiversity is often found in these habitats, this is only possible with considerable human subsidy and so self-sustainable communities are not maintainable. Often, these habitats lack the ecosystem services of natural communities.

Jeremy Lundholm investigates the unique plant communities that occur in urban environments by focusing on the spontaneous flora of built surfaces. Lundholm uses a similar comparative approach to argue for the convergence of a similar hard surface flora worldwide. Many of the typical species found in these environments are native species with ecological requirements similar to those found in these urban habitats. High biodiversity is found in these habitats because of the environmental conditions which facilitate colonization and establishment by both plant specialists and generalists. Hard surfaces at the same time present selective pressures that also facilitate the evolution of species unique to anthropogenic environments. Despite their often unique nature, this chapter concludes by emphasizing the ecological services that these habitats can provide.

This section concludes with two general chapters that focus on broader scale investigations of urban vegetation and wildlife. Christopher Dunn and Liam Heneghan provide an overview of urban vegetation that incorporates many of the same themes and habitat types (e.g. remnant vs. designed) that were explored in the earlier chapters. They provide a broad review of urban floristics and more complex topics that determine the biodiversity and change in urban vegetation. The themes they explore, habitat transformation, habitat fragmentation, environmental conditions, and human design, bring together the insights developed in the proceeding chapters. Clark Adams and Kieran Lindsey's chapter continues this investigation of the themes explored in the urban vegetation chapters to characterize the biodiversity of urban wildlife populations and the role of humans on their abundance and structure. Their review of the literature is used to identify common species in different cities, explain how they have adapted to urban ecosystems, and discuss human–wildlife conflicts. They emphasize the similarities of urban wildlife vertebrate assemblages worldwide because species are either preadapted to urban conditions or are generalists that can adapt and survive in the urban ecosystem. Another theme they explore is the role humans can play subsidizing these populations by providing supplemental food and shelter. They also examine the role of the built environment as both a conflict and subsidy for vertebrate wildlife populations. They conclude by deciding that humans represent a keystone species and play a fundamental role in maintaining biodiversity in urban systems.

These chapters emphasize that environmental variables are not the only drivers responsible for the biodiversity of urban environments. The built environment and the role of humans play an equally important role in defining the biological character of the city. This idea is a theme that is echoed repeatedly in the subsequent sections of this book.

CHAPTER 2.1

Plant Communities of Urban Wetlands: Patterns and Controlling Processes

Andrew H. Baldwin

2.1.1 Introduction

Throughout human history, humans have lived near or in wetlands. The high productivity of wetlands provided early human settlers with an abundance of food and materials for construction and daily life, and in many parts of the world people continue to rely on wetlands for subsistence (Mitsch & Gosselink 2007). Their flat topography and nutrient-rich soils made them excellent locations for building and agriculture, although drainage was usually necessary. Furthermore, wetlands are often located near navigable waterways, making them an important location for ports and industries that require large water supplies (Pinder & Witherick 1990). For these reasons, it is no wonder that many urban centres developed and displaced wetlands associated with rivers, lakes, and coastlines (Fig. 2.1.1).

The development of urban areas, however, came with a cost to wetlands, resulting in direct impacts such as filling, drainage, and excavation, as well as indirect impacts caused by, for example, changes in hydrology, releases of pollutants, and introductions of non-native plants and animals (Ehrenfeld 2000a; Wang *et al*. 2007). Inevitably, these impacts had consequences for humans, who had benefited from ecosystem services such as floodwater storage, water quality improvement, and the supply of food and materials from the biological diversity of wetlands. Today, wetlands remaining in urban areas are important in disproportion to their size due to the high density of human populations who benefit from their services and the scarcity of such habitat for wildlife (Mitsch & Gosselink 2007).

While hydrology is the most important variable underlying the development and functioning of wetlands, vegetation is often the most visually obvious character. The structure of vegetation often gives a particular type of wetland its name, for example heath bog, cattail marsh, reedswamp, or woody swamp. The species composition and diversity of vegetation that develops in a particular wetland integrates the geological, climatic, hydrological, and biogeochemical processes of that wetland. In turn, the plant community directly affects habitat for invertebrates, fish, and wildlife, and plays a central role in nutrient cycling, organic matter accretion, and support of detrital-based food webs in and outside of the wetland. Changes in hydrology, nutrient loading, non-native species pools, or other processes altered by urbanization will change the biological diversity of wetland plant communities by altering the relative abundance of species and increasing or decreasing the number and composition of species. Because of the critical role of the plant community in ecosystem processes, understanding how urbanization affects wetland plant communities can inform environmental managers and policy makers on how best to plan development projects, retrofit stormwater management systems, manage non-wetland vegetation, and restore degraded wetlands.

The objectives of this chapter are two-fold. The first is to describe the characteristics of vegetation in urban wetlands, with examples from different parts of the world. The second objective is to summarize the direct and indirect impacts of urbanization on the species composition and diversity of wetland

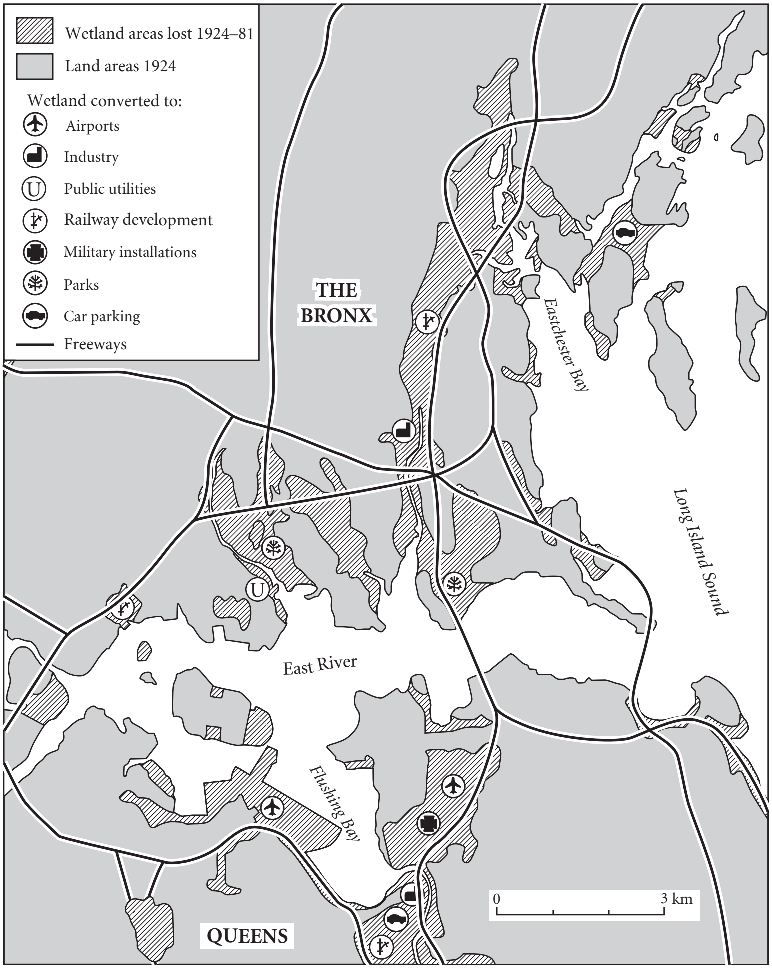

Figure 2.1.1 Wetland loss due to urbanization in Queens and the Bronx boroughs, New York City, 1924–1981. From Pinder and Witherick (1990). Used with permission

vegetation, again supported by examples from around the globe. After addressing these objectives a synthesis of important themes and a prospective view of urbanization and wetland vegetation are presented.

A note on terminology: for simplicity throughout this chapter I use the term 'plant biodiversity' as an umbrella term encompassing the composition, number, and relative abundance of plant species, as well as the structural (e.g. woody, herbaceous) and spatial (e.g. zonation, patchiness, isolation) dimensions of variability in plant communities across individual wetlands and landscapes.

2.1.2 Wetland plant biodiversity in urban areas

Plant communities of urban wetlands are often characterized by disturbance-adapted species that are weedy or invasive, and in many cases non-native (Ehrenfeld 2000a; Choi & Bury 2003). Nonetheless, these habitats may be biodiversity hotspots in urban areas. In a survey of 996 sites in the Hauts-de-Seine district bordering Paris, France, aquatic habitats had the highest 'Index of Floristic Interest' (among 23 habitat types), despite low species richness (number of species), because they sup-

Figure 2.1.2 The last naturally occurring tidal freshwater marsh in Washington, DC, located adjacent to Dueling Creek, a tributary of the Anacostia River. While the plant community of this wetland contains many species typical of tidal freshwater marshes in non-urban watersheds, invasive plants such as *Phalaris arundinacea*, *Phragmites austalis*, *Lythrum salicaria*, and *Typha* spp. are also common. Photograph by A.H. Baldwin

ported rare, native, and common species of high conservation interest (Muratet *et al*. 2008). Similarly, in the Pochefstroom municipal area in South Africa, wetlands are diverse in both species and community type, due in large part to anthropogenic disturbance over many years (Cilliers *et al*. 1998). In Washington, DC, remaining naturally occurring tidal freshwater marshes have similar plant species density (species richness within study plots) as relatively undisturbed marshes in non-urban settings (Baldwin 2004).

While the diversity of species and communities may be high in urban wetlands, much of the species richness can be attributed to non-native and invasive native plants. A survey of 45 naturally occurring wetlands and 51 mitigation (restored or created) wetlands in the rapidly urbanizing city of Portland, Oregon, reported a total of 365 species, but more than half of these in both types of wetlands were non-native species (Magee *et al*. 1999). The most common species were the non-native genotype of *Phalaris arundinacea* and the grasses *Holcus lanatus* and *Agrostis gigantea*. Other common non-native species in urban wetlands of the US include *Lythrum salicaria*, *Typha angustifolia*, and non-native genotypes of *Phragmites australis* (Fig. 2.1.2) (Baldwin & DeRico 1999; Panno *et al*. 1999; Choi & Bury 2003). Some native species that are adapted to rapid seed dispersal and have tall growth forms also may become dominant in urban wetlands, particularly in restored or created sites, one of the most common in marshes being *Typha latifolia* in the eastern US (Baldwin pers. obs.). Non-native species may also occur in seed banks and be dispersed via water in urban wetlands (Neff & Baldwin 2005; Neff *et al*. 2008). Forested wetlands in urban areas of the US east coast are also colonized by non-native species, some of the most common being the grass *Microstegium vimineum*, the shrubs *Lonicera tatarica* and *Rosa multiflora*, the vines *Lonicera japonica* and *Celastrus orbiculatus*, and the trees *Acer platanoides* and *Paulownia tomentosa* (Ehrenfeld 2008). In Beijing, China, a total of 47 non-native species are considered invasive, including at least two wetland-adapted species, *Alternanthera philoxeroides* and *Ambrosia trifida* (Wang, *et al*. 2007; USDA Plants Database, plants.usda.gov, accessed 5 October 2009). The grass *Cynodon dactylon* is considered invasive in urban wetlands in South Africa (Cilliers *et al*. 1998).

In addition to plant biodiversity at the scale of individual wetland areas (α or alpha diversity), the pattern of diversity of community types across the landscape (γ or gamma diversity) may also differ between urban and non-urban wetlands. These differences may arise due to the fragmentation of individual wetlands and the isolation of wetlands associated with development of the urban environment. It might be expected that fragmentation would decrease diversity of community types across the landscape, but some evidence suggests that anthropogenic disturbances are sufficiently varied to maintain heterogeneity of wetland types in urban settings (Cilliers *et al*. 1998). Changes in biodiversity due to fragmentation and isolation are considered in more detail in the next section, along with other influences of urbanization on plant biodiversity in wetlands. Some of these effects of urbanization on wetland ecosystems and their causal mechanisms are summarized in Table 2.1.1.

2.1.3 Effects of urbanization on wetland vegetation

2.1.3.1 Direct impacts

Direct effects on wetland vegetation are quantitative changes in the extent of wetlands caused by eliminating wetlands through dredging, filling, or draining. In some cases these changes may also increase the extent of wetlands, either unintentionally by altering hydrology, or intentionally by restoring and creating wetlands or construction of stormwater ponds or other aquatic features. While the vegetation of individual wetlands is lost (or created) by direct impacts, wetland plant biodiversity continues to exist on a landscape scale, which may be altered directly by isolation and fragmentation of remaining wetlands. Below, changes in extent of wetlands due to urbanization are summarized briefly, followed by a review of how isolation and fragmentation affect wetland plant biodiversity.

Losses of wetlands due to urbanization

Urbanization has caused losses of wetlands around the world. In the US, wetlands were destroyed during development of sections of Washington, DC, New York, Seattle, Boston, Dallas, Miami, Philadelphia, Juneau, New Orleans, and Chicago (and their suburbs) (Schmid 1994). During 1954 to 1974 more than 3.6 million ha of wetlands in USA were lost due to urban development with losses decreasing to about 39,000 ha total for the decade of the mid-1970s through the mid-1980s (Dahl & Johnson 1991; Schmid 1994). Along the coast of Long Island Sound, dredging, filling, and draining reduced tidal marshes along the New York (southern) side of the Sound from 4,900 to 2,600 ha, and along the Connecticut (northern) shoreline destroyed about 45 per cent of 95 km^2 of salt marshes between 1914 and 1959 (Niering 1970). In Europe, the port of Rotterdam was expanded to tens of square kilometres in size to accommodate large ships and refineries beginning after the First World War, which included damming of tidal rivers, dredging shallows, reclaiming tidal mudflats, and reshaping of islands, all of which eliminated wetlands; expansion of ports in Marseilles and LeHavre, France, had similar consequences (Pinder & Witherick 1990). In another example, between 1984 and 2001 the areal extent of urbanized land 'infrastructure' increased by 48 per cent, 99 per cent, and 180 per cent, respectively in three wetland complexes in the Mar Menor lagoon on the southeast coast of Spain on the Mediterranean Sea (Esteve *et al*. 2008). These increases destroyed 4.3 per cent, 3.3 per cent, and 18.5 per cent of each wetland complex (the total area of urban infrastructure increased from 106 to 178 ha out of an area of 1,716 ha).

Urbanization has also eliminated wetlands in Asia. In Singapore, more than 50 km^2 of land has been reclaimed from the sea for development, mostly from the intertidal zone, including mangroves and tidal flats (Pinder & Witherick 1990). China has seen huge recent growth in urbanization: the number of cities increased from 190 to 770 between 1980 and 2003, with urban populations increasing from 18 to 39 per cent of the population (Wang *et al*. 2007). In the early 1970s, there were about 80,000 ha of wetlands in Beijing's territory, which decreased to 27,300 ha by 2004 due to urbanization (Wang, *et al*. 2007).

Table 2.1.1 Common ecosystem characteristics of urban wetlands, relative to non-urban wetlands

Ecosystem aspect	Properties of urban wetlands	Causal factors
Hydrology	Higher stormflow peaks	Watershed imperviousness leading to more rapid runoff
	Lower base groundwater flows	Reduced recharge of regional aquifers due to watershed imperviousness, stream channel down-cutting and leaky sewer pipes lower water tables
	Increased freshwater flow to estuarine and coastal wetlands (decreased salinity)	More freshwater runoff from impervious areas; more irrigation of urban and agricultural lands
	Decreased stream–floodplain interaction	Flood control measures (e.g. levees), channelization
Geomorphology	Fewer gradual slopes	Filling-in of higher elevations of wetlands for development
	Straighter channels (less sinuosity)	Channelization, increased scouring due to higher stormflow peaks
	Stream channel erosion and down-cutting	Higher stormflow peaks
	Less microtopographic variation	Filling in lower elevations by excessive sediment
Water quality and materials transport	Higher loading of nutrients, salt, and toxins via surface and groundwater	Runoff from urbanized areas
	Higher sediment loading	Runoff from construction and other unvegetated sites, stream channel erosion due to high stormflows
	Lower dissolved oxygen	Inputs of organic matter from combined sewer overflows
Climate	Warmer air temperature	Heat island effect
	More air pollutants (oxidants, particulates)	Fossil fuel combustion, industrial and construction activities
	Lower net radiation	Reflection of incoming solar radiation by gases and particulates
Disturbance regimes	More hydrologic disturbance	Scouring and erosion due to high stormflows
	More sediment deposition (inhibits seed germination, buries seedlings)	More suspended sediment entering wetlands due to high stormflows
	Suppression of fires	In fire-frequented ecosystems near urban developments
	Increase in herbivory	Proliferation of urban-adapted wildlife species such as non-migratory Canada Goose
	More opportunities for colonization	More bare or unpopulated areas due to more frequent disturbance
	Lower resilience	Fewer native species, more opportunities for colonization by disturbance-adapted species
	More human physical disturbance	Trampling, rubbish from human activity
Spatial aspects	Wetlands smaller and farther apart; larger wetlands fragmented into smaller pieces	Construction of roads, buildings, pipelines, parking lots, and other infrastructure
Plant community structure and function	More non-native plant species	More frequent disturbance; more populations of non-natives nearby
	Weedy or invasive plant species	Species adapted for rapid propagule dispersal and fast growth favoured
	Lower native plant taxa richness	Smaller, more isolated wetlands may lead to local species extinctions
	Lower community type diversity	Reduction in geomorphological complexity
	Clonal, tall plants favoured; high primary productivity	High nutrient loadings competitively favour plants such as *Typha* spp., *Phragmites australis*, and *Phalaris arundinacea*
	Less seed dispersal between wetlands	Greater distance between wetlands

Changes in plant biodiversity due to isolation and fragmentation

The direct losses of wetlands due to draining, filling, or excavating of course eliminate the plant communities present and their biodiversity. However, these losses can affect the biodiversity of wetlands remaining in the urban landscape by isolating them from other wetlands or fragmenting large wetlands into smaller patches. Changes in plant biodiversity result from lower seed and propagule dispersal between wetlands, caused by lack of vegetated dispersal corridors and interruption of hydrologic connections between wetlands (and associated propagule dispersal).

Much of the available information on isolation and fragmentation of wetlands comes from studies of floristic quality indicators or metrics used to assess the condition of wetlands. These indices generally rely on a panel of experts who rank different plants species according to how often they are likely to be encountered at disturbed sites in a region (Andreas & Lichvar 1995; Lopez & Fennessy 2002). Essentially, by surveying the plant communities in wetlands with different surrounding land-uses the rankings are used to calculate an index that reflects the level of 'disturbance' indicated by the plant species present. Variables such as surrounding land-use, distance between wetlands, degree of hydrologic alteration, and water quality can then be correlated with the index values for each wetland. In a study of 20 depressional wetlands in Ohio, Lopez and Fennessy (2002) found that sites with lower floristic quality index values (i.e. more 'disturbance' species) were associated with more urban land-use and greater distances to other wetlands. Smaller and farther-apart wetlands in Illinois also tended to have lower floristic quality indices (Matthews *et al*. 2005) and a land development intensity index was similarly found to be related to lower floristic quality indices in Florida wetlands (Cohen *et al*. 2004; Reiss 2006). However, it is not clear if lower floristic quality indices are indicative of lower plant biodiversity, because most formulations evaluate only native species richness and the 'disturbance' ranking is zero for non-native species (Ervin *et al*. 2006; Miller & Wardrop 2006), which may or may not be favoured by urbanization. In fact, urbanization may result in higher numbers of species than comparable wetlands in non-urban areas due to the contribution of non-native species and greater disturbance-mediated coexistence of species (Ehrenfeld 2000a; Baldwin 2004). For example, Lopez *et al*. (2002) reported that the taxa richness of submergent plants increased with area of urban land cover in 31 Ohio depressional wetlands at the species, genus, and family taxonomic levels. Furthermore, the total number of all species (and genera and families) was not significantly related to urban land cover. Similarly, in forested wetlands in New Jersey, the total and mean number of species per site type were about twice as high in suburban sites as in control (non-urban) sites (Ehrenfeld & Schneider 1993).

Even though taxonomic richness may not be directly related to urban land cover, increasing distance between wetlands may be. Lopez *et al*. (2002) found that richness at the genus and family taxonomic levels were negatively related to inter-wetland distance. Similarly, indices of diversity were negatively correlated with increasing distance between wetlands for some plant guilds. Together with the richness measures, the results indicate that as the distance between wetlands increases there is decreasing heterogeneity in the distribution of plant species within individual wetland sites.

2.1.3.2 Indirect impacts

Indirect effects are qualitative changes in plant biodiversity resulting from urbanization, such as increases and decreases (or elimination) of individual species. Normally these indirect impacts are observed at the scale of individual wetlands and develop gradually due to changes in environmental conditions. Here the causes of wetland vegetation change due to urbanization are summarized, followed by examples of urbanization-mediated shifts in wetland plant biodiversity.

Causes of change in wetland vegetation

The vegetation of wetlands develops in response to a particular suite of hydrological, geomorphological, and climatic conditions, and interactions between biota and these processes (Mitsch & Gosselink 2007). Urbanization creates a physically and biologically different environment from non-urban settings (Ehrenfeld 2000a), which in turn alters the fundamental controls on vegetation structure and function in urban wetlands (Table 2.1.1). Watershed urbanization alters patterns of runoff and groundwater recharge, increasing stormwater flood peaks and drying wetlands. Stream channel down-cutting due to flashy hydroperiods that induce scouring and erosion and leaky sewer lines may both lower water tables in wetlands, further draining wetlands. Runoff from construction sites and high stormflows increase sediment transport into wetlands and more frequent floods may increase export of organic matter from floodplain wetlands. Some of these hydrological changes alter geomorphology through erosion or sedimentation, but alterations such as channelization of streams and levee construction may decrease stream–wetland connectivity or alter wetland hydroperiod.

Watershed urbanization may also increase nutrient loading rates (e.g. increased runoff from fertilized lawns and effluent from wastewater treatment plants) and inputs of toxins due to spills or leakages of oil and gas, pesticide application, and industrial activities (Gosselink & Maltby 1990). Increased freshwater runoff may reduce salinity in estuarine wetlands (Greer & Stow 2003), while damming, ditching, and reservoir construction may increase salinity in coastal wetlands (Ehrenfeld 2000a). It is well known that climate and air quality are also altered by urbanization, which results in a 'heat island' effect of higher temperatures, higher atmospheric concentrations of oxidants, nutrients, and dust, and lower wind speed and net radiation (Ehrenfeld 2000a). In some settings, herbivores may become more abundant in urban settings. One example is non-migratory populations of Canada geese (*Branta canadensis*), which has established breeding populations in or near wetlands and other aquatic habitats in many cities and can have strong effects on wetland plant communities due to grazing or trampling of plants (Baldwin & Pendleton 2003).

Changes in wetland plant biodiversity attributable to urbanization

These changes in hydrology, geomorphology, and water quality can cause subtle and dramatic shifts in wetland plant biodiversity. Here a few examples are presented in detail to illustrate some of the ways that urbanization affects wetland vegetation.

Forested wetlands dominated by Atlantic white-cedar (*Chamaecyparis thyoides*) are a species-rich but relatively rare wetland type characterized by acidic, low-nutrient conditions that occurs along the Atlantic and Gulf of Mexico coasts of the US. Joan Ehrenfeld and associates have studied the influences of suburbanization on these wetlands in New Jersey. Vegetation changes associated with urbanization in these wetlands include a decrease in *Chamaecyparis* seedling density and *Sphagnum* spp. ground cover and an increase in weedy non-native species (Ehrenfeld & Schneider 1991). These changes in vegetation were associated primarily with higher concentrations of ammonium and orthophosphate in groundwater and surface water. While human activities caused either stabilization or more fluctuation in water level, water level changes were not correlated closely with plant species composition. Nonetheless, the plant communities of developed sites had more plants adapted to non-wetland conditions (facultative-upland and upland species), and fewer true wetland species (obligate species) than non-urban sites (Ehrenfeld & Schneider 1993). Later studies indicated that the pattern of invasion of non-native species is more complex, as not all urban sites, including small patches, are heavily invaded (Ehrenfeld 2005) and the pattern of invasion differs among different urban land-uses, with both adjacent industrial land-use and vegetated upland related to lower invasion than in residential areas (Ehrenfeld 2008).

Fens are another type of low-nutrient wetlands that may be particularly vulnerable to contaminants in groundwater. In two species-rich Illinois fens, groundwater plumes of sodium and chloride ions (from a septic tank at one fen and a road receiving de-icing salts at the other) were closely related spatially to the occurrence of *Typha angustifolia*, an invasive plant tolerant of brackish water, and the absence of the typical fen species *Scirpus acutus* (Panno *et al.* 1999).

Estuarine and marine wetlands are also subject to indirect effects due to urbanization. In the Los Peñasquitos Lagoon in San Diego, California, an order-of-magnitude increase in dry season stream discharge occurred during the twentieth century due to watershed urbanization. Greater freshwater discharge has decreased estuarine salinity and thereby facilitated the invasion of less-salt-tolerant species, increasing the areal extent of brackish marsh and riparian vegetation and decreasing the areal extent of salt panne and mudflat habitats (Greer & Stow 2003). In a similar example, in the Mar Menor coastal lagoon on the Mediterranean coast of Spain, increases in freshwater runoff due to agriculture (primarily) and urbanization have caused widespread conversion from more- to less-salt-tolerant habitats (salt steppe into salt marsh and salt marsh into reedbed) (Esteve *et al*. 2008). Urbanization-related changes in nutrient loading as well as salinity can also alter coastal wetland vegetation: in a study of 22 salt marshes in Rhode Island, shoreline development was associated with lower soil salinity and higher nitrogen availability, and strongly correlated with the cover of the invasive grass *Phragmites australis* (Silliman & Bertness 2004). In the same study, *Phragmites* dominance was related to an almost-three fold reduction in plant species richness.

The vegetation typical of some wetlands results from natural disturbance processes, and ceasing or altering those regimes results in changes in plant communities. One study in a 103-ha wetland in the urbanizing city of Orlando, Florida, found that fire suppression, combined with altered hydrology, was facilitating the invasion of the tree *Acer rubrum* and the loss of the native sawgrass, *Cladium jamaicense* (Knickerbocker *et al*. 2009).

2.1.4 Synthesis and prospective view

The studies reviewed here provide clear evidence of the overwhelming but complex effects of urbanization on wetlands. The construction of buildings, parking lots, roads, parks, sewers, flood control structures, landfills, and other urban features has directly resulted in the loss of wetlands and their associated species and community diversity. Furthermore, the wetlands remaining in the urban landscape are fragmented and isolated, which reduces exchange of propagules and leads to lower plant community heterogeneity. Furthermore, changes in hydrology, geomorphology, climate, and nutrient and sediment loading, as well as toxins and urban fauna resulting from urbanization, continue to shape plant biodiversity in wetlands. While the number of plant species in many urban wetlands is similar to, or even higher than, their non-urban counterparts, much of the total species richness results from non-native or invasive species. Nonetheless, wetlands remain hot spots of plant biodiversity in cities, supporting species not found in other urban habitats. Furthermore, by their very rarity, the remaining wetlands are proportionally more important as sources of plant propagules and habitat for wildlife.

Given their importance for plant biodiversity, as well as other functions such as habitat, water quality, and flood storage, preservation of remaining wetlands and restoration of former wetlands should be a priority in urban areas. Preservation and restoration efforts often have the support of the general public, who benefit from the ecosystem services of wetlands and appreciate the aesthetic and ecotourism aspects of wetlands in the urban settings. One obstacle to preservation is a view that the presence of certain plant species, both native and non-native, represent a 'degraded' plant community, or that wetlands in urban areas have low 'floristic quality'. These phrases are value judgements about the merits of certain species, and often assume that non-native plants or those native plants that become dominant in urban wetlands have little ecological or human value. However, this is simply not true, as all of these plants help to improve water quality, sequester carbon, reduce shoreline erosion, provide fish or wildlife habitat, or provide other services and functions. It is worth bearing in mind that the presence of these species and communities is the result of our own actions in urban environments. As long as we have cities we will continue to support wetland plant species and communities adapted to the urban environment. More appreciation of the importance of this urban wetland plant biodiversity will go a long way toward conserving these important ecosystems.

CHAPTER 2.2

Potemkin Gardens: Biodiversity in Small Designed Landscapes

Martin F. Quigley

The phrase 'Potemkin village' has become a label for anything that has surface but no substance, a two-dimensional façade. Its origin was in an act of landscape deception: in 1787, after the army of Catherine the Great, led by Field Marshall Potemkin, had conquered the Crimea, the Empress and the Field Marshall led an entourage of courtiers and foreign allies through the region. Since the land was sparsely settled before the conflict, and had been devastated in the fighting, Potemkin arranged for a series of 'villages' to be created as stage sets along the route, made with painted screens and temporary structures. These scenes were even peopled with villagers and livestock, some of whom were moved nightly to stay ahead of the tour. The intent was to impress the visitors, traveling through a bleak and unpopulated landscape, with the amenity and development potential of this area by creating a bucolic vision of stability and productivity.

2.2.1 Introduction

Urban gardens fulfil a human need for beauty and a connection with the natural world, however tenuous that link may be in a biological sense. The nineteenth-century landscape designer Frederick Law Olmstead, who created the template for city parks in the US, was not a pioneer in his belief that green and open parklands were essential to the health of city populations; this idea had started in Britain almost at the dawn of the industrial revolution, and had gained currency in post-Napoleonic France as well. It is now accepted that even very small gardens and green spaces, such as vignettes from hospital windows, container plantings, or even single trees in the most urbanized of sites, can have a significant and positive effect on human well-being. Humans simply feel and function better when they have access to parks, gardens, or greenery in their daily routines (Nassauer 1997). Whether such small urban gardens and planted areas have a meaningful role in ecosystem function is a larger and more problematic issue. Studies of biodiversity in the urban landscape have recently proliferated (e.g. Savard *et al*. 2000; Zerbe *et al*. 2002; Jim & Chen 2008; Colding & Folke 2009; Underwood *et al*. 2009), and their findings reflect both the fragility and tenacity of plant and animal populations and communities in severely altered or newly created habitats.

2.2.2 Species diversity

Most commonly, the term biodiversity refers to the number of species in a community or habitat, with no distinction made between rare or abundant species. Ecologists often distinguish between species richness—the total number of species—and species diversity, which incorporates the relative abundance of each species in addition to the total number (Carroll & Salt 2004). Magurran (2004), calling species the 'common currency of diversity', defines biodiversity simply as 'the variety and abundance of species in a defined unit of study'. Magurran also emphasizes that 'species abundance distributions can be used to describe the structure of communities and shed light on the ecological processes that underlie that structure'. It is this aspect of process that seems often to be absent from the species counts in urban plant communities. Kinzig *et al*. (2001) describe a dichotomy in current modelling of ecosystems: modellers either consider heterotrophic diversity as 'fat on the autotrophic backbone or ecosystems, or in sharp contrast, consider autotrophs as merely fodder for ecosystem processes conducted primarily by heterotrophs'. In most studies of urban landscapes, heterotrophs, non-vascular plants and fungi, along with the significantly modified abiotic conditions, are given scant mention. Kinzig *et al*. (2001), however, note that: 'variation in diversity in one trophic level is likely to modulate the impacts of variation in another trophic level'. Therefore, in the extremely altered abiotic and biotic conditions of urban garden spaces, there may be multiple trophic disconnects among those plants that are installed for human amenity. More importantly, functional or trophic biodiversity has sometimes been explored in extant native patches augmented with exotic species (Storch *et al*. 2007 Loram *et al*. 2008;), but rarely in *de novo* landscape installations such as borders, small gardens, and urban ornamental plantings (but see Gaston *et al*. 2005).

Most studies of urban ecosystems focus on function and/or biodiversity in fragments of indigenous landscape, e.g. drainage ways, 'urban wilderness', and other intentional or inadvertent remnants (Hester *et al*. 1999; Nature in the City 2008). Such investigations do not typically include landscapes that have been significantly altered or those that have completely replaced original plant communities (lawns, gardens, planters, golf courses, roadway verges, and medians). These created or designed plantings commonly bear scant resemblance to indigenous plant assemblages in structure or function. Dailey *et al*. (1997), in a discussion of ecosystem services, focus on the value of 'natural' ecosystems, asserting that human disruptions to these systems 'are difficult or impossible to reverse at any time scale relevant to society'. This position deprecates the possibility that any human-installed landscape could contribute meaningfully to ecosystem function and taxonomic diversity.

In general, optimistic reports of urban taxonomic diversity are focused on those remnant patches of indigenous vegetation that have been altered but not totally destroyed, sometimes including those that have been augmented with additional plantings (see Dunn and Heneghan, Chapter 2.4). Many of these areas, such as large parks, greenbelts, and riparian zones, have been further altered by cultural maintenance such as selective species removal, heavy pruning, and the introduction of non-native plants, or native but formerly extirpated species. While evaluating the effects of urbanization on plant assemblages and biodiversity, many investigators quantify the extirpation of native species but do not consider the introduction of new species, either those formerly and locally indigenous, or those originally alien to that habitat (cf. Bastin & Thomas 1999; Cornelis & Hermy 2004; Grimm & Redman 2004; Mehtälä & Vuorisalo 2005; Sal'nikov & Pilipenko 2005; Alvey 2006, Godefroid & Koedam 2007; Knapp *et al*. 2008b; Lawson *et al*. 2008; Ricotta *et al*. 2008; Puth & Burns 2009; Underwood *et al*. 2009, Biodiversity Conservation, Sydney Olympic Park). A more inclusive approach includes and enumerates introduced species, as seen in studies by Acar *et al*. (2005), Akinnifesi *et al*. (2009), Davies *et al*. (2009), Jim (2002), Millard (2008), Mueller (2007), and Ramage and Dukes (unpub).

2.2.3 Structural biodiversity

Since local biodiversity is affected by the size of a habitat patch or community, as well as its distance from other habitats, urban plant patches may be significantly less speciose than wildland areas of the same size (see Dunn and Heneghan, Chapter 2.4). On the other hand, human agency can install very high numbers of species (Smith *et al*. 2005) in relatively small areas—though such plant 'communities' cannot survive without continued human intervention. Biodiversity in cities may also be higher than in surrounding agricultural areas because farmed monocultures are extremely simplified landscapes compared to either natural ecosystems or to ornamental plantings. But this still begs the question of the functionality of installed urban landscapes, as opposed to partially 'native' parklands and other large landscape fragments within the urban mosaic. Beyond considerations of taxonomic variety exist landscape linkages that are essential to the maintenance of species diversity and richness (Jongman 2004), and for most urban designed landscapes and gardens, ecosystem links are tenuous at best.

Even if species numbers seem high per unit area, taxonomic diversity in urban plantings may be deceptive in another way: in annual and herbaceous

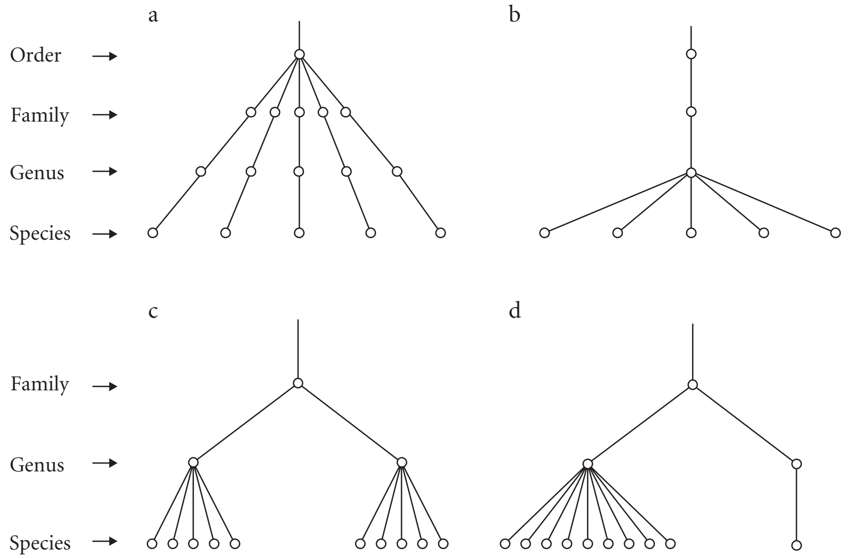

Figure 2.2.1 Taxonomic distinctness is based on the the average pairwise path lengths between species in an assemblage, based on presence/absence data and ignoring species abundances. The four hypothetical assemblages are therefore ranked in an intuitive way. (Reprinted with permission from Magurran 2004, Blackwells)

perennial plantings, many species (and cultivars) may belong to relatively few genera or families (e.g. Asteraceae) (see Fig. 2.2.1). Functional and stratum diversity is more important for landscape sustainability than are numerous but closely related species within a single stratum.

Though suburban gardens, particularly in Great Britain (see Gaston *et al*. 2005), may be planned for explicit interspecific function, such as insect and vertebrate food sources and nesting sites, few urban gardens or ornamental plantings are created to establish plant interactions, microclimate conditions, nutrient cycling, or interspecific facilitation. The greatest hindrance to functional biodiversity in urban gardens and other plantings is that design intentions are usually very specific for human perception of the aesthetic, and are focused on visual rather than functional values. The designed and constructed urban landscape consists of hardscape (that is, abiotic components: irrigation, growing media, paving, utilities, furniture, signage, fountains, and other features) and only secondarily the 'soft-scape' of plant materials chosen for static characteristics (colour, texture, size) or for their maintenance requirements. In cities, the hardscape elements are generally the primary focus of design, and are given the larger share of the budget. Even if consideration is given to providing adequate growing conditions for these installed plantings in soil media, nutrients, and irrigation, the profound changes in trophic interaction, abiotic conditions, mycorrhizal symbionts, subsurface hydrology, and most especially soil compaction (Quigley 2004; Pavao-Zuckerman 2008) may profoundly influence viability of both indigenous and introduced plants. Early failure of planted trees in particular, for a variety of reasons, is extremely common in urban plantings (Jim 1993; Bassuk & Trowbridge 2004), so that healthy tree canopy cannot be presumed to exist, even where a good variety of tree species has been planted.

Despite urban garden designs that lack any ecological basis in their conception, local taxonomic biodiversity may occasionally be higher in altered landscapes than in intact indigenous patches, for two reasons. First, human landscape modifications and disturbance may increase the abundance of normally uncommon ruderals, and large canopy openings combined with understory disturbances may also promote germination and growth of shrubs, lianas, and herbaceous species that would

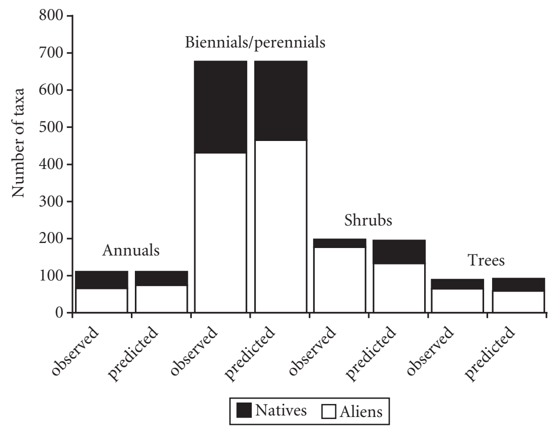

Figure 2.2.2 The observed and expected contributions of alien and native taxa to different growth forms in the garden flora of 61 urban gardens, Sheffield, UK. (Reprinted with permission from Smith *et al.* 2005, Elsevier)

otherwise be absent in an intact patch (Quigley & Platt 2003). Second, the introduction of non-native species, and the absence of competitive exclusion, may significantly raise the number of species occurring in urban landscapes as opposed to suburban gardens (Knapp *et al*. 2008a; Stewart *et al*. 2009).

2.2.4 Design

Horticultural introductions to urban areas are as old as human commerce, and a very significant component of city garden biodiversity. However, there is increasing popular emphasis in urban garden and landscape design for the use of 'native plants', on the assumption that species adapted to the pre-urban ecosystem will now be equally successful in constructed landscapes (Fig. 2.2.2). Natives are assumed to be 'better adapted' (Kendle & Rose 2000). However, there is some fallacy in this assumption. Native plants are adapted to specific 'native' conditions of soils, hydrology, temperature, and trophic status. If removed from these conditions and treated as any other horticultural introduction, they may be no more adapted to urban conditions than are any other introduced species from another continent. This in fact is why so many 'native plant gardens' in cities are such disappointing failures: their designers assume that being native to a local climate is the only prerequisite for plant health and growth. Some researchers do recognize that non-native plant species are in fact part of the local diversity (Kendle & Rose 2000; Akinnifesi *et al*. 2009). Worldwide, many alien plants introduced into cities have persisted for centuries, but have not spread into adjacent rural areas (Botham *et al*. 2009); this may be attributed to their ability to flourish only without competition, or to their dependence on human agency for reproduction and dispersal.

The non-native ornamental plants that are introduced by humans, while numerically diverse by species counts, may be very limited in higher taxonomic diversity. Studies from around the globe also suggest that certain taxa, such as beetles, rodents, some birds, and ruderal plant species, may flourish in urban zones. In Great Britain, especially, residential gardens are promoted as habitat for wildlife as well as human comforts (Hemenway 2001). However, other essential ecosystem components, notably fungi and woody plant populations, are rarely maintained or re-established in the scope of human planting installations.

It has been hypothesized that designed and installed landscapes *de novo* can remediate, or even restore, ecosystem function at a significant level. A common assumption in many papers is that remnant fragments of native vegetation in parks, riparian zones, gardens, and suburban 'green spaces' provide a viable matrix for the survival of

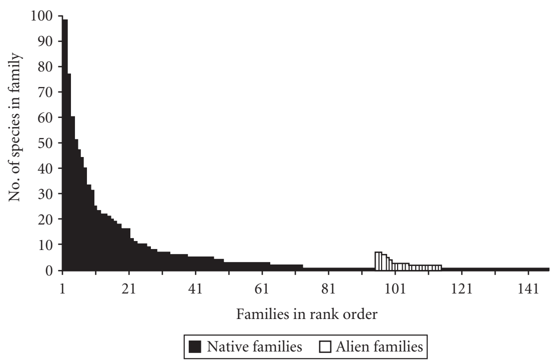

Figure 2.2.3 Species accumulation curves for inventories of vascular plants in 61 urban gardens, Sheffield, UK. Curves are based on 100 randomly shuffled runs. (Reprinted with permission from Smith *et al.* 2005, Elsevier)

significant portions of pre-disturbance communities, though this is often not the case (see Fig. 2.2.3). From a design perspective, the single greatest obstacle to such recovery, aside from soil and hydrological perturbations, is the absence or elimination of one or more vegetative strata from almost all installed landscapes (for example, only turf and shade trees in a park, or only herbaceous plants in a border).

Urban gardens in general are designed for static visual appeal, often requiring that plant growth and floral or fruit development are to be suppressed by shearing and 'shaping'. Public expectations for installed landscape are usually very specific and very circumscribed, calling for reliable greenery, seasonal colour, and above all tied to a somewhat illogical requirement of 'low maintenance'. All landscape elements, whether hardscape or plantings, have absolute requirements for regular maintenance. If proper plants are well chosen for the specific site conditions, they are by definition lower in required maintenance than are plants not well adapted to the site. But for most designers, materials and layouts conceived primarily for a product of low maintenance reduces plant choices to a very limited palette of 'tried and true' cultivars that cannot begin to replicate functional or structural diversity. Further, the horticultural care given for such plantings, whether prescribed by specifications in construction documents, or performed ad hoc by minimally trained workers, treats them more as furniture than as living organisms.

Diversity trends in urban areas are affected by two very different trajectories of plant choice and installation. Professionally designed gardens or other landscape elements tend to have larger numbers of plants, but representing fewer species, than do amateur or home gardens where many species occur, but each represented by very few specimens or individual plants. Landscape design professionals tend to have very limited botanical or horticultural exposure during their training. Simultaneously with the introduction of computer assisted design (AutoCAD) into all landscape architecture curricula, the instruction of botany, horticulture, ecology, and even plant identification has all but disappeared from most design programs particularly in the US. The new computer courses required that other offerings be reduced, and plant related courses were those most deleted from curricula. Inexplicably, then, choosing the living elements of the landscape have been largely relegated to contractor responsibility. Especially in the US, many landscape architects receive almost no education in horticulture, botany, or ecology; as a consequence, they are almost proudly ignorant of plant identity or processes, which are to be left to the maintenance workers. Their planting plans tend to be based on short lists of very generalist species, and plants are selected and combined without reference to similarity of horticultural requirements. Landscape architects and garden designers commonly choose plants for single ornamental qualities, not for dynamic attributes or their interaction with other species. One tree is chosen for 'autumn colour', another for 'light shade'. A shrub is chosen for a single specific function—pedestrian barrier, visual screen, fast growth rate—or for appearance (dark evergreen foliage, for example). This kind of plant selection reduces plants to the status of furniture. Thus, for example, a flowering shrub that is used as a hedge may be shorn annually, precluding any floral display. A clump-forming understory tree chosen only for its spring flower display may be planted as a single-trunk specimen in full sun exposure, ensuring a shorter life-span under challenging conditions. Herbaceous perennials and blooming annuals are generally selected for the single attribute of seasonal flower colour and duration of bloom, and only secondarily for size or foliage or the specific site conditions. Both herbaceous and woody plants with completely different water or nutrient requirements are often installed willy-nilly, in ignorance of their disparate horticultural requirements. Rarely is any attempt made to install, in urban plantings, the full array of strata (groundcover, shrub layer, understory trees, and canopy) that a natural woodland landscape would have. Urban plantings instead often resemble agricultural layouts: single species rows or masses, plants arrayed geometrically, individual plants not touching, and a great deal of naked soil or mulch between them.

Some urban landscape design is explicitly intended to be two-dimensional, such as the ubiquitous 'foundation plantings' that consist of a single row of a single species of shrub, installed at the foot of all walls wherever possible. Foundation shrubbery and edge plantings with no living ground cover are usually over-mulched with organic layers (bark, shredded hardwood or softwood), or inorganics (fabric, stone, shredded tires, etc.). Such plantings are more like green street furniture than living organisms.

Another two-dimensional landscape is the urban lawn. Instead of a multi-species turf as sometimes found in nature, containing legumes among many other herbaceous species, city lawns are intended, installed, and maintained to be monocot monocultures, though they may contain a significant number of native and introduced herbaceous weeds. Even in a regularly mowed lawn in temperate North America more than 100 herbaceous species were counted in a 900 m^2 area (Carroll & Salt 2004). However, lawns have also been labelled 'biological deserts' (CBIN), not only as species monocultures, but as sites rendered all but sterile with constant applications of pesticides and herbicides in an attempt to eliminate all other organisms than the selected turf cultivar. In temperate New Zealand, Stewart *et al*. (2009) found significant diversity of herbs and forbs in (mostly non-native) urban lawns, and an interesting decline in species diversity as lawn size increased. They also found a higher native 'weed' presence in the most-tended residential lawns (frequent mowing and removal of clippings), while less intensively maintained turf in park areas had more competitive exclusion of both native and introduced weed species. The dynamics of turf are highly variable, and it is a pity that the almost universal public expectation of single-species lawns, chemically and mechanically maintained, precludes the development of species-rich and self-sustaining herbaceous turfs as seen in moist climates where grazing animals are kept.

Popular expectations of monoculture are not confined to lawns and annual plantings. Especially in North America, urban street trees are almost always installed in monospecific rows, with trunk spacing legislated to preclude canopy contiguity or overlap. Neither Dutch elm disease, the Chestnut Blight, the Emerald Ash Borer, or other pests and pathogens that have eliminated vast numbers of urban trees has been sufficient to change a stubborn belief in the aesthetic requirement for single species plantings along city streets. The urban forest—the sum of street trees, park trees, and those on private property—may have a certain numerical diversity (Jim 2000), but generally lacks any canopy integration, and certainly lacks any understory component whatsoever, whether juveniles of the canopy species, or shrubs and sub-canopy trees of other species. Tallying these isolated tree species in terms of biodiversity is certainly a Potemkin-like exercise in self-deception: the overall variety has no real basis in ecosystem function.

Urban institutional or commercial entries and courtyards, like small residential gardens, tend toward more displays of herbaceous plants installed through a blanket of mulch than to layered perennial groundcovers and woody species. Seasonal annuals predominate in such places, though perennials are now enjoying a resurrection of sorts, especially in private gardens. Part of this new aesthetic is the inclusion of a growing variety of perennial bunch grasses with diverse ornamental characteristics, though again, even a high number of grasses does not increase trophic diversity. Though the number of species and cultivars being installed in urban gardens and small plantings seems to be increasing, there is no concomitant increase in structural or functional diversity. One hampering condition of the urban installed landscape is the very tightly circumscribed limit of work for each planting: individual projects do not foster landscape connectivity. Project lines are religiously held: there shall be no work done, or plants installed, on the other side of an often arbitrary delineation.

Herbaceous borders and very small gardens in residential areas, however, may show higher diversity to be positively correlated with higher housing density (Marco *et al*. 2008). This is particularly true in cultures where culinary plants, independent of ornamental value, are an essential household feature (and see Hemenway 2001). In northern Brazil, for example, Akinnifesi *et al*. (2009) found very high diversity (186 species in 68 families) in relatively small home gardens. In contrast to temperate North America, where most vegetable and fruit species are introductions from Eurasia, in these Brazilian gardens the majority of fruiting trees were natives. The authors suggest that because some of these species are endangered in the wild even small urban gardens may be significant in their preservation. This strategy has been suggested for urban gardens in a variety of climates, but all plants in urban gardens are *ex situ* at best; human agency is required to maintain the specimens, and both pollen and propagule dispersal is extremely limited.

Some specific landscape functions (e.g. bioretention areas and green roofs) may preclude the installation of diverse plant communities in urban plantings (Kazemi *et al*. 2009). Large- scale roof plantings in temperate zones tend to be planted with succulents, presumed to be 'low maintenance'. Such areas are not intended to function as landscape components, but appear as large discrete planters in the urban fabric. Wet spots created in drainage zones are quickly overtaken by sedges and rushes. These types of plant groupings, however, though physically distinct or isolated from other patches, appear to be particularly susceptible to colonization by air-borne or bird-vectored weeds, and often provide habitat for many species of insects, birds, and mammals. The development of these community patches may be discouraged by legislated maintenance requirements, however, or the fear that biomass accumulation will impair stormwater filtration and flow.

In extant natural habitats, biodiversity measures are commonly based on an implicit (if unspoken) assumption of temporal and spatial continuity: the organisms being counted are components of a self-sustaining community structure and are able to maintain their populations through reproduction, immigration, and dispersal (Daily *et al*. 1997; Alvey 2006; Puth & Burns 2009). In urban constructed landscapes, however, ornamental and horticultural considerations, not to mention severe limitations in size, generally override any idea of self-sustaining native plant gardens. In such small areas, plant combinations are usually installed for strictly ornamental purposes (in herbaceous borders, planters, medians, and gardens over structure) or for some other specific and direct human benefit (shade, screens and barriers, stormwater basins for filtration, for example). Such gardens are neither perceived nor intended to have any reproductive or structural continuity from one year to the next, or in the case of some ornamental annuals, from one season to the next. Instead, they are completely dependent on human installation and maintenance rather than self-sustainable ecosystem function. Therefore, the appraisal and even the existence of such diversity may be without significant ecological value, even if species are numerous.

Is a three-dimensional urban landscape mosaic, with a good measure of both taxonomic and functional diversity, an impossible goal? While species richness in raw numbers may remain relatively stable, or even increase, as indigenous and fragmented habitats are altered by urban sprawl and ornamental plantings are installed there are essential and overriding changes in both functional and structural biodiversity, negating the ecological value of basic taxonomic diversity. There is no current evidence that the creation of designed urban landscapes, from small gardens to larger and usually linear areas of installed plantings, can reproduce true biodiversity or function in a self-sustaining community.

2.2.5 Conclusion

Given the limited botanical and horticultural education of most landscape architects and designers, and the often unrealistic expectations of the consuming public, biodiversity as such may be seen as irrelevant in urban plantings. The overriding design goal for most urban landscapes, and certainly for most of the very small installations that occur in high-density urban areas, is indeed a kind of Potemkin façade: a snapshot selection of plants chosen for two-dimensional visual effect, and without biological consideration. Small urban plantings are deconstructed fragments of imaginary landscapes, lacking meaningful structural and functional diversity. As our carriages speed through the urban thoroughfares, we respond to artificial and unsustainable gardens and greenery as if they were actually replicates of natural scenes, but the depth and strata that comprise a real plant community do not exist. Therefore, while urban gardens and other installed plantings may be species rich by some measure, their lack of connectivity, function, or self-sustainability makes them Potemkin landscapes at best.

CHAPTER 2.3

Vegetation of Urban Hard Surfaces

Jeremy Lundholm

2.3.1 Introduction

Urbanization has long been noted as a force reducing vegetative cover with increasing proximity to the urban core (Sukopp *et al*. 1979), but despite this, there is an even longer history of interest in the plants that colonize anthropogenic surfaces including building walls, pavements, and ruins. The construction of buildings and human settlements typically involves replacement of soils and natural vegetation with harder surfaces of stone, concrete, and asphalt. Botanical investigations have identified a number of plant species that thrive on these anthropogenic surfaces and many have sought to understand the diversity and origins of these types of vegetation (Rishbeth 1948; Segal 1969; Darlington 1981). Darlington (1981) estimates that for every 10 hectares of horizontal urban land, there is approximately one hectare of wall in Europe. Hard surfaces are considered harsh substrates for vegetation due to lack of rooting space, low moisture availability, and disturbances such as trampling. Nevertheless, these habitats harbour an interesting and potentially important spontaneous flora.

2.3.2 Hard surface types

Vertical hard surfaces can be divided into several main categories: free-standing walls, river walls, building walls, and retaining walls (Brandes 1992) (Table 2.3.1). Horizontal hard surfaces are perhaps less varied but nevertheless can be divided into pavements and rubble (Table 2.3.1), with rubble grading into any number of 'wasteland' habitat types. Walls are generally better studied with the bulk of studies originating in Europe and the Mediterranean.

2.3.2.1 Walls

The literature on walls generally deals with old stone walls: old walls have opportunities for colonization by many species. One of the key variables that determine species composition of wall vegetation is the material from which the walls were originally constructed. Stone walls can be divided into dry (piles of stones with no connective matrix) (Fig. 2.3.1a) and mortared (substances used to bond stones together) (Fig. 2.3.1b). Mortared walls have changed over the centuries with the development of improved mortar mixtures. Early European walls used softer mortars of calcium carbonate and clay which provide good opportunities for colonization by vascular plants (Darlington 1981). With the widespread adoption of Portland cement in the late 1800s the mortar became harder and more alkaline. It is widely understood that the composition of both stones and mortar of walls in turn influences species composition. Other types of walls, including those constructed of mud and bricks, support different species than mortared walls (Varshney 1971). Dry stone walls may be harder to colonize than mortared walls due to greater aeration leading to lower moisture content, and support vegetation distinct from that of mortared walls (Payne 1989).

In general, while rocks can sometimes be colonized directly by algae, cyanobacteria, lichens, and bryophytes, most biomass in these ecosystems comes from vascular plants which colonize the spaces between rocks or bricks (Lisci & Pacini 1993). Suitable substrate for colonization depends on the presence of cracks in the hard surface and the breakdown and/or deposition materials that accumulate to create a substrate for plant roots. Colonization of walls

Table 2.3.1 Characteristics of main types of urban hard surface habitats

Type	Main Characteristics
Vertical	Gravity prevents much substrate from accumulating, exposure to wind/thermal stress
Free-standing walls	Drought-prone if isolated
Dry stone walls	Moisture and substrate very limiting to plant growth
Mortared walls	Best-developed wall vegetation, breakdown of mortar provides space and substrate for plant growth
Building walls	Less rooting space than free-standing walls, more frequent maintenance, less vegetation
River walls	Influence of flooding and sediment deposition,
Retaining walls	Plant access to soil behind façade greatly increases productivity
Horizontal	
Pavement cracks/edges	Trampling is major disturbance to plants, nutrients can accumulate, compacted soils
Roofs	Little trampling, roof construction techniques determine amount of substrate that accumulates, impermeable surfaces can trap water and support wetland species
Rubble	Little trampling, rubble sites typically succeed to more productive communities

by plants and other photosynthetic organisms depends on several key environmental gradients. Perhaps most important is time since construction. Initial substrate conditions on mortared walls are often highly alkaline (pH 11–12) and time is required for mortar to weather and become more neutral. Temperature fluctuations aid breakdown of hard mortar and allow more room for plant roots. Once plants establish, roots can mechanically expand cracks and trap particles, increasing the organic content of the substrate and further reducing alkalinity (Darlington 1981). Many authors note that older walls have greater coverage of vegetation (Kent 1961), with coverage peaking between 100–500 years after construction for European walls. Recent studies confirm that more weathered surfaces support more

Figure 2.3.1 a) Dry stone wall (Shaanxi Province, China); b) Mortared stone wall with fern (Hong Kong, China); c) Wall base with accumulation of debris and plants (Guelph, Canada); d) Wall top with *Poa compressa* (Halifax, Canada). All photos: J. Lundholm

trees on tropical retaining walls (Jim 2008). While the alkalinity of mortar may create a hostile chemical environment for plant root growth, a study in Poland found that with birch trees growing on walls, substrate nitrogen and phosphorus content were not limited to tree photosynthesis (Trocha *et al*. 2007).

Moisture availability emerges as the most important environmental constraint on wall vegetation development (Woodell 1979). At a regional scale, wall vegetation reaches the greatest coverage and species diversity in oceanic climates with high rainfall and humidity and relatively low temperature fluctuations, like western Europe (Segal 1969; Darlington 1981). Free standing walls outside areas of high humidity are less colonized by vegetation (Gilbert 1992), because they warm and dry quickly in the sun (Darlington 1981). Likewise, common algal assemblages on hard surfaces (Prasiolales) are primarily found in cool humid climates (Rindi 2007). In more arid areas, where annual precipitation is below 300 mm there is seldom any wall vegetation at all (Weinstein & Karschon 1977). At local scales, wall aspect has a major effect on vegetation coverage and composition, likely due to its influence on humidity and surficial moisture (Woodell 1979; Rindi 2007). Vegetation differences between north and south facing walls are a common feature of many early reports (Darlington 1981). In the northern hemisphere, south-facing walls with the same substrate tend to have more crustose lichens and bare surfaces, whereas greater coverage of vascular plants and pleurocarpous mosses are on north-facing walls (Darlington 1981), but other moss species may prefer the sunny sides (Rishbeth 1948). While light can limit some wall vegetation (Segal 1969), most evidence suggests that more sheltered locations support more biomass (Jim 2008). River wall vegetation is generally not limited by moisture, but faces other constraints such as more frequent repairs to the infrastructure or replacement of walls that prevents long-term development of vegetation (Francis & Hoggart 2008).

The slope of a wall affects both water and substrate retention, with more vertical surfaces generally being dryer and presenting fewer opportunities for plants to establish (Darlinton 1981). Some walls have a range of slopes or other features that present distinct habitats for plants and other organisms. Wall zonation is a well-known feature in Europe, with at least four distinct microhabitats appearing on most free-standing stone walls (Segal 1969; Darlington 1981): the wall base, typically the wettest of wall microhabitats, is horizontal and thus tends to accumulate debris and nutrients (Fig. 2.3.1c), leading to soil development in some cases and lowering of pH, which can in turn support species (calcifuges) which are adapted to lower pH soils than the basic substrates found higher on the wall; the middle level, which is most exposed can sometimes retain more nutrients closer to the wall base and can also favour species that require more fertile conditions (Oberdorfer 1975); the upper level of vertical surfaces often has greater vegetation cover than the middle, most exposed surfaces, and is influenced by abiotic and biotic features of the wall tops; wall tops (of free-standing walls) typically have greater species diversity than vertical faces due to greater access by animals that can transport propagules, and accumulation of more substrate than possible on vertical surfaces (Fig. 2.3.1d) (Duchoslav 2002; Pavlova & Tonkov 2005). Other features, such as horizontal ledges, also allow for greater substrate accumulation than vertical surfaces and can support more vegetation (Lisci & Pacini 1993). Ledges or other features that block rainfall from reaching surfaces can also constrain the growth of cyanobacteria and algae (Rindi 2007). Other spatial patterns of vegetation can be created by the configuration of mortar and rocks/bricks that determine where colonization is more likely (Darlington 1981). These microhabitat features are all primarily defined by the quantity and quality of rooting substrate that can accumulate, and the availability of moisture (Lisci & Pacini 1993). While most plants tend to be more abundant lower on vertical surfaces, likely due to greater seed rain and moister microsites (Lisci & Pacini 1993), trees on Hong Kong walls are more abundant on taller walls, because the lower areas of these walls are disturbed by human activities (Jim 2008).

2.3.2.2 Pavements

Pavements are often described as horizontal walls in the literature, since in the past they were often

Figure 2.3.2 a) Pavement crack vegetation with trampling tolerant plants (*Plantago major*, *Polygonum* sp.; Halifax, Canada); b) Algae and cyanobacteria colonizing a parking lot (Hawaii, USA); c) Common cliff tree species on a river wall (*Thuja occidentalis;* Guelph, Canada); d) Strangler fig on a wall. (Hong Kong, China). All photos: J. Lundholm

composed of the same materials as walls. Several important differences result in distinct vegetation on paved surfaces, the most important of which is the presence of trampling (Fig. 2.3.2a). The least trampled pavements tend to have similar vegetation to that of wall bases (Woodell 1979) but in cases where pavements are more exposed to light and wind than wall bases, the vegetation can differ substantially (Lundholm & Marlin 2006). Like vertical hard surfaces, the vegetation in paved areas depends on the provisioning of rooting substrate: wider cracks between paving stones or other hard surfaces leads to vegetation more similar to that on roadsides (Segal 1969). Climate is also important in determining the composition of pavement vegetation, with more drought-resistant species occurring more inland, and with halophytes occurring closer to the coast (Segal 1969), however in many northern cities, salting of roads and sidewalks is increasingly common, and vegetation of inland pavements must have some tolerance of salt (Woodell 1979). In common with wall bases, horizontal pavements receive high nutrient levels due to runoff from buildings or direct deposition by animals (Segal 1969; Woodell 1979).

Trampling leads to several ecological effects that constrain vegetation development on paved surfaces. First, trampling causes compaction of what little substrate is available for plant growth. This can result in poor conditions for bacterial nitrification, leading to a build-up of organic nitrogen (Segal 1969). Trampling disturbs and relocates substrate and directly damages plants, leading to a small set of trampling-tolerant species that dominate in these areas. Some plant species typical of walls can colonize pavements, but typically only in areas close to walls that receive little trampling (Darlington 1981). Trampling-tolerant species typically have prostrate growth and include *Sagina procumbens*, *Poa annua*, *Bryum argentium* (a moss), and *Plantago major* (Darlington 1981; Brandes 1995). These species are found in similar habitats on both sides of the Atlantic (Darlington 1981; Lundholm and Marlin

2006), but there are very few accounts of vegetation on paved surfaces from other regions.

2.3.3 Biota

The vegetation of hard surfaces comprises not just vascular plants but, for most authors, any photosynthetic sessile organisms. The distinct biology of these taxonomic groups determines differences in their distribution across the variety of hard surfaced habitats, but moisture limitations appear to be a common constraint on their growth and development.

2.3.3.1 Algae/cyanobacteria

Green algae and cyanobacteria are considered together because they colonize similar habitats in urban areas, often areas unavailable to plants requiring substrate for rooting. They are largely limited by moisture availability but are nonetheless common on vertical and horizontal hard surfaces, especially in oceanic climates (Darlington 1981) (Fig. 2.3.2b). They require at least low light levels, and reach greater abundances in the absence of tree cover (Bellinzoni *et al*. 2003). Of the two broad groups, cyanobacteria are usually considered more stress-tolerant and able to colonize drier locations than green algae (Bellinzoni *et al*. 2003). Identification of taxa in these groups is difficult, but genetic analyses reveal greater biodiveristy in algal colonies than predicted by morphological traits alone (Rindi 2007): around 100 species each of green algae and cyanobacteria have been reported from urban areas. Some species show substrate specificity, with some growing preferentially on mortar and some on bricks or stones. Of the hard surface habitats mentioned above, wall bases appear to host the most productive growth of algal/cyanobacterial assemblages, likely due to greater moisture and nitrogen availability, with areas frequented by dogs associated with nitrophilous species (Rindi 2007).

2.3.3.2 Lichens

Hard surfaces are typically the most important habitats for lichens in urban environments (Seaward 1979). Lichens that typically colonize natural rock surfaces usually find urban stone surfaces to be suitable habitat. They respond to the environmental differences in various microhabitat features of walls, as do the other components of the vegetation, with different species showing varying tolerance of drought, exposure, and nutrient enrichment. Lichens such as *Xanthoria* spp. are particularly noted for the association with bird perches (Darlington 1981). Most hard surfaces can be colonized by crustose lichens, typically the most stress-tolerant morphological group, but the more ‘three-dimensional’ forms, foliose and fruticose lichens, are often limited by pollution in urban areas (Darlington 1981).

2.3.3.3 Bryophytes

Mosses are common on both horizontal and vertical hard surfaces and are often the first to colonize new sites (Darlington 1981). Some mosses are restricted to rock surfaces, others restricted to areas where weathering has resulted in fine substrates, while some are equally abundant on either (Guggenheim 1992). Acrocarpous (erect growth forms) mosses tend to be more stress-tolerant than pleurocarpous (prostrate growth forms), and on stone walls there are common patterns of zonation related to exposure to the sun and drying conditions (Darlington 1981). Some acrocarpous species form cushions which appear to store water and remain wet long after the surrounding hard surfaces have dried out (Woodell 1979). Some species are adapted to high nutrient conditions and are associated with wall bases and pavements. *Bryum argenteum* is considered the quintessential urban moss, tolerant to trampling, nitrophilous, and xerophytic (with silvery leaf hairs that likely increase reflection of solar radiation) and has a nearly global distribution (Kimmerer 2003). Submerged river walls host the common aquatic moss *Fontinalis antipyretica* (Rishbeth 1948). Liverworts are rare on hard surfaces, only occurring in locations with continuous water availability (Darlington 1981).

2.3.3.4 Ferns

Of the vascular plants, ferns are perhaps the most typical of walls, especially those species originating on natural cliffs (Woodell 1979). The genus *Asplenium* includes many species found on walls throughout Europe, including some such as *A. ruta-muraria*, 'wall-rue', which is considered almost restricted to stone walls (Darlington 1981). Ferns require ample moisture especially at juvenile (gametophyte) life stages, but some wall ferns have drought avoidant traits in their sporophyte stage, such as leaf drying and rolling up, with rapid reconstitution upon increases in moisture. Ferns are rarely found on horizontal hard surface habitats.

2.3.3.5 Gymnosperms

Conifer trees and shrubs are uncommon elements of wall vegetation. These trees often grow in a stunted condition (Woodell 1979), resulting in similar growth forms to those on natural cliff habitats (Larson *et al*. 2000) (Fig. 2.3.2c).

2.3.3.6 Angiosperms

Hundreds of species of flowering plants form the dominant vegetation on hard surfaces in many parts of the world. Most are at least somewhat tolerant of drought, but the dominant life forms present are determined apparently by climatic factors (Brandes 1992). Plants of all groups are likely to show adaptations against drought: succulent habit, leaf hairs, or inrolled leaves (Woodell 1979). European wall angiosperms tend to be also drawn from species adapted to high calcium and high nitrogen environments (Darlington 1981). Hemicryptophytes (perennial plants with buds located at or near the substrate surface) dominate wall and pavement vegetation in Atlantic Europe (Rishbeth 1948; Woodell & Rossiter 1959) and central Europe (Duchoslav 2002), chamaephytes (buds located above the soil but not above 25 cm) dominate in Mediterranean Europe (Brandes 1992), therophytes (annuals) in India (Varshney 1971), and phanerophytes (trees and shrubs) in Israel. There are exceptions to these patterns as well: Greek and Bulgarian walls support mostly annuals, with hemicrypophytes the second most abundant group (Krigas *et al*. 1999; Pavlova & Tonkov 2005). Annuals are also common on walls in Israel (Weinstein & Karschon 1977), likely due to the high species richness of annuals in desert environments. Annuals are also common on pavements (Segal 1969). The annual life strategy is likely important for drought avoidance. Within the wall flora, there are well-known patterns of abundance in specific microsite features, with some species showing strong preferences for wall tops, wall faces, and wall bases (Darlington 1981).

Like other plants, trees are greatly influenced by the availability of substrate for rooting, with greater numbers of trees in areas with longer joints between stones (and larger stones) and greater substrate amounts (Jim 2008). Although *Betula pendula*, *Acer pseudoplatanus*, and others can successfully colonize walls (Kent 1961; Trocha *et al*. 2007), angiosperm trees are uncommon on walls in Europe. In Hong Kong, trees from humid-tropical biomes colonize stone retaining walls, where soil is available behind the wall, allowing some trees to grow to large sizes. Most of these species are native trees and are dominated by pioneer/ruderal life strategies (Jim 2008). Some have 'strangler fig' habits, allowing them to colonize precarious vertical surfaces (Fig. 2.3.2d), with strangler figs also occurring on Brazilian city walls (dos Reis *et al*. 2006). While mycorrhizal associations are important in most angiosperm groups, the only study of these mutualisms in artificial hard surface environments was done on *Betula pendula* communities in Poland (Trocha *et al*. 2007). Mycorhizal communities similar to those in early successional horizontal habitats were detected, and may play an important role in the nutrition of wall trees.

2.3.4 Colonization and dynamics

The vertical nature of walls presents challenges for potential colonists: gravity easily removes substrate and seeds, hence botanists have been intrigued by the dispersal mechanisms of the wall flora. Some authors speculate that as an interruption of horizontal air flow, walls actually receive more propagules than horizontal surfaces, but less are actually retained (Darlington 1981). It is thought that cyanobacteria and green algae are not limited by dispersal to hard surfaces: rainwater contains green algal propagules, and wind transports bacterial and fungal spores (Rindi 2007). Water flow patterns on the surface, however, result in the relocation of propagules and cause spatial patterns such as vertical stains, where cyanobacterial colonies track drips from building roofs down the walls (Rindi 2007). Plant colonization of vertical wall surfaces is dominated by wind dispersed propagules (Kent 1961), whereas wall tops are often colonized by species whose seeds are transported on animals (Duchoslav 2002). Some wall species also spread vegetatively, using rhizomes or stolons (Lisci & Pacini 1993). Pavement vegetation is thought to be dispersed in part by trampling (*Bryum argenteum* fragments can grow new plants), and some seeds can be transported long distances on car tires (Darlington 1981). River walls are often dominated by plants that use water as a vector (hydrochory) (Francis & Hoggart 2008).

Wall vegetation is similar to early-successional communities on level ground habitats (Darlington 1981), but harsh conditions that preclude the development of soil result in long-term persistence of this stage. Individual plants, once established, can be highly persistent, with reports of the same populations or even individuals occupying the same sites for decades (Kent 1961). Pavement vegetation is also maintained in a perpetually early-successional state due to trampling and barriers to soil accumulation. Rubble habitats, conversely, can undergo rapid change from open to closed vegetation (in a few decades) (Sukopp *et al*. 1979) due to the ease at which organic material accumulates. Since wall construction techniques have changed over the last two hundred years, many species that were formerly common have become rare (Kent 1961), and the prevalence of modern construction materials such as glass and metal cladding may render spontaneous wall vegetation a relic of earlier phases of human technological culture. As a response, there are groups devoted to the preservation, restoration, and new construction of dry stone walls in the UK and the recognition of the unique character of the wall flora appears to be a key motivator (Payne 1989).

2.3.5 Origin of hard surface floras

In Europe, where the majority of ecological surveys of hard surface environments have been carried out, many of the typical species found there are native species. Botanists have thus asked where these native species lived before the relatively recent development (in evolutionary time) of artificial walls and pavements. Most accounts of wall flora are quick to point out the obvious ecological similarities between stone walls and natural cliffs. Many distinctive wall plants originate in rock outcrop habitats (Brandes 1992). Since rock cliffs are rare components of most natural landscapes (Larson *et al*. 2000), the proliferation of walls in cities represents a significant expansion of habitat for rock specialist species (Woodell 1979). Hard surfaces, whether artificial or natural, are clearly stressful environments compared with the forest habitats considered to be the dominant vegetation in most temperate biomes: lack of substrate leads to a risk of desiccation (but see Larson *et al*. (2000)), or otherwise limits productivity by constraining the maximum size of plants able to root in fissures and cracks. The analogy between walls and pavements and natural vertical and horizontal hard surface habitats works at a coarse level, but many authors have identified important differences. Urban pavements are exposed to trampling (Woodell 1979) and support a species-poor flora of trampling-tolerant taxa, several of which are considered to have no native habitats. These species have likely arisen due to novel selection pressures in the last several millenia as anthropogenic pavements became more common (e.g. *Plantago major*, *Poa annua*, and *Polygonum aviculare*) (Scholz 1996). Natural pavements tend to be species-rich (Schaefer and Larson 1997) and untrampled, but do resemble urban pavements and shallow soil rubble areas where trampling is absent (Fig. 2.3.2).

Many of the lichens, mosses, and vascular plants colonizing walls are considered rock specialists, but most authors recognize a more generalist, opportunistic component of wall vegetation (Woodell & Rossiter 1959), featuring species that occur in many different habitats. The mix of rock specialists, species with strong affinities to other habitats (such as halophytes in seaside areas), and generalists depends on the region, but most studies report rock specialists from around 10 per cent (Woodell & Rossiter 1959), to 15 per cent (Weinstein & Karschon 1977), to 20 per cent (Karschon & Weinstein 1985). Additionally, specialists on other conditions can occur on walls or pavements. For example, the natural habitat for *Bryum argenteum* is areas enriched in nitrogen, such as rocky ledges in seabird colonies (Kimmerer 2003), and analogous conditions are provided in pavement cracks and wall bases.

Opportunistic species also colonize from nearby habitats, thus propagule availability plays a large role in determining the ultimate character of wall vegetation (Duchoslav 2002). Also, walls themselves have key differences from natural rock outcrops: the presence of mortar or other binding materials, isolation from other habitats, and disturbances such as intentional cleaning can result in a different set of physical conditions compared to those in natural habitats (Segal 1969; Duchoslav 2002; dos Reis *et al*. 2006). Outside Europe, wall floras may contain very few rock specialists, as in Brazil where most wall species are agricultural weeds (dos Reis *et al*. 2006). Hong Kong wall trees are mostly native species that tend to be early successional ruderals, not conservative stress-tolerant specialists (Jim 2008).

Escudero (1996) argues that an emphasis on the unique, specialist components of natural rock habitats has lead to oversight of the abundance of generalist native species found on cliffs. Thus, like natural cliffs, walls contain a mix of rock specialist and generalist, often ruderal species. Many typical wall species are not native, but have originated in other regions (Sukopp *et al*. 1979; Gilbert 1992), nevertheless, they often have rock outcrop origins in their native regions. Some of these are garden escapes, pointing to the role of local proximity of seed sources as a key determinant of hard surface vegetation composition. In Europe, many of these non-natives originate in warmer regions and exploit the urban heat island or warmer microsites created by south facing walls (Woodell & Rossiter 1959; Sukopp *et al*. 1979; Láníková & Lodosová 2009). Other species found on walls are identified with 'waste places' (Woodell & Rossiter 1959; Weinstein & Karschon 1977; Sukopp *et al*. 1979; Scholz 1996). While many consider these and other anthropogenic habitats, such as chalk grasslands in Britain, to be wholly novel, outside a small list of species with no natural habitats (Scholz 1996), most of these species existed long prior to anthropogenic land clearing, and thus required persistent open habitats (Grubb 1976; Marks 1983; Wittig 2004). These open habitats included cliffs, cobble beaches, dunes, and other areas permanently kept free of forest, and these habitats are recognized as the origins of much of the urban flora, not just hard surface areas (Wittig 2004). Thus many species not currently considered to be restricted to rocky habitats are still likely to have had their origins in a small subset of permanently open original habitats. Whether urban hard surfaces thus represent an analogue of a set of rare natural habitats or a novel habitat is a matter of perspective: the presence of anectophytes (species that evolved in anthropogenic environments), for example the green alga *Trentepohlia iolithus* is only known from building surfaces in Europe (Rindi 2007), suggests that ecological novelty results in new selective pressures and rapid speciation, whereas the frequent recurrence across entire regions of conservative native rock specialist plants on stone walls points to ecological similarities with natural rock outcrops.

2.3.6 Theoretical frameworks

Ecologists have classified plant species into general types representing strategies for coping with different components of a habitat templet (Taylor *et al*. 1990). One of the most widely used is Grime's C–S–R scheme which proposes three distinct axes along which species are exposed to selective pressure: disturbance (frequency or magnitude of biomass removal), stress (resource limitations or physical factors that reduce growth rates), and competition (greater standing crop due to absence of stress and disturbance) (Grime 1977). Wall and pavement vegetation is usually classed as stress-tolerant, due to limits to soil accumulation, nutrients, and water, but when species from individual floras are classified intermediate strategies emerge as the most common (Lisci 1997), and in some cases competitive strategies, usually associated with the most productive habitats, dominate (Duchoslav 2002). This suggests that the wall habitat is not monolithically a stressful environment; many of the microhabitat features described in the above sections creates heterogeneity that can be exploited by species adapted to quite different sets of conditions.

A simpler classification has been proposed as an alternative to Grime's triangle and may explain some of the life history strategy patterns in wall vegetation. Taylor *et al*. (1990) propose two axes, one of carrying capacity (maximum biomass or fertility of the habitat) and one representing the distance between carrying capacity (maximum biomass) and actual biomass, with a gradient in biomass-removing disturbance producing the difference. This scheme proposes that competition can be important in stressful environments too, thus traits such as tolerance of low moisture availability may be associated with competitive ability in these habitats. Also, pavement vegetation occurs in relatively low fertility and high disturbance environments, a section of habitat templet space considered to be uncolonizable by plants in the C–S–R scheme. Regardless, both schemes are based on the premise that the plant species most suited to local microsite conditions will form the vegetation.

The idea that urban hard surfaces attract a similar flora is borne out by the many descriptive studies across Europe and the Near East. Species are consistently found in the same types of microsites across whole regions (Woodell 1979), and many species are found on walls or pavements even in different regions or continents (Weinstein & Karschon 1977; Krigas *et al*. 1999; Brandes 2001; Lundholm & Marlin 2006; Davies & Rushton 2008; Jim 2008). When wall vegetation differs from that in surrounding level-ground habitats, it is often due to the greater abundance of rock specialist species on the walls (Weinstein & Karschon 1977; Krigas *et al*. 1999; Pavlova & Tonkov 2005). We can thus argue that urban hard surfaces present a consistent set of habitat templets regardless of where they are found.

On the other hand, many studies show high turnover in species composition between sites, with a strong influence of local level-ground communities, for example, with more agricultural weeds on walls outside of town (Weinstein & Karschon 1977), or high abundance of sea cliff plants in seaside towns (Segal 1969). Some of these species may be ephemeral in these habitats, colonizing, but not able to form self-sustaining populations. Ecologists call these colonization events 'mass-effects', which result in a flora with wall specialists and other species that are just there by virtue of abundant propagule production from other nearby habitats (Shmida & Ellner 1984) (Duchoslav 2002). Likewise, dispersal limitations of rock specialist plants may result in the 'right' species not reaching appropriate microsites. Typically, pavements have closer matches with surrounding habitats than do walls (Pavlova & Tonkov 2005), perhaps because more propagules can reach pavements or greater resource availability can support more species from the local flora than can walls. In other cases, walls contain many species not commonly found in other habitats in the local area, with many of these being rock outcrop specialists (Krigas *et al*. 1999; Pavlova & Tonkov 2005), thus allowing the walls to act as refugia for species not suited to local level-ground habitats. It is clear that both dispersal from surrounding areas of generalists and ruderals, and the provision of distinct niches for specialist species that might be locally rare, are features of urban hard surfaces. These habitats represent an opportunity to study deterministic niche-based processes and stochastic dispersal processes as contributors to community structure but, to date, little experimental work of this nature has been conducted in cities.

2.3.7 Problems caused by vegetation on hard surfaces

Plants can damage stone walls over time, but is it generally supposed that physical weathering must be well advanced before plants and lichens can appreciably affect wall structure (Seaward 1979; Woodell 1979). The mechanisms that increase available substrate for plants are the same as those that degrade the physical structure of the wall: acids produced by roots, physical penetration and expansion of crevices, and creation of habitat for microorganisms and invertebrates that can enter buildings (Woodell 1979). Woody plants are typically thought to be the greatest threat to stone or brick structures, and also increase the risk of fire (Lisci & Pacini 1995), but the first vegetated roofs in urban Germany were constructed to reduce the risk of fire to wooden buildings (Köhler 2006). Additionally, biofilms made by cyanobacteria and other organisms can increase the accumulation of atmospheric pollutants on the hard surface, while other microorganisms colonize the rock or bricks themselves as endoliths (Crispim & Gaylarde 2005). Vegetation can pose specific challenges on river walls: maintenance is frequently required in these systems and plants make access to the wall surface more difficult. The increased weight of the vegetation can make walls unstable, and the vegetation can encroach upon the water channel itself, causing problems with flow patterns and/or navigation (Francis & Hoggart 2008).

On horizontal pavements, the most common problems reported are 'conflicts' between urban tree roots and the paved surfaces, with roots forcing up or cracking paving (Randrup *et al*. 2001). In oceanic climates, such as on the west coast of the US and Canada, mosses are considered a nuisance when they colonize hard surfaces such as roofs (Kimmerer 2003). In general, there are often complaints about the visual appearance of spontaneous vegetation in cities (Woodell 1979). In places of architectural interest, plants can obscure buildings (Lisci & Pacini 1995), and algae are often cleaned off surfaces to enhance visual appeal (Rindi 2007). It is telling that the majority of ecological studies of pavement vegetation are tests of weed control methods (Rask & Kristoffersen 2007)! Much like their animal counterparts, the rat, the cockroach, and the pigeon (Larson *et al*. 2004), plants attracted to the hard surface habitats that mimic their habitats of origin are not always welcome on structures built to fulfil human purposes.

2.3.8 Benefits of hard surface vegetation

While botanists and plant sociologists have approached the wall and pavement floras largely as subjects of interest in their own right, there are other benefits that might appeal to a broader constituency. The ability for walls to support rare taxa is largely only reported from Europe, nevertheless 9 red listed species grace the walls of Zürich (Guggenheim 1992), and 20 the walls in the German province Niedersachen (Brandes 1987). In Italy, walls and ruins support conservative rock specialists and provide corridors for these species to gain access to urban areas (Celesti-Grapow & Blasi 1998). Many European plant species are more abundant on walls than their original natural rock outcrop sites (Gilbert 1992), and with changes in construction techniques (such as the development of weather-resistant mortars) leading to reduced opportunities for vegetation development, removal of vegetation from walls, and urban re-development, some wall plant communities are considered endangered (Werner *et al*. 1989).

While it is acknowledged that spontaneous vegetation on hard surfaces provides visual relief in urban areas (Woodell 1979), there have long been efforts to intentionally plant stone walls (Darlington 1981). Such initiatives must create a balance between the benefits and liabilities of plants and other vegetation on walls, and it is this quest that has grounded the development of green façade and wall technology, beginning in central Europe over 100 years ago (Köhler 2008). Green facades in particular, avoid the problem of plants damaging building structures by the construction of supports for plant growth separate from the wall, and provide several benefits, including reduction of summer temperatures and the trapping of particulates in urban air. River walls can be used to conserve plant species, but altered to enhance the habitat by adding organic materials such as coir bundles and by constructing more permanent structures to hold rooting substrate (Francis & Hoggart 2008). Green roofs mimic shallow substrate rock outcrop or meadow habitats and provide many benefits as urban ecosystems, including biological conservation, reductions in energy consumption, stormwater management, and air pollution mitigation (Oberndorfer *et al*. 2007). Many of the commonly used plants for green roofs are also frequently found spontaneously on walls (e.g. Pavlova & Tonkov 2005; Köhler 2006) and natural rock habitats (Lundholm 2006). A variant of this approach involves the re-creation of rubble fields on rooftops to support a variety of rare species that are being lost as British cities are redeveloped (Grant 2006).

Studies of hard surface vegetation are overwhelmingly concentrated in Europe, and despite the number of such studies, most of these are descriptive in nature. Given the growing urbanization of the planet, these habitats will increase in area worldwide, and the differences in construction techniques between old European stone surfaces and modern surfaces require further study. Walls and pavements represent a useful complement to studies of more productive urban habitats. While natural rock outcrops are distinct features of the landscape, and represent extreme spatial discontinuities and steep environmental gradients, urban hard surfaces are one of the dominant land covers in cities. There is a large potential for scientific study of these surfaces, not only because of their potential to provide significant ecosystem services, but also to further our understanding of the processes controlling the composition of communities and the maintenance of species diversity.

Acknowledgements

I thank Dr Norbert Müller of the University of Applied Sciences in Erfurt for introducing this North American to the anectophyte concept. This chapter is dedicated to Dr Doug Larson of the University of Guelph, who inspired me to continue down the hard surface path.

CHAPTER 2.4

Composition and Diversity of Urban Vegetation

Christopher P. Dunn and Liam Heneghan

2.4.1 Introduction

Urban ecology has emerged from a long period of neglect to become an important disciplinary focus. In the USA, this neglect has been attributed to a reluctance by ecologists to study 'unnatural' systems in cities, or those that have otherwise been subject to extensive human impacts (McDonnell *et al*. 1997). Such hesitancy has not been as evident in other parts of the world. Fortunately, urban ecology (broadly) and urban vegetation ecology (more narrowly) are becoming important research topics in many parts of the world.

In recent years, urban studies in the USA have benefited from the creation of two urban Long Term Ecology Research (LTER) sites, Baltimore and Phoenix, funded by the US National Science Foundation (NSF). Along with this reorientation in thinking about cities is a growing appreciation of the role that urban and suburban areas play in preserving biodiversity, as is recognition that practices such as restoration and phytoremediation in cities may complement more traditional conservation strategies including restoration and management of nature preserves (Dobson *et al*. 1997).

Considerations of urban vegetation from an ecological viewpoint have been relatively recent, with interest rapidly growing since the 1970s. Much of the early interest was by landscape architects and urban planners (e.g. McHarg 1963) who were intent on blending function with form and aesthetics. In many respects, the vegetation in cities is a most prominent and publicly-valued feature (aesthetically, physically, and politically) of urban areas, second only to the built environment. Many of the world's cities have tree ordinances, celebrate Arbor Day, and support parks departments and other public agencies focused on managing and enhancing their aesthetics and sustainability. Furthermore, many homeowners spend considerable time and money on gardens (gardening being one of the most popular leisure activities in many countries).

Vegetation can be as important as the built environment in defining the character of a city. For most urban residents, vegetation is part of the backdrop, rather than an integral and dynamic component of the infrastructure that is just as deserving, or in need, of maintenance and restoration as roads and buildings. However, as residents gain a greater appreciation of urban vegetation, it is likely that they progress from an awareness of the plants around them (native and non-native) before considering how composition or structure might change from place to place within the city, or from the city to the countryside. Eventually, we would hope, residents would take a deeper interest and begin to appreciate the obvious dynamics (seasonal to long-term) of vegetation and become involved, as volunteers perhaps, in community-based restoration efforts.

It is largely this progression that we follow here; namely, from the broad view of city floristics to considerations of more complex issues of differences in forest structure and dynamics along urban–rural gradients, to effects of invasive plants on restoration efforts, among others. Although much of the research has been based in North America and Europe, we include as much literature from Asia and the tropics as space allows.

2.4.2 Urban floristics

2.4.2.1 Native and non-native flora

The rather complex nomenclature used by many European botanists to classify plants by their time of introduction, mode of introduction, and extent of naturalization makes comparisons among world regions challenging. Epoecophytes, for instance, are those plants naturalized in man-made and disturbed habitats. Hemiagrophytes, by contrast, are those that have naturalized primarily in semi-natural or disturbed habitats. With respect to time of introduction, archaeophytes and neophytes are defined as non-native species arriving in Central Europe prior to 1500 CE and since, respectively. To further refine 'neophyte,' kenophytes are those plants introduced between the sixteenth century and the end of the nineteenth century. A review of this nomenclature is provided by Mosyakin and Yavorska (2003) and references therein.

Putting aside for the moment definitions of native and non-native and debates over time of introduction and immigration, it is evident that much of the early interest in the plants of urban areas was expressed in floristic inventories. In his excellent review, Sukopp (2002) documents urban floras from London and Paris dating from the early seventeenth century. Schouw (1823, as cited in Sukopp 2002) was apparently the first to use the term (albeit in German) 'urban plants' and to describe many of them as being of foreign origin.

Interest in origin (and time of introduction) of non-natives continues to this day. Such species are of interest and relevance for two, related, reasons. Firstly, non-natives can be used as indicators of human impact on the landscape (whether urban or not; see Cilliers and Siebert, Chapter 3.2). And secondly, cities can be considered as sinks for non-natives and then, once such species are established, as sources for other areas within and outside the city.

Many authors (including Pyšek 1998; Sukopp 2002; La Sorte *et al*. 2008) have documented the flora of European cities and towns, noting the abundance of native and non-native species. A review of the literature suggests that the flora of most cities consists of around 30 to 50 per cent non-native plants (Table 2.4.1). This holds true for most cities in Europe and the USA. The central European cities studied by Pyšek (1998), for example, averaged 260 non-native plant species (range 94 to 748), representing 40 per cent of the total urban flora.

Kyiv (Ukraine) is shown to support 536 non-native plant species (Mosyakin & Yavorska 2003). Although Kyiv is far larger (824 km^2) than the largest city in Pyšek's (1998) study (Berlin at 480 km^2), it is roughly in the mid-range of total non-native species for other cities cited in Pyšek (1998). Mosyakin and Yavorska (2003) do not provide data on numbers of native taxa. However, they do note that just three plant families (Asteraceae, Poaceae, and Brassicaceae) account for nearly 40 per cent of all non-native species in the urban flora. In a survey of European cities, Ricotta *et al*. (2009) show that this restricted phylogenetic diversity of non-native floras is widespread and is a result, in part, of the unique environmental filters of urban areas.

Interestingly, the largest group of non-native taxa that has become established in Kyiv during the twentieth century is North American (36 per cent of all recent introductions). In fact, one major railroad right-of-way in Kyiv is so dominated by North American plant taxa (*Acer negundo*, *Prunus serotina*, *Solidago* spp., among many others; S.L. Mosyakin, pers. comm. and C.P. Dunn, pers. observ.) that its nickname among local botanists is 'American Canyon'.

In one of the few studies in China, Hu *et al*. (1995) describe briefly the nature of the urban vegetation in Tianjin and its successional development. They note that of the 1,049 species encountered, 38 per cent are non-native. This percentage is consistent with most other cities (Table 2.4.1).

An interesting exception is the urban flora of five Italian cities (Celesti-Grabow & Blasi 1998) in which non-natives account for only 12–26 per cent of the urban flora (Table 2.4.1). The highest proportion of non-natives (26 per cent) occurs in Milan. As in Kyiv, Milan's urban flora contains a considerable presence of North American species. In general, the urban flora of the Italian cities is more similar to its regional flora than is the case in other parts of the world. This lesser representation of non-native species could be attributable to a number of factors, as suggested by Celesti-Grabow and Blasi (1998). First, there is a high level of connectivity between the

Table 2.4.1 Floristic and physical attributes of urban areas. Within a country, cities are listed in order of increasing size. Data attributed to Pyšek (1998) are from sources cited therein. Sizes of Italian cities were not provided in Celesti-Grapow and Blasi (1998) and were obtained from various online resources. City sizes provided in Clemants and Moore (2003) and Hu *et al*. (1995) include surrounding metropolitan areas, not just city per se; thus, sizes are considerably higher than those cited in other studies. * Guangzhou, China data are urban trees only

City	Country	Native	Alien(%)	Total	Area (km^2)	Popn. (000s)	Reference
Dublin	Ireland	157	158 (50)	315	n/a	506	La Sorte *et al*. (2008)
Exeter	UK	260	228 (47)	488	n/a	118	La Sorte *et al*. (2008)
Edinburgh	UK	94	239 (72)	333	n/a	449	La Sorte *et al*. (2008)
Leeds	UK	241	180 (43)	421	n/a	715	La Sorte *et al*. (2008)
Birmingham	UK	332	246 (43)	578	n/a	977	La Sorte *et al*. (2008)
London	UK	498	673 (57)	1,171	n/a	7,172	La Sorte *et al*. (2008)
Vienna	Austria	728	748 (51)	1,476	414	1,600	Pyšek (1998)
Arnsberg	Germany	446	167(27)	613	n/a	74	Pyšek (1998)
Neumünster	Germany	352	129(27)	481	72	83	Pyšek (1998)
Frankfurt	Germany	456	389(46)	845	248	645	Pyšek (1998)
Köln	Germany	571	376(40)	947	400	970	Pyšek (1998)
Berlin (W)	Germany	841	577(41)	1,418	480	1,900	Pyšek (1998)
Kostelec	Czech Rep.	234	103(31)	337	14	0.005	Pyšek (1998)
Brno	Czech Rep.	335	429(56)	764	200	344	Pyšek (1998)
Jaroslaw	Poland	166	165(50)	331	40	30	Pyšek (1998)
Tarnow	Poland	376	159(30)	535	72	119	Pyšek (1998)
Poznan	Poland	700	200(22)	900	115	600	Pyšek (1998)
Warsaw	Poland	766	343(31)	1,109	430	1,600	Pyšek (1998)
Kyiv	Ukraine	n/a	536(n/a)	n/a	824	2,600	Mosyakin & Yavorska (2003)
Ancona	Italy	222	30(12)	252	125	103	Celesti-Grapow & Blasi (1998)
Cagliari	Italy	230	31(12)	261	86	218	Celesti-Grapow & Blasi (1998)
Palermo	Italy	244	39(14)	273	160	734	Celesti-Grapow & Blasi (1998)
Milan	Italy	160	55(26)	215	182	1,400	Celesti-Grapow & Blasi (1998)
Rome	Italy	326	46(12)	372	1285	2,800	Celesti-Grapow & Blasi (1998)
Christchurch	New Zealand	106	380(78)	486	272	n/a	Stewart *et al*. (2009)
New York	USA	1,649	881(35)	2,530	10 909	17 500	Clemants & Moore (2003)
Philadelphia	USA	1,612	922(36)	2,534	6,687	3,800	Clemants & Moore (2003)
Chicago	USA	1,176	577(33)	1,753	5,763	7,400	Clemants & Moore (2003)
Detroit	USA	1,121	495(31)	1,616	5,094	4,000	Clemants & Moore (2003)
Boston	USA	1,252	1,054(46)	2,306	4,617	3,500	Clemants & Moore (2003)
Twin Cities	USA	1,131	270(19)	1,401	4,421	2,300	Clemants & Moore (2003)
Washington, DC	USA	1,561	813 (34)	2,374	3,755	4,600	Clemants & Moore (2003)
St. Louis	USA	1,352	404(23)	1,756	3,196	1,600	Clemants & Moore (2003)
Guangzhou*	China	132	122(48)	254	56.9	2,130	Jim & Liu (2001)
Tianjin	China	656	393 (38)	1,049	7,400	11 150	Hu *et al*. (1995)

countryside and cities and thus a proximity to seed sources of native species. Furthermore, many native Mediterranean plant species can tolerate arid conditions and are therefore well-adapted to the conditions within cities, such as limited water supply, high temperature, and high solar input.

All floristic compilations have gaps and other limitations, such as thoroughness of the inventories, taxonomic questions (species vs. subspecies, etc.), inclusion or not of cultivars, time since last inventory, and reliance on previous inventories. For example, for the flora of West Berlin, Pyšek (1998) uses a 1988 data source, listing 841 native plants and 577 non-natives, for a total flora of 1,418 species. For the same city, La Sorte *et al*. (2008) rely on an older study from 1974 which lists 414 natives and 561 non-natives, for a total flora of 975 species. Nonetheless, regardless of city or source of data, the proportion of non-natives is remarkably consistent over large geographic regions.

2.4.3 Does size matter? Cities and vegetation patches as habitat islands

Since the publication of MacArthur and Wilson's (1967) groundbreaking work in island biogeography theory (IBT), many attempts have been made to test the theory using habitat patches as well as cities in place of oceanic islands. Even if patterns supportive of IBT emerge, very little added ecological understanding of the ecology of cities and urban vegetation is revealed.

2.4.3.1 Species richness and city size

Pyšek (1998) has attempted to correlate the numbers of non-native and native species to various attributes (area, population, density, altitude, temperature, etc.) of 54 central European cities. The results were mixed and confounded by attempts to distinguish between archaeophytes and neophytes. The total species number (native plus non-native) was positively correlated with city size, whether expressed as land area or as human population. In addition, neophytes (both total number and percentage of total flora) increased with city size.

Population size of 22 European cities appears weakly related to total number of species, but unrelated to numbers of archaeophytes, neophytes, and non-natives when considered separately (Table 1 in La Sorte *et al.* 2008). With dates of floristic surveys referred to by La Sorte *et al.* (2008) ranging from 1971 to 2003, it is not surprising that correlations are problematic. Similarly, Arroyo *et al.* (2000) conclude that number of alien plants is significantly correlated with degree of urbanization of Chilean cities or political regions rather than with city or population size. Neither study provides much in the way of explanatory analysis other than to postulate that cities provide more suitable habitats for non-native species and, presumably, larger cities have more such habitat.

In eight large north-eastern US cities described by Clemants and Moore (2003), the proportion of non-native species is very similar to those provided in Pyšek (1998); however, this proportion was correlated significantly with longitude, not with city size. Clemants and Moore (2003) speculate that longitude is a surrogate for time since introduction of non-native plants, with cities on the East Coast (e.g. Boston and New York City) having been subject to introduction of non-native plants earlier than those further inland and further from seaports (e.g. Detroit, Chicago). This is a reasonable assumption. Nevertheless, other potential confounding factors, not considered by Clemants and Moore (2003), include thoroughness of the floras consulted, years since the floras were last updated, and length of time that floristic studies have been conducted. Thus, time, rather than longitude per se is likely a main factor accounting for non-native species presence.

2.4.3.2 Species richness and patch size

Major components of urban vegetation exist as patches of habitat (including vacant lots) and as remnants of natural vegetation that, owing to fragmentation and urbanization (see Cilliers and Siebert, 3.2), have become isolated within an urban matrix. Here, too, attempts have been made to correlate species richness with some measure of area (typically, patch size).

An early North American examination of such a correlation is Crowe's (1979) inventory of 26 vacant urban lots in a section of Chicago, USA (Table 2.4.2). The results of Crowe (1979) are useful in that the lots were of known age (obtained from building demolition records), thus simulating the exposure of new habitat on oceanic islands to species immigration. Using conventional IBT, he found that flowering plant species richness generally followed IBT tenets; namely: (1) species richness in older lots appeared to have reached equilibrium, (2) the slope of the species–area curve is within the range predicted by IBT, and (3) equilibrial species richness appears to be lower in more isolated lots. What is missing here, however, is the European attention to modes and timing of immigration/introduction

Table 2.4.2 Attributes of remnant vegetation patches in some US urban areas. TVL = temperate vacant lot; TF = temperate forest

City	Region	Native	Alien (%)	Total spp (range)	Patch size, number, & type	Reference
Chicago	Northern USA	n/a	n/a	128 (9–69)	111–7,371 m^2 26 TVL	Crowe (1979)
Minneapolis	Northern USA	327	181 (36)	508	118 ha 30 TF	Hobbs (1988)
Worchester	Eastern USA	38	28 (42)	66	<1–200 ha 32 TF	Bertin *et al*. (2005)
Boston	Eastern USA	n/a	n/a	n/a	400 ha 1 TF	Drayton & Primack (1996)

and distinctions between native and non-native elements of the flora.

Hobbs (1988) attempted a more rigorous appraisal of the application of IBT to urban vegetation. She sampled 30 upland forest stands within the urban matrix of St Paul–Minneapolis ('Twin Cities') in Minnesota, USA (Table 2). Stands ranged in size from 1–8.6 ha and varied with respect to historic and current disturbance (primarily trampling and mowing). She concluded that total, native, and non-native richness of vascular plants were significantly and positively correlated with stand area.

These studies do not validate the application of IBT to patches of urban vegetation; one reason being that isolation (distance from a given urban patch to its nearest neighbour) is unrelated to species richness. Crowe (1979) showed a weak correlation between nearest neighbour distance and richness when considering only the older lots.

Remnant patches of vegetation that have become isolated owing to habitat destruction (agriculture, urbanization, etc.) in the vicinity of Milwaukee (Wisconsin, USA) do not support the species–area relationship (Dunn & Loehle 1988). All forest patches supported both native and non-native species and exhibited a rather pronounced 'edge effect'. Thus, the numbers of plant species, native or not, differed little among patches regardless of area.

2.4.3.3 Beyond description

What is generally missing from most descriptive floristic studies is a discussion of causality (admittedly challenging) and implications for management. With respect to the former, why should larger cities harbour more non-native species, assuming that the vast majority of the urban areas are unsuitable for plants? It would be interesting to have some sense of the number, type, size, and dispersion of suitable habitat in the cities as they possibly relate to presence of non-native species.

La Sorte *et al*. (2008) attempted to clarify some dynamics in the complex considerations of city size (population), species, and time. Their examination of ß-diversity (change in species composition, or 'turnover', across larger geographic scales) suggests that archaeophytes are not likely to further increase in range (having already reached equilibrium) and that it is neophytes that are rapidly expanding in response to urbanization. These more recent introductions are an important indicator of intense anthropogenic impacts on the landscape.

In a study of the flora (native and non-native) in Britain, Roy *et al*. (1999) use regression modelling to better understand the relationship between the degree of urbanization and local species pools. They show that about 50 per cent of plant species with a strongly positive relationship to urban land cover are not native to Britain and that 100 per cent of species with a strongly negative relationship are British natives. Although their analyses do not show urban areas to be richer overall than the surrounding countryside (in contrast to many continental studies), they do demonstrate that urban areas have a higher proportion of non-native species. This increased proportion of non-natives, as shown by their statistical analyses, is related in part simply to their increased number in cities. What is troubling, however, is that the increased proportion of non-native species also appears to be a result of local extinction of native species. Regression analyses predict various degrees of loss of native species, depending on which part of Britain is in question: there is a lower risk of loss as one goes north (Roy *et al*. 1999).

2.4.4 The planted cityscape

Vegetation is deliberately planted in residential and non-residential urban locations for aesthetic, ecological, and functional purposes. This vegetation can provide values such as screening of unsightly vistas, the abatement of noise, and the shading of city streets (McPherson *et al*. 1997). City trees may provide fuel (Hamabata 1980) and other products for local residents. The contribution of the urban forest to lowering the albedo of metropolitan surfaces can mitigate heat island effects (McPherson *et al*. 1997).

Indirectly, planted city vegetation may provide habitat for urban biodiversity, provide sequestration for urban atmospheric pollutants including carbon (Nowak & Crane 2002), and buffer sensitive riparian zones by reducing nutrient inputs into urban waterways (Groffman *et al*. 2002). The management of vegetation in urban habitats can result in net carbon sequestration in the soil (Pataki *et al*. 2006). In general, the significance of the urban forest as a carbon sink will depend upon the degree of management; this is particularly true when management requires the operation of equipment powered with fossil fuel (Nowak & Crane 2002).

2.4.4.1 Urban lawns and gardens

Other important elements of urban vegetation include lawns, gardens, and street trees. Each contributes in some way to ecosystem processes, land valuation, aesthetics, and sense of well-being. Lawns and gardens, in particular, have been the subject of rigorous and quantitative research.

Falk (1980) studied two lawns near Washington, DC, which differed in age and management. More intense management (fertilization and irrigation, in addition to mowing) resulted in lower plant species richness (11 vs. 22 species) and higher annual gross productivity than with mowing only. However, annual net primary productivity did not differ significantly, being 1,669 g m^{-2} under management and 1,649 g m^{-2} under minimal management. These values compare favourably with temperate (100–1,500 g m^{-2}) and tropical (200–2,000 g m^{-2}) grasslands and with crops such as maize (1,066 g m^{-2}) (Falk 1980).

Intensive management of urban lawns has been criticized widely as being environmentally unsound and wasteful of energy and water. Recently, Cheng *et al*. (2008) reported on the ecology of 28 Midwestern US lawns subject to one of three types of management: professional application of herbicides and fertilizers, homeowner applied, and no chemical input. Four grass species were common throughout, with a total of 40 weeds (mostly Eurasian) noted. Predictably, professionally managed lawns were judged to be more favourable aesthetically and had fewer weeds. Chemical management (professional and homeowner applied) had a detrimental impact on soil foodweb health and increased severity of rust infestation compared to no chemical inputs, leading to less sustainable systems.

In a study of urban lawns in Sheffield, UK, Thompson *et al*. (2004) examined the species composition and diversity of 52 private lawns (Table 2.4.3). Here, very little chemical treatment is used. A total of 159 species were inventoried. All of the 25 most common species were native. In fact, native

Table 2.4.3 Comparison of urban lawn attributes in Sheffield, UK (Thompson *et al*. 2004) and Christchurch, New Zealand (Stewart *et al*. 2009). Percentages in brackets are species numbers relative to total number of species

City	# Lawns	# Species	# spp in > 50%	in only 1 lawn	Aver no. natives	Of 25 most common spp.
Sheffield	52	159	14 (9%)	60 (38%)	94	all native
Christchurch	327	127	9 (7%)	49 (38%)	13	3 native

plant species ranged from 83 to 94 per cent across all lawns. The single most important factor accounting for species richness in these lawns was area. Distance to city edge (source of propagules) and location within the city (variation in local climate) were also related, in a lesser way, to species richness. Just as Falk (1980) suggested that urban lawns are similar to other grasslands with respect to productivity, so Thompson *et al*. (2004) show that species accumulation curves for lawns are similar to those for semi-natural grasslands.

By contrast, a study of lawns in Christchurch, New Zealand (Stewart *et al*. 2009) suggests that species richness is negatively related to lawn area. This result runs counter to many other studies of species–area relationships (and to those of Thompson *et al*. 2004), yet no hypotheses are offered in the way of explanation. The Christchurch lawns also differ from those in Sheffield in representation of native species (more natives in Sheffield than in Christchurch), but are remarkably similar in other measures, including the percentage of species occurring in more than 50 per cent of all lawns and the percentage occurring in only one lawn (Table 2.4.3).

The British fascination with gardens is evidenced by a continuing series of academic studies of home gardens in Sheffield, England, beginning with Thompson *et al*. (2003). These studies examine the various ways by which such gardens contribute to the broader biodiversity of Sheffield, 23 per cent of its land area being domestic gardens (Gaston *et al*. 2005). Thompson *et al*. (2003) inventoried 60 gardens in Sheffield ranging in age from 5–165 yr and from 32–940 m^2 in area. A total of 438 species were inventoried, 33 per cent of which were British native and 67 per cent non-native. Most of the 20 most common plant species were weeds, both native and non-native. With respect to broader issues of urban plant biodiversity, Thompson *et al*. (2003) suggest, firstly, that because many species are grown in low densities, they are likely to be less apparent to herbivores or pathogens. Secondly, gardens provide refugia for some widely planted native species that are uncommon in the wild.

The potential for encouraging homeowners to make gardens more suitable for increased urban biodiversity (first of plants, then of other organisms including wildlife) is high (Rosenzweig 2003). Gaston *et al*. (2005) report on a series of garden treatments designed to test the effectiveness of increasing biodiversity. Such treatments included placement of artificial nests for both solitary bees and bumblebees, of dead wood for fungi and other saprobes, of nettles (*Urtica dioica*) for butterfly larvae, and of small artificial ponds. Treatments, installed across 34 gardens in Sheffield, yielded mixed results; the most successful being nest sites for solitary bees and ponds. As Gaston *et al*. (2005) acknowledge, the low success rate could be due in part to the limited duration of the tests (three years). Yet, that is a timescale that might be unacceptably long for many gardeners. Consequently, there is some imperative to devising and testing treatments likely to have a considerably greater chance of success.

Loram *et al*. (2008) add to our understanding of urban gardens, relative to other greenspace, by comparing composition of gardens in five cities in the UK. A total of 1,051 plant taxa (29 per cent British natives and 71 per cent non-native) were identified in 267 sampled gardens. These percentages were almost identical across all five cities (Table 2.4.4). As might be expected, the majority of non-native species are of European and Asian origin, with lesser contributions from North and South America, Africa, and Australia/New Zealand. As with most other studies of urban flora in each city, the number of species in gardens was positively correlated to garden size. The most important factors accounting for striking similarities in species richness and composition among the five cities appear to be availability of plants, management of gardens, and socio-economic status of owners (Loram *et al*. 2008), rather than environmental factors related to local climate and environmental conditions.

Table 2.4.4 Species richness in five British domestic gardens. Modified from Loram *et al*. (2008).

City	Total species	Native (%)	Alien (%)
Belfast	479	152 (32)	327 (68)
Cardiff	633	183 (29)	450 (71)
Edinburgh	538	160 (30)	378 (70)
Leicester	639	186 (29)	454 (71)
Oxford	583	168 (29)	415 (71)

When examining the overall urban flora (Table 2.4.1), it is not clear if authors include domestic gardens in their inventories; however we assume not. Considering that gardens represent 18–27 per cent of the area of many UK cities (Loram *et al*. 2008), it is clear such gardens represent a huge number of species and contribute enormously to the floristic diversity and ecology of cities.

2.4.4.2 Street trees

Although there is considerable literature on the costs and benefits, ecological and otherwise, associated with urban forestry programmes and planting patterns within cities (e.g. Dorney *et al*. 1984), much research in the past several decades has concentrated on assessing the suitability of tree species for urban locations and evaluating the health of subsequent plantings (Dorney *et al*. 1984; Xiao & McPherson 2005).

In a Milwaukee (Wisconsin, USA) suburb, 1,110 individual trees (of 38 taxa) were inventoried along 4.1 km of transects which included street trees (planted and maintained by the municipality) and residential lots (Dorney *et al*. 1984). This suburb (Village of Shorewood) supported 42 per cent woody plant cover. Street tree plantings tended to have low species diversity (11 taxa) compared with front yards of residences (30 taxa), presumably reflecting a more economical approach on the part of the municipality to tree diversity and planting programmes. The density of plantings, however, was very similar along streets (25 per cent of total trees) and front yards (28 per cent of total). Backyards, the least visible component of the urban environment, supported the highest tree density (36 per cent of all trees). Backyards are not always as well-tended, with some of the density, therefore, being adventives. Jim and Liu (2001) report that roadside plantings in Guangzhou, China, are also poorer in species richness than other urban land features.

In many US cities, elms (*Ulmus* spp.) have been major elements of street plantings and of the urban forest more broadly (Dunn 2000), often having been planted as monocultures. As elms have been decimated by Dutch elm disease, replacements have tended to include hardy trees (e.g. Norway maple, *Acer platanoides*) that are tolerant of many urban conditions, but are less appealing and often weedy. Trees best suited for urban locations are generally those regarded as having appropriate aesthetic characteristics, and possessing sufficient hardiness to survive the peculiar environmental conditions associated with urban environments (Ware 1994).

In a number of cities impacted by Dutch elm disease, somewhat greater diversity of trees was re-planted along streets. However, monocultures of honey locust cultivars (*Gleditsia* sp.), ash (*Fraxinus* spp.), and other taxa have replaced the elm monoculture in many US cities. Thus, many cities are vulnerable to further outbreaks of devastating diseases and pests. The spatial configuration of the forest is an important consideration in the management of municipal urban forestry in the face of pest outbreaks. The aggregated spatial patterns of urban vegetation distribution can influence the spread of insect pests. For instance, expected damage in the US from Emerald Ash Borer, *Agrilus planipennis*, a pest native to Asia, will be facilitated by the prevalence and spatial patterns of ash plantings (MacFarlane & Meyer 2005).

Regardless of what might make sense economically, politically, or ecologically, the public tends to have strong preferences and opinions regarding urban tree plantings. In Downers Grove (a suburb of Chicago, USA), residents were more pleased with the benefits of street trees (aesthetics, feeling closer to nature, increased property values, etc.) than annoyed by other features (falling debris, leaf fall in autumn, nuisance insects, diseases). In addition, neighbourhoods with greater species diversity of street trees were rated higher than those with lower diversity (Schroeder & Ruffolo 1996). These attitudes are similar to those of residents of Sacramento (California, USA) for whom the benefits of aesthetics and other perceptions outweigh negative attributes of trees (Sommer & Summit 2000). Furthermore, Sommer and Summit (2000) found that positive perceptions by residents could be directed towards greater engagement in community tree planting and disease monitoring efforts, leading to stronger social ties.

2.4.5 Ecology of remnant vegetation in urban areas

A conspicuous feature of urban vegetation is that a considerable portion is both planned and planted (Baycan-Levent & Nijkamp 2009). This 'intentional vegetation' has numerous characteristics that differ from remnant habitat fragments, such that studies on planned vs. remnant vegetation represent discrete areas of research. However, in recent years, recognition of the value of urban vegetation, planted and remnant, for the provision of ecosystems services and for facilitating attainment of conservation goals has resulted in studies that do not discriminate between these vegetation types.

Williams *et al*. (2009) propose a conceptual model for predicting urban vegetation changes in which a series of filters change habitat availability; namely, the spatial arrangement of habitats, the pool of plant species, and evolutionary selection pressures on populations in the urban locations. Thus, the goals of sustainable management and of conservation of biological diversity could be facilitated.

2.4.5.1 Urban–rural gradients

One of the more productive paradigms for investigating ecological patterns in the urban environment is the use of urban–rural gradients (McDonnell & Pickett 1990; McDonnell *et al*. 1997). Several studies have examined forested vegetation from rural sites to those more proximate to urban centres (McDonnell & Hahs 2008; Paoletti 2009). In the New York City metropolitan area, forest remnants at the urban end of the gradient tend to have lower stem density, depauperate understory, and (as noted earlier) a significant representation of non-native species (McDonnell *et al*. 1997).

In their study of woody vegetation along a forest-to-urban gradient in Ohio (USA), Porter *et al*. (2001) examined six sites along a gradient of increasingly urban land-use. They, too, reported an increase in the representation of non-native woody plant species with increasing urbanization. Of the 57 woody species sampled, 17 (30 per cent) were non-native. Species richness and native species richness differed between habitat types, with richness generally decreasing as sites became more 'natural'. Species diversity peaked in a recreational area, and was lower in both more natural and highly urbanized environments.

By contrast, Guntenspergen and Levenson (1997) found few significant differences in species diversity and composition in sugar maple (*Acer saccharum*) dominated stands along an urban–rural gradient in south-eastern Wisconsin, USA. However, despite a clear human-caused disturbance gradient among stands, there was some increase in the shrub layer of 'opportunistic' species such as white ash (*Fraxinus americana*) and choke cherry (*Prunus virginiana*).

As Guntenspergen and Levenson (1997) observe (and as substantiated by other finer-scaled studies such as Zipperer 2002), the scale (or grain) at which observations are made could affect the ability to detect meaningful changes along such urban–rural gradients. The Wisconsin study was designed at a coarse scale, whereas others, including Zipperer (2002) were conceived at a finer scale.

In addition to gradients of disturbance type and intensity (trampling, cutting, mowing, etc.), there is a whole other suite of impacts from various types and sources of pollution that can be studied along urban–rural gradients. In a recent study, Paoletti (2009) examined the effects of ozone (O_3) along such gradients in three Italian cities. For several reasons, O_3 levels are higher in rural and suburban areas than in cities. Although vegetation can be impacted by O_3 as well as be responsible for some of its production via formation of biogenic hydrocarbons, the levels of O_3 are sufficiently high (based on European Union criteria) to impact forest health (Paoletti 2009).

In their review of 300 urban–rural gradient studies McDonnell and Hahs (2008) noted that in general the predictions of ecologists that organisms largely respond negatively to urbanization is too simplistic. They note that 40 per cent of the gradient studies focused on birds and insects, whereas only 14 per cent dealt with plants. Clearly, a great deal more can be, and needs to be, clarified about the relationship of human and urban impacts on the vegetation structure and dynamics of forest patches.

2.4.5.2 Vegetation dynamics in remnants

Dynamics of vegetation within patches (at any one point along the urban–rural gradient) have been examined extensively. The landscape (including urban) ecology literature dealing with vegetation is enormous. Earlier work in the US has been presented in Burgess and Sharpe (1981) and Dunn *et al*. (1991), among others. In an early study in Japan, Hamabata (1980) describes the effects of human use on 74 species in the herb layer of forest patches in suburban Tokyo. These remnants were classified along a continuum from current 'agricultural' use (firewood) to no agricultural use for a considerable time, with transitional stands in between. Using principal components analysis, Hamabata (1980) demonstrated a largely negative relationship between the extent of surrounding urbanization and the abundance of shrubs and the mean lifespan of herb layer species. Isolation of stands within an urban matrix appears unfavourable for long-term survival of typical forest flora.

Norway maple (*Acer platanoides*) has been a widely planted tree in the US since the demise of American elm (*Ulmus americana*) (Dorney *et al*. 1984; Nowak & Rowntree 1990; Zipperer 2002). The invasive potential of Norway maple is well known; however, its invasion dynamics in urban remnant forests has not been extensively studied. Bertin *et al*. (2005) sampled the vegetation of 32 urban woodlands (Table 2.4.2) in Worcester, Massachusetts (USA) and concluded that Norway maple has the potential to profoundly transform native woodlands, particularly those on mesic sites, at the expense of native trees such as white ash (*Fraxinus americana*) and red oak (*Quercus rubra*).

In Syracuse, New York (USA), Zipperer (2002) studied human disturbances as they relate to vegetation structure and composition of the tree layer in urban forests. He distinguished between remnant and regenerated forest, with the former having never been cleared and the latter having been cleared in the past. As might be expected, regenerated urban patches displayed more evidence of human disturbance, greater density of younger trees, and greater presence of non-native trees than remnant patches. Species richness was higher in regenerated patches; however, this reflects a sizeable presence of nonnative species (including sizeable numbers of Norway maple) which accounted for 23 per cent and 7 per cent of tree species in regenerated and remnant patches, respectively. In fact, in regenerated patches a full 48 per cent of all tree-sized stems were of introduced species.

Despite the type and nature of disturbances, the influx of invasive species, and management challenges, Guntenspergen and Levenson (1997), Zipperer (2002), and many other authors are right to note that urban forest patches provide important and necessary ecosystem function and social benefits. Such functions and benefits are free, in large measure, of the sizeable expenditures of energy and resources needed to maintain other types of urban vegetation, such as street trees, parks, and lawns.

2.4.5.3 The problem of invasive plants in remnants

The threat from invasive species is second only to habitat loss as a cause of extinctions (Soulé 1990). The impact of invasive plants on forest patches (within or outside cities) can be profound (Hobbs & Mooney 1998). Although invasion biology is currently receiving much attention, the ecology of many non-native problematic species is still greatly under-studied in natural settings and much less in urban areas (see Cilliers and Siebert, Chapter 3.2). However, some understanding of invasion biology offers approaches to predicting potential invaders based on demographic and historical factors (Reichard & Hamilton 1997; Heger & Trepl 2003). The application of invasion biology knowledge is essential for effective management of the urban forest (e.g. Borgmann & Rodewald 2005).

In their study urban gradients, McDonnell and Pickett (1990) ask if the regeneration of current or prior dominants is limited in forests at the urban end of the gradient. The answer, in some case, will be yes. For instance, Burke and Grime (1996) have shown that a combination of soil nutrient enrichment and disturbance enhance the invasibility of experimental plots. The role of feedback from soil microbiota in determining invasion success has been illustrated by a number of recent studies (e.g. Klironomos 2002; Stinson *et al*. 2006).

Furthermore, there is a growing appreciation of the significance of the impact of invaders on ecosystem processes. Recent studies have shown that non-native plants in the USA, such as *Alliaria petiolata*, alter mutualistic associations in soils via allelochemicals to the disadvantage of native species. The non-native shrub, *Rhamnus cathartica*, common in many eastern North American woodlands and urban remnants, significantly alters decomposition rates in some Chicago (USA) remnants (Heneghan *et al*. 2004, 2007) and alters soil nitrogen pools (Corbin & D'Antonio 2004).

Even if these invasive species were completely eliminated from urban woodlands, there will still be a lingering impact to soil ecology and chemistry, a 'ghost of invasion past' or recent. Attempts to re-establish native species, then, could be more of a challenge than simply replanting. Clearly, to be successful, management plans for invasive species and restoration in the urban context must incorporate a system level perspective.

2.4.6 Drivers of biodiversity and change in urban vegetation

2.4.6.1 Climate change

Urban vegetation is commonly viewed (unintentionally, perhaps) as static. Very few studies of urban vegetation are long-term. Yet, as cities grow, increased pressure on urban vegetation will result. Furthermore, climate change, which has already been shown to affect plant phenology (e.g. Miller-Rushing & Primack 2008), could have a significant and negative impact on urban vegetation.

How urban vegetation will respond to climate change, particularly on top of the already well-documented urban heat island effect (Grimm *et al*. 2008; see Cilliers and Siebert, Chapter 3.2), is an open question. Amelioration of the urban heat island effect could include planting in strategic areas. However, in arid areas, planting would require additional inputs of water, which is already becoming a scarce resource (Grimm *et al*. 2008). Research on the effects of elevated temperatures on remnant vegetation in cities could provide clues regarding ecosystem response to global climate change (Sukopp & Wurzel 2003; Grimm *et al*. 2008). In addition, the importance of urban vegetation in carbon sequestration is not fully understood. Takahashi *et al*. (2008) found that turf, tree-planting areas, and otherwise unmanaged coppiced woodland all acted as carbon sinks.

The effects of increased temperature on distribution of non-native species have been studied for many years in Europe. As climate continues to change, it is likely that many existing non-native species will continue to increase and spread, and that new invasions will occur (Sukopp & Wurzel 2003). Furthermore, climate change could radically alter how gardeners garden and what can be readily cultivated. Plant hardiness zone maps are already being redrawn (Marris 2007).

In addition to new climate regimes affecting urban vegetation, it is increasingly clear that the converse is also true: that continued urbanization and land-use changes will affect climate (Kalnay & Cai 2003).

2.4.6.2 Socio-economics

In addition to urban habitat diversity, availability of plant materials, and other factors described here, various measure of wealth, income, and social status have also been shown to be correlated with aspects of urban plant diversity. Whitney and Adams (1980), for instance, used household income as one variable in an analysis of the development of new vegetation complexes in Chicago.

In Campos dos Goytacazes (a city in Brazil's Rio de Janeiro State), Pedlowski *et al*. (2002) demonstrated that wealthier neighbourhoods have higher diversity and numbers of trees than poor neighbourhoods. Similarly, in Phoenix (USA), the diversity of plants in neighbourhoods in which residents earned above the median annual income was twice that in less wealthy areas (Hope *et al*. 2003).

Loram *et al*. (2008) suggested that economic resources of householders in Britain is as important as other factors in explaining garden richness and composition. As Pedlowski *et al*. (2002) note, these discrepancies in income and urban plant diversity (and thus of desirability of particular neighbourhoods) can be seen as a political and social inequity for which there should be political and social remedies.

2.4.6.3 Other drivers

Additional drivers clearly affect urban floristic composition. Besides climate change and socio-economic, others have been described above in other contexts. To avoid redundancy, suffice it to say that urban habitat diversity, urban soils (often altered in various ways by non-native species and other anthropogenic impacts; see Sauerwein, Chapter 1.3), and sources and sinks of propagules are important considerations.

One factor, however, that merits some attention is traffic patterns and modes of transport. Vehicles travelling along an urban motorway in Berlin were responsible for transporting 204 species in one year, 50% per centof which were non-native (von der Lippe & Kowarik 2007). Furthermore, traffic travelling out of the city carried and deposited more species of seeds than traffic entering the city (von der Lippe & Kowarik 2008). Thus, cities, which are often considered as sinks, might be just as important sources of propagules, particularly of non-native species.

2.4.6.4 Implications for biodiversity conservation

There are many other drivers that affect the structure and composition of the urban flora. Williams *et al*. (2009) identify four urban filters (habitat transformation, habitat fragmentation, urban environmental conditions, and human preferences), which individually and collectively result in the loss and gain of species and, thus, select for particular suites of species that define the urban flora of any given location.

Politics and economics drive land-use decisions in cities everywhere, often to the detriment of urban ecosystems. However, with the increasing awareness of sustainability, carbon offsets, and other conservation-minded policies, the prospect of healthier and more widespread urban vegetation has never been greater.

The framework developed by Williams *et al*. (2009), together with other political considerations (e.g. 'urban environmental justice' in Pedlowski *et al*. 2002) could quickly advance our understanding of urbanization and urban floristic dynamics, leading to improved environments for city dwellers and management tools for urban floras.

Efforts such as Chicago Wilderness (Retzlaff 2008) have engaged local, state, and federal agencies, along with scientists and the general public in an enormous effort to restore natural systems within an urban context and, thereby, restore a balance between people and nature (see also Rosenzweig 2003).

2.4.7 Looking ahead

Several researchers have argued for a more experimental, rather than experiential, approach in advocating for rigorous and designed experiments within cities (Felson & Pickett 2005). The idea is that ecologists would be included explicitly in collaborative projects with architects, landscape architects, engineers, and others to test design alternatives from aesthetic, function, from an ecological perspectives. Felson and Pickett (2005) describe some examples of projects in various US cities that fit such a model. To these, could be added the extraordinary efforts of 'Chicago Wilderness' (Chicago, USA) in which just such an approach is being brought to integrating biodiversity conservation and restoration in an urban context (Retzlaff 2008).

Clearly, there is a great deal yet to learn about vegetation and ecosystem restoration in an urban context. The application and testing of ecological concepts stand to benefit immensely from *de novo* restoration of vacant sites, such as establishment of functional and sustainable green roofs and development of sustainable systems on urban brownfield sites.

As climate change, globalization, transport, and other factors continue to impact the natural and built environments, it becomes increasingly likely that the benefits of urban greenery will be viewed as more important than its composition. Yet greenery can be both healthy and native. As Janzen (1998) has described for the tropics, the human footprint is so pervasive that any hope of ecological restoration has to include a continued human presence. This sentiment is echoed to some extent by Rosenzweig (2003) who suggests that the next frontier in conservation ecology will be re-introducing city dwellers to 'nature' and 'reconciling' them with nature on their terms, rather than focusing on restoration of largely inaccessible sites with which city dwellers have little connection.

Many public, private, and non-governmental organizations can play significant roles in assuring civil engagement regarding management of and advocacy for urban vegetation. One such group that is making a strong effort with respect to civic engagement is the botanic garden community, including the American Public Gardens Association, as is the Brazilian Network of Botanic Gardens (Pinheiro *et al*. 2006).

Whether vegetation within cities is natural or not might matter in some meaningful ways, particularly when larger issues of ecosystem function are considered. On the other hand, many urban residents are satisfied with some measure of green, whether or not it is natural, planted, native, or non-native. With respect to how vegetation can ameliorate climate within cities (McPherson *et al*. 1997), provide psychological and other health benefits (Fuller *et al*. 2007), and offer recreational opportunities including gardening, it is worth considering the full suite of benefits provided by all vegetation.

CHAPTER 2.5

Anthropogenic Ecosystems: The Influence of People on Urban Wildlife Populations

Clark E. Adams and Kieran J. Lindsey

2.5.1 Introduction

Since the mid-1980s, the number of books on urban wildlife management topics has been growing steadily; examples include:

1. urban ecology and sustainability (Whiston-Spirn 1985; Platt, Rowntree, & Muick 1994),
2. urban wildlife habitats (L. Adams 1994),
3. urban planning (Tyldesley 1994),
4. urban species identification (Landry 1994; Shipp 2000),
5. human–wildlife conflicts (Hadidian 2007; Conover 2002),
6. urban wildlife law (Rees 2003),
7. human dimensions (Decker, Brown, & Siemer 2001; Manfredo *et al*. 2008), and even
8. urban wildlife management (Adams & Lindsey 2006, 2009).

As a result, when we were asked to contribute a chapter to this book we asked ourselves 'What still needs to be addressed? Is there anything we would have liked to delve into more deeply than space allowed in *Urban Wildlife Management*?'.

Much has been written on the impact—positive and negative—of urban wildlife on people. The impact of people on urban wildlife receives less attention but it continues to fascinate both of us given the role *Homo sapiens* plays as the primary driving force influencing the entire assemblage of urban species. Humans influence the presence or absence of nearly every taxonomic group of fish, amphibians, reptiles, birds, and mammals found in human-dominated landscapes, sometimes intentionally, sometimes not. Human influence dictates where urban wildlife will be found (i.e. habitat selection within the urban matrix), when various species appear or disappear, what the faunal assemblages will consist of, and variations in population densities. We tend to agree with Paul Rees (1997): 'One of the more fascinating dimensions of ecology is the adaptive co-evolution of species, particularly those that have hitched their wagons to the human train'. In fact, creation of anthropogenic ecosystems and the overwhelming influence humans have on the wildlife found in them argues for consideration of *Homo sapiens* as an urban 'keystone' species.

2.5.2 Definitions of 'urban' on a global scale

An 'urban area' is defined as having an increased density of human-created structures in comparison to the areas surrounding it, but it also depends on an increased density of people. Urban areas may be cities, towns or conurbations (i.e. many different metropolitan areas, including suburbs, connected to one another), but the term is not commonly extended to rural settlements such as villages and hamlets. Working definitions of 'urban' vary by country, and even by county or regions within countries (United Nations 2008). For example, urban may be defined in terms of a specified population density (e.g. the number of people or dwellings within a defined unit of space), as the administrative centres of municipalities, or by political designation. Figure 2.5.1 illustrates the frequency of human residence in urban areas on a global scale. Of note is the high level of urban residents in North America, South America, Australia, and European countries. It would be reasonable to expect the factors that promote or preclude the existence of various species of wildlife in urban areas will be similar in these countries.

An ecological definition of urban was provided by Rees (1997), who summarized it this way: 'Large concentrations of people and industrial activity that consume more available energy and material than can be produced, and produce more wastes (entropy) than can be assimilated within the relatively small areas they occupy'.

2.5.3 Humans as a keystone species

There is evidence that prior to contemporary time, native people were keystone predators who once structured entire ecosystems. They influenced the distribution, abundance, and behaviour of wildlife, although they lacked any effective conservation strategy regarding their primary prey items (Jerozolimski & Peres 2003; Kay 2007). When it comes to urban areas, Rees (1997) claims 'humans are the creators of cities and therefore the keystone species in the urban system'. Humans have the ability to alter both the abiotic and biotic conditions and the structure and function of ecosystems; that is, people can change the natural to preferred (less objectionable) habitats. More specifically, human activities that support consideration as a keystone species are as follows (O'Neill & Kahn 2000; O'Neill 2001, Cunningham & Cunningham 2009).

1. Constructing both dispersal barriers and invasion pathways through habitat fragmentation.
2. Introducing exotic species and causing mass faunal extinctions.
3. Changing ecosystem stability by altering environmental constraints and biota.
4. Manipulating disturbance regimes by suppressing or increasing probabilities of occurrence.
5. The effect on communities ripples across trophic levels.
6. Changing competitive relationships.

Humans are an urban keystone species in the same way beaver (*Castor canadensis*) are a wetland keystone species—through alteration of natural

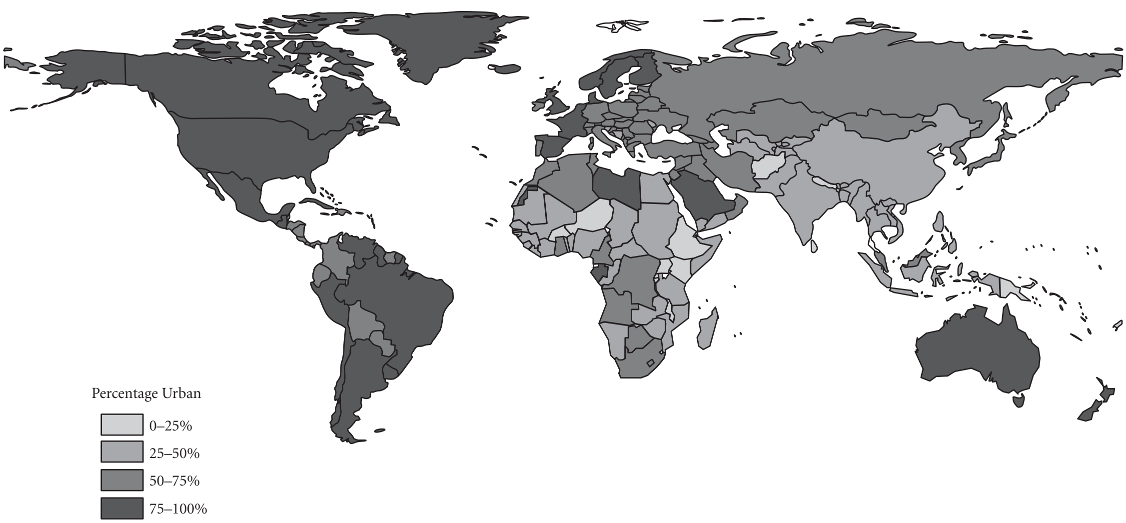

Figure 2.5.1 Urban population frequencies throughout the world (United Nations Development Program 2009, used with permission)

landscapes (terrestrial and aquatic) and the addition of built structures or infrastructures (e.g. dams, buildings and roads). The existence of urban ecosystems is solely the product of human influences that modify (directly or indirectly) nearly every aspect of their abiotic and biotic surroundings.

The assemblages of urban fauna should be predictable, in large part, by the characteristics that define humans as keystone species. There are hundreds of sources of information in the scientific and popular literature that identify common urban wildlife species, explain how they have adapted to urban ecosystems, and discuss human–wildlife conflicts. Our review of this literature provides an illustration of the assemblages of urban fauna on a global scale.

2.5.4 Assemblages of urban vertebrates worldwide

Throughout our text on *Urban Wildlife Management* (Adams & Lindsey 2009), we constructed many lists of fish, amphibian, reptile, and mammal species found in anthropogenic ecosystems throughout the United States. These species lists were developed using data sets from diverse sources including: reports of road-killed animals; fish assemblages in polluted environments; bats and birds that occupy highway bridges; bat and bird fatalities resulting from window, buildings, and communication and wind tower collisions; and those species most directly involved in human–wildlife conflicts. Our focus was on urban wildlife management in North American, with an emphasis on the US and Canada. As a result, we did not review the literature on urban faunal assemblages around the globe. The list of species in Table 2.5.1 is almost certainly incomplete and there are many reasons for this. Research results in peer-reviewed journals, government reports, and more popular outlets are overwhelmingly dedicated to the study a single species or species group rather than whole communities. This leads to an abundance of literature on a small subset of the total species composi-

Table 2.5.1 Urban wildlife in cities and countries outside the United States[1]

Wildlife species	Location	Reference
Australian Brush-turkey (*Alectura lathami*) Australian magpie-lark (*Grallina cyanoleuca*) Australian magpies (*Gymnorhina tibicen*) Australian Wood Ducks (*Chenonetta jubata*) Black Flying-fox (*Pteropus alecto*) Common bush tail opossum (*Trichosurus vulpecula*) Galah (*Eolophus roseicapillus*) Grey-headed Flying-fox (*P. poliocephalus*) House Sparrow (*Passer domesticus*) Laughing Kookaburra, (*Dacelo novaeguineae*) Little corella (*Cacatua sanguine*) Long-billed corella (*C. tenuirostris*) Pacific Black Duck (*Anas superciliosa*) Rainbow Lorikeet (*Trichoglossus haematodus*) Red Flying-fox (*P. scapulatus*) Ringtail opossum (*Pseudocheirus peregrines*) Silver gull (*Larus novaehollandiae*) Spotted Turtle-Dove (*Streptopelia chinensis*) Sulphur-crested Cockatoo (*Cacatua galerita*) Superb Lyrebird (*Menura superba*) Torresian crow (*Corvus orru*) Welcome Swallow (*Hirundo neoxena*) White ibis (*Threskiornis molucca*)	Towns and cities throughout Australia	Temby 2004

Beaver (*Castor fiber*)	Norway	Parker & Rosell (2003)
Coyotes (*Canis latrans*)	Calgary, Alberta, Canada	Lukasik & Alexander (2008)
	Vancouver, British Columbia, Canada	Boelens (2006)
Asian elephant (*Elephas maximus*)	Sri Lanka	Bandara & Tisdell (2003)
Buffalo, baboons, leopards, lions, wild pigs, lizards, hyena,	Kariba town, Zimbabwe, Africa	Svotwa *et al*. (2007)
jackals, rats, crocodiles, and hippopotamus[2]	Mozambique	Anderson & Pariela (2005)
Swiss red fox (*Vulpes vulpes*)	Zurich, Switzerland	Gloor (2002)[3]
	Oslo, Norway	
	Bristol, Edinburgh, London, and Oxford, England	
	Arhus, Denmark	
	Wales	
	Stuttgart, Germany	
	Sapporo, Japan	
	Munich, Germany	Konig (2008)
	Berlin, Germany	Schöffel *et al*. (1991)
	Melbourne, Australia	Marks & Bloomfield (1999)
	Toronto, Ontario, Canada	Adkins & Stott (1998)
	Copenhagen, Denmark	Willingham *et al*. (1996)
Feral rock doves (*Columba livia*)	Amsterdam, The Netherlands	Buijs & Van Wijnen (2001)
Kestrel (*Falco tinnunculus*)	Berlin, Germany	Kubler *et al*. (2005)
Southern brown bandicoot (*Isoodon obesulus*)	Sydney, Australia	Dowle & Deane (2009)
Long-nosed bandicoot (*Perameles nasuta*)		
Stone martens (*Martes foina*)	Europe	Herr *et al*. (2008)
Monkeys[2]	Chandigarh, India	Singh (1968)
Brazilian free-tailed bat (*Tadarida brasiliensis*)	Urban centers, Chile	de Mattos *et al*. (2000)
Vampire bats (*Desmodus rotundus*)	Throughout Brazil	Mayen (2003)
Canada goose (*Branta canadensis*; with 11 subspecies)	Australia, New Zealand, United Kingdom, Netherlands, Belgium, Scandinavia	Dawes (2008)
Wild boar (*Sus scrofa*)	Berlin, Germany	Kotulski & Konig (2008)
Feral cats (*Felis catus*)	Britain	Page & Bennett (1994)
	San Juan, Puerto Rico	USDA (2003)
	Australia	Dickman (1996)
	Auckland, New Zealand	Giles & Clout (2003)
Rhesus monkey (*Macaca mulatta*)	Aligarh, India	Imam & Yahya (2002)
Raccoons (*Procyon lotor*)	Japan, Caribbean Islands, Germany, Soviet Union	Ikeda *et al*. (2004)[3]
African elephants (*Loxodonta africana*)	Zimbabwe	Hoare (1999)
White-tailed deer (*Odocoileus virginianus*)	Jamaica, British Isles, Czechoslovakia, Finland, Yugoslavia, New Zealand, Cuba, The Virgin Islands, Curacao, and other Caribbean Islands	Chai (2003)[3]
Crows (*Corvus corax*)	Kagoshima, Japan	Fackler (2008)

[1] Rural (agricultural) human/wildlife conflicts not included, e.g. Siex & Struhsaker (1999).
[2] Scientific nomenclature not given.
[3] Other urban locations of this species were taken from citations or information in manuscript.

tion for any specific urban area and worldwide (Table 2.5.2). Furthermore, research is usually conducted using a small geographic scale of analysis, and a focus on species that are highly adapted to urban habitats.

Our review of the global literature on urban species suggests that for many species, if human urban settlements occur within the historic geographic range, the species will occupy the human settlement in numbers greater than was the case in their natu-

Table 2.5.2 Estimated total numbers[1] of vertebrate species worldwide

Fish	25,050
Amphibians	4,000s
Reptiles	7,000
Birds	10,000
Mammals	4,600
Total	46,510

Source: http://encarta.msn.com/encyclopedia_761558106/Vertebrate.html#s3.

[1] Species numbers vary based on data sources.

ral environment. Moreover, there is little humans can do to prevent the presence of some species into their urban settlements. This theme applies particularly to species that are either naturally well suited to the resulting ecosystem or able to adapt to the new conditions.

2.5.5 Similarities and differences in urban vertebrate assemblages

Other than taxonomic differences, the published literature suggests there are more similarities than differences between urban vertebrate assemblages worldwide. For example, birds and mammals dominate the literature, probably because they are the most conspicuous species and therefore easier to study. Additionally, they are the more charismatic species; unless one is an angler or a herpetophile, people do not usually get too worked up about fishes and frogs. Of course, there have been authors who have published work on urban fishes, amphibians, and reptiles, but we found few articles on species from these three groups that are endemic to the United States and other countries around the world, with the exception of Mitchell *et al*. (2008). We were more successful in the identification of examples of all five classes of urban vertebrates by examining the literature on human–wildlife conflicts in urban ecosystems, or environmental impacts on wildlife resulting from urbanization (e.g. road-kills, pollutants, and infrastructure development) (Adams & Lindsey 2009).

The literature on human–wildlife conflicts in urban ecosystems is dominated by birds and mammals on both the North American and the international scale. In the US, the urban human–wildlife conflict 'top ten list' of species or species groups consists of: raccoons (*Procyon lotor*); coyotes (*Canis latrans*); striped and spotted skunks (*Mephitis spilogale* and *M. putoris*, respectively); beaver; white-tailed, mule, and key deer (*Odocoileus virginianus* and *O. hemionis,* respectively, and *O. virginianus clavium*); Canada geese (*Branta sp*); fox and gray squirrels (*Sciurus niger* and *S. carolinensis*, respectively); opossums (*Didelphis virginianus*); foxes (*Vulpes vulpes*); and blackbirds (multiple species). Other conflicts were the result of human interactions with crows (*Corvus brachyrhynchos*); gulls (multiple species); and feral hogs (aka wild boars, *Sus scrofa*), cats (*Felis catus*), and pigeons (aka rock dove, *Columba liva*) (Adams & Lindsey 2009). Surprisingly, residents in urban communities outside the United States report human–wildlife conflicts with some of the same species as in the US (Table 2.5.1), including: beaver, raccoons, coyotes, foxes, Canada geese, pigeons, feral hogs, and cats, white-tailed deer, gulls, and crows. Many of these species were transplants from the United States to other countries and continents. The Canada goose, raccoon, and white-tailed deer have become dominant urban species in many countries outside North America, and it appears the red fox has become an urban resident throughout Europe, Australia, the US and Canada. We also found the some zoogeographic differences in the common urban species between countries (Table 2.5.1).

2.5.6 Managing wildlife in anthropogenic ecosystems

Managing vertebrate wildlife in human-dominated areas can be a herculean effort. There are many hurdles that need to be negotiated before a successful outcome can be achieved. When the preferred outcome is not as concrete as increasing a population to allow for a larger sustainable harvest, or reducing crop or livestock losses, for example, even determining what constitutes success can be challenging. Managing wildlife in anthropogenic habitats poses very different challenges than addressing a similar problem in natural/rural habitats for the following reasons (Adams & Lindsey 2009):

1. There are fewer sources of state and federal funding for management programmes.

2. It requires that many layers of jurisdiction be considered and brought into the process.
3. The scales of analysis are small, with many legal and physical impediments in highly fragmented landscapes.
4. It requires extensive training and experience in the human dimensions of wildlife management.
5. There is limited academic and agency acceptance and participation.
6. Residents have heterogeneous attitudes and expectations related to wildlife.
7. There is high public demand for inclusion in the management process.
8. The potential threat to public health from zoonotic diseases and parasites needs to be considered.
9. The management objective is to reduce artificially abundant wildlife populations in perpetuity.
10. Not all managers have the necessary training in wildlife management.

2.5.7 Required adaptations to exist and thrive in urban ecosystems

As a keystone species, humans cause sudden and dramatic changes to nearly any ecosystem they exploit. The usual effect of human encroachment is to simplify and destabilize ecosystem structure and function. When other animals are confronted with these changes to the ecosystem there are three possible outcomes: adapt, move, or die. Those vertebrate species that can survive in urban settlements are either lucky (i.e. they just happen to require the specific conditions found in their urban ecosystem) or they have a distinct set of characteristics that allow them to adjust and survive, possibly even thrive. Those characteristics include (Adams & Lindsey 2009):

1. They are physiologically tolerant of extreme variations in abiotic conditions.
2. They have a large zoogeographic distribution (i.e. several continents).
3. They are generalists rather than specialists when accessing and using available food, shelter, and water resources.
4. They have high reproductive and survival rates.
5. They are able to habituate to human activity.
6. They have few competitors and/or predators (usually the first species groups removed from urban settlements).
7. They are able to tolerate and negotiate highly fragmented landscapes with abundant edge.
8. They have high rates of recruitment through immigration.

The characteristics of successful urban species present a real paradox for human residents when the topic of 'managing' overabundant wildlife becomes an issue in the community. Humans are more than willing to provide supplemental food sources and shelters in order to observe urban wildlife, but this activity invariably leads to rapid population growth of those species using these resources. Overpopulation of these species creates a host of human–wildlife conflicts that have negative health and economic consequences for humans. People are then faced with the dilemma of reducing—but not eliminating entirely—population levels, and more often than not using strategies that do not attempt to prevent the human behaviours that caused the population explosion in the first place (Fig. 2.5.2).

2.5.8 The built environment as hazard and habitat

The 'built environment' is both a detriment and benefit to the survival of various wildlife species. On the one hand, it seems that if humans build it, some species of urban wildlife will run into it—the impact of building and tower strikes on migrating birds, for example, is significant, possibly even devastating. In addition, an urban settlement represents a huge obstacle to survival for scores of once endemic vertebrates. On the other hand, buildings, bridges, and towers have been a benefit to the survival and proliferation of many species including eagles, falcons, bats, and swallows.

Figure 2.5.2 Residents of Lubbock, TX bringing food to a large flock of overwintering Canada geese in the winter. (Photo by Kaitlin Haukos)

2.5.8.1 Buildings

The assemblage of vertebrate species that have accepted urban structures and their associated features (e.g. attics, roofs, lawns, and gardens) as alternative substrates to conduct their life cycle activities include at least 204 bird, 51 mammal, and 47 amphibian/reptile species (http://www.enature.com/). Buildings and other tall structures take an incredible toll on bird and bat populations, particularly during migration seasons (Fig. 2.5.3). In Chicago, tall buildings may be killing 2,000 birds a year during the peak of migration. From 1968 to 1998, more than 26,000 migrating birds died from crashing into a single building along the Chicago lakefront (De Vore 1998).

Windows account for at least 100 million migrant and resident bird fatalities each year in the United States. Collisions occur during all seasons; all times of day; and with windows facing any direction. There are several structural designs and/or window modifications that will mitigate bird fatalities (Adams & Lindsey 2009), but approximately 25 per cent (n = 225 of 917; Table 2.5.3) of the avian species endemic to the United States and Canada have been documented as window-strike victims (Klem 1989).

2.5.8.2 Communication towers

Communication towers cause an estimated 1.2 million bird deaths per year. A summary of 47 studies of bird kills at communication towers included 184,797 birds and 230 different species (Shire *et al*. 2000). Again, as with the case of window casualties, these losses represent approximately 25 per cent of the avian species in the United States. A total of 1,157 birds representing 50 different species were killed by wind turbines at the Altamont Pass Wind Resources Area from May 1998 to May 2003 (Smallwood & Thelander 2008). Using several references, Erickson *et al*. (2001) reported 33,000 bird fatalities with wind turbines, of which 34 per cent were diurnal raptors, 32 per cent protected passerines, 14 per cent non-protected birds, 9 per cent owls, and 4 per cent waterbirds and/or waterfowl. In contrast, only 10 bat species have been verified as having collided with wind turbines; most of the fatalities were comprised of three species of tree bats that migrate long distances and do not hibernate, including the hoary (*Lasiurus cinereus*), eastern red (*Pipistrellus subflavus*), and silver-haired bat (*Lasionycteris noctivagans*) (Johnson & Strickland 2003, Table 2.5.4). Species lists identifying the birds that fall victim to tower collisions can be found in

Figure 2.5.3 This photo shows a sample of birds collected beneath a high-rise building in Toronto during one migration season. (Mark Jackson and Fatal Light Awareness Program)

Table 2.5.3 Avian species most frequently reported striking windows in the United States and Canada (Klem 1989)

American Robin (*Turdus migratorius*)	White-throated Sparrow (*Zonotrichia albicollis*)
Dark-eyed Junco (*Junco hyemalis*)	Ruby-throated Hummingbird (*Archilochus colubris*)
Cedar Waxwing (*Bombycilla cedrorum*)	Tennessee Warbler (*Vermivora peregrine*)
Ovenbird (*Seiurus aurocapillus*)	Yellow-bellied Sapsucker (*Sphvrapicus varius*)
Swainson's Thrush (*Catharus ustulalus*)	Purple Finch (*Carpodacus purpureus*)
Northern Flicker (*Colaptes auratus*)	Common Yellowthroat (*Geothlvpis trichas*)
Hermit Thrush (*Catharus guttatus*)	Rose-breasted Grosbeak (*Pheucticus ludovicianus*)
Yellow-rumped Warbler (*Dendroica coronata*)	Gray Catbird (*Dumatella carolinensis*)
Northern Cardinal (*Cardinalis cardinalis*)	Wood Thrush (*Hvlocichla mustalina*)
Evening Grosbeak (*Coccothraustes vespertinus*)	Indigo Bunting (*Passerina cyanea*)

Table 2.5.4 Number and total proportion of bats collected at the wind towers in 10 different states from 1998 to 2002 (Johnson *et al*. 2003 and 2004; Johnson & Strickland 2003 and 2004)

Species	Number of carcasses	Per cent of identified fatalities
Hoary Bat (*Lasiurus cinereus*)	601	41
Red Bat (*Lasiurus borealis*)	441	30
Eastern Pipistrelle (*Pipistrellus subflavus*)	142	10
Silver-haired Bat (*Lasionycteris noctivagans*)	113	8
Little Brown Bat (*Myotis lucifugus*)	109	7
Unidentified	25	2
Big Brown Bat (*Eptesicus fuscus*)	22	1
Northern Long-eared Bat (*Myotis septentrionalis*)	7	<0.5
Mexican free-tailed Bat (*Tadarida brasiliensis*)	1	–
Long-eared Myotis Bat (*Myotis evotis*)	1	–
Total	1,462	100

several tables and citations provided in *Urban Wildlife Management* (Adams & Lindsey 2009).

2.5.8.3 Roads

One method of determining which wildlife species are present in urban communities is to conduct a census of road-killed animals. The information derived is not an entirely accurate portrait of the wildlife community because the data will vary based on a species' ability to cross the road (e.g. roadrunners versus tortoises), the road's location, number of lanes and speed limit, and the scale of analysis. Because of the slow movements of amphibians and reptiles, as well as the relative distance smaller animals must travel to cross the pavement, these groups are more likely to be road-kill victims.

Table 2.5.5 presents data from a census of road-killed animals over ca. 12 km (7.4 miles) of suburban roads in Indiana from March 2005 to July 2006. The researchers combined their tally of road-killed herptofauna with two other studies, resulting in a total of 42,502 dead amphibians and reptiles over

Table 2.5.5 Vertebrate species recorded along four Tippecanoe County, Indiana, USA survey routes, 8 March 2005–31 July 2006. Overall total = 10,515 road-kills. (*indicates species of special conservation concern in Indiana). (Glista *et al*. 2008)

Mammalia			Aves		
Scientific name	**Common name**	**TTL**	**Scientific name**	**Common name**	**TTL**
Blarina brevicauda	Northern Short-tailed Shrew	14	*Agelaius phoeniceus*	Red-winged Blackbird	8
Canis familiaris	Domestic Dog	1	*Branta canadensis*	Canada Goose	2
Canis latrans	Coyote	1	*Butorides virescens*	Green Heron	1
Didelphis virginiana	Opossum	79	*Cardeulis tristis*	American Goldfinch	1
Felis catus	Domestic Cat	5	*Cardinalis cardinalis*	Northern Cardinal	9
*Lasiurus borealis**	Eastern Red Bat	1	*Chaetura pelagica*	Chimney Swift	36
Marmota monax	Woodchuck	1	*Colaptes auratus*	Northern Flicker	1
Mephitis mephitis	Striped Skunk	16	*Dumetella carolinensis*	Gray Catbird	1
Microtus ochrogaster	Prairie Vole	1	*Eremophila alpestris*	Homed Lark	1
Microtus pennsylvanicus	Meadow Vole	15	*Hirundo rustica*	Barn Swallow	5
Mus musculus	House Mouse	2	*Melanerpes erythrocephalus*	Red-headed Woodpecker	2
Mustela vison	Mink	6	*Melospiza melodia*	Song Sparrow	9
Odocoileus virginianus	White-tailed Deer	4	*Molothrus ater*	Brown-headed Cowbird	2
Ondatra zibethicus	Muskrat	10	*Otus asio*	Eastern Screech Owl	6
Peromyscus spp.	Deer/White-footed Mouse	39	*Passer domesticus*	House Sparrow	15
Procyon lotor	Raccoon	43	*Passerina cyanea*	Indigo Bunting	3
Scalopus aquaticus	Eastern Mole	4	*Phasianus colchicus*	Ring-necked Pheasant	2
Sciurus carolinensis	Eastern Gray Squirrel	23	*Porzana carolina*	Sora	1
Sciurus niger	Eastern Fox Squirrel	27	*Quiscalus quiscula*	Common Grackle	6
Sorex cinereus	Masked Shrew	1	*Spizella passerina*	Chipping Sparrow	1
Spermophilus tridecemlineatus	13-lined Ground Squirrel	6	*Sturnella magna*	Eastern Meadowlark	2
Sylvilagus floridanus	Eastern Cottontail	37	*Sturnus vulgaris*	European starling	11
Tamiasciurus hudsonicus	Red Squirrel	6	*Tachycineta bicolor*	Tree Swallow	1
Tamias striatus	Eastern Chipmunk	7	*Troglodytes aedon*	House Wren	1
Vulpes vulpes	Red Fox	1	*Turdus migratorius*	American Robin	18
?	unknown bat	2	*Zenaida macroura*	Mourning Dove	4
?	unknown mammal	8	?	unknown bird	56
Total		**360**	**Total**		**205**

Amphibias			Reptilia		
Scientific name	**Common name**	**TTL**	**Scientific name**	**Common name**	**TTL**
Ambystoma tigrinum	Eastern Tiger Salamander	142	*Chelydra serpentina*	Snapping Turtle	23
Bufo americanus	American Toad	111	*Chrysemys picta*	Midland Painted Turtle	28
Hyla spp.	Tree Frog	1	*Elaphe obsoleta*	Black Rat Snake	5
Pseudacris crucifer	Spring Peeper	8	*Elaphe vulpina*	Fox Snake	9
Rana catesbeiana	Bullfrog	1,671	*Graptemys geographica*	Northern Map Turtle	1
Rana clamitans	Green Frog	172	*Nerodia sipedon*	Northern Water Snake	1
Rana palustris	Pickerel Frog	18	*Storeria dekayi wrightorum*	Midland Brown Snake,	19
*Rana pipiens**	Northern Leopard Frog	74	*Terrapene carolina*	Eastern Box Turtle	1
Rana spp.	unknown ranid	7,602	*Thamnophis sirtalis*	Common Garter Snake	35
?	unknown frog	10	*Trachemys scripta*	Red-eared Slider	13
Total		**9,809**	?	unknown snake	4
			?	unknown turtle	2
			Total		**141**

488 survey days. One way to interpret these data is that roads may be having as much of an effect on the loss of herptofauna populations as habitat destruction (to which, of course, roads contribute), climate change, infectious diseases, and UV radiation (Adams & Lindsey 2009).

To offset the road-kill carnage, wildlife crossings are being built in the United States and Europe to provide safe passage over and under major highway systems. These crossings may consist of bridges covered with native vegetation to make the passage more inviting to wildlife than bare concrete. Another approach to wildlife crossings is a box culvert or a riparian corridor under the highway. Fencing can be used, as appropriate, to funnel wildlife into these safe passages rather than cross in the midst of heavy traffic. Depending on their design, however, these structures may also efficiently funnel prey species toward waiting predators! More information on wildlife crossings can be obtained by accessing the Wildlife Crossings Toolkit (www.wildlifecrossings.info) developed by the USDA Forest Service.

2.5.8.4 Bridges

In stark contrast to the hazard posed by roads, some wildlife species have taken advantage of the nesting, roosting, and dispersal opportunities provided by bridges, underpasses, overpasses, and culverts (Fig. 2.5.4). Cliff swallows (*Hirundo pyrrhonota*), have expanded their range across the Great Plains and into eastern North America. Originally a cave-nester, the barn swallow (*Hirundo rustica*) has almost completely converted to nesting under or inside structures such as buildings and bridges (Brown & Brown 1999). The cave swallow (*Hirundo fulva*) has undergone a dramatic range expansion in Texas and has also colonized south Florida by adopting bridges and culverts as nesting sites. Other birds that make use of bridges include the eastern phoebe (*Sayornis phoebe*), northern rough-winged swallow (*Stelgidopteryx serripennis*), pigeon (*Columba livia*), and osprey (*Pandion haliaetus*). Over half of the 45 species of bats found in the US use bridges as roosting sites (Table 2.5.6).

2.5.8.5 Infrastructure development

There is a profound effect on ecosystem structure when natural soil, streams, trees and other native plants are replaced by impervious surface-covers (e.g. sidewalks and roads), channelized waterways, buildings, turfgrass and other exotic flora. High rise buildings and urban residences, windows, and towers (communication and wind) are dominant structures in cities and urban neighbourhoods. Some comfort may come from the realization that, given time and even a small opportunity, secondary succession of endemic species will eventually move in to replace the urban artifice (see Chernobyl discussion below).

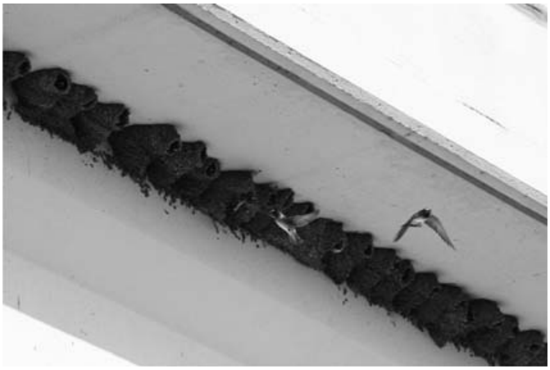

Figure 2.5.4 Cliff swallows nesting (N = 100 nests) under a bridge overpass in College Station, TX. (Photo by Clark E. Adams)

2.5.8.6 Pollutants

One of the greatest hazards to humans and wildlife living in developed areas is the witches' brew of toxic chemicals present in some urban soils, in air, and in water. In fact, at some time or another, humans have probably produced and released into the environment, directly or indirectly, every conceivable toxicant. The list of toxic chemicals is frighteningly long, but these substances can be categorized as follows: hazardous materials (HAZMATS), pesticides, herbicides, rodenticides, heavy metals, industrial discharges, and pharmaceutical products (Adams & Lindsey 2009). Radionuclides can also be added to the list. It is known that chronic exposure to high levels of background radiation causes cellular and other genetic abnormalities in various vertebrates. Research on particular species or species groups included Baker *et al*. (1996) for small mammals, Sugg *et al*. (1996) for channel catfish (*Ictalurus punctatus*), Møller *et al*. (2005) for barn swallows, and Geras'kin *et al*. (2008) for several fish, amphibian, and mammal species.

Environmental pollutants play an important role in the mortality of some wild species, depending on varying thresholds of tolerance. As a result, the diversity of urban vertebrate assemblages will decrease, but the density of more tolerant species groups will increase. This is particularly evident in urban fish assemblages (Table 2.5.7). The interested reader can find an in-depth discussion of the selective effect of environmental lead, mercury, and DDT on urban robins (*Turdus migratorius*), peregrine falcons (*Falco peregrines*), waterfowl, fish, frogs, and alligators (*Alligator mississippiensis*) in Adams and Lindsey (2009) and elsewhere.

Table 2.5.6 Bat species known to occupy highway bridges in the United States (Keeley & Tuttle 1999)

Common names	Scientific names
1. Big brown bat	*Eptesicus fuscus*
2. Big–free-tailed bat	*Tadarida molossa*
3. California leaf-nosed bat	*Macrotus californicus*
4. California myotis	*Myotis californicus*
5. Cave myotis	*Myotis velifer*
6. Eastern long-eared myotis	*Myotis evotis*
7. Eastern pipistrelle	*Pipistrellus subflavus*
8. Evening bat	*Nycticeius humeralis*
9. Fringed myotis	*Myotis thysanodes*
10. Gray myotis	*Myotis grisescens*
11. Indiana bats	*Myotis sodalis*
12. Little brown myotis	*Myotis lucifigus*
13. Long-legged myotis	*Myotis volans*
14. Mexican free-tailed bats	*Tadarida brasiliensis*
15. Mexican long-tongued bat	*Leptonycteris nivalis*
16. Northern long-eared Myotis	*Myotis septentrionalis*
17. Pallid bats	*Antrozous pallidus*
18. Rafinesque's big-eared bat	*Plecotus rafinesquii*
19. Silver-haired bat	*Lasionycteris noctivagans*
20. Southeastern myotis	*Myotis austroriparius*
21. Western or Townsend's big-eared bat	*Plecotus townsendi*
22. Western pipistrelle	*Pipistrellus hesperus*
23. Western small-footed myotis	*Myotis subulatus*
24. Yuma myotis	*Myotis yumanensis*

2.5.8.7 Adaptation and resiliency

The built environment is both a boon and a bane to wild vertebrates within and outside anthropogenic ecosystems. The presence of urban wildlife points to the incredible resiliency of wildlife when subjected to the lethal effects of human-built structures. The data provided above are not isolated occurrences; they happen daily or at least seasonally, year afteryear, and yet these species persist. As the species with the highest recorded mortality from both window and tower collisions, the oven bird (*Seiurus*

Table 2.5.7 Fish species in urban stream communities tolerant of wide ranges in chemical and physical changes (Barbour *et al*. 1999)

Common name	Scientific name	Trophic designation*
1. Banded killifish	*Fundulus diaphanous*	I
2. Black bullhead	*Amerus melas*	O
3. Blacknose dace	*Rhinichthys atratulus*	G
4. Bluegill	*Lepomis macrochirus*	I
5. Bluntnose minnow	*Pimephales notatus*	O
6. Brown bullhead	*Ameiurus nebulosus*	I
7. Catfish	*Ictalurus spp.*	G
8. Central mudminnow	*Umbra limi*	I
9. Comely shiner	*Notropis amoenus*	I
10. Common carp	*Cyprinus carpio*	O
11. Creek chub	*Semotilus atromaculatus*	G
12. Eastern mudminnow	*Umbra pygmaea*	G
13. Fathead minnow	*Pimephales promelas*	O
14. Golden shiner	*Notemigonus crysoleucas*	O
15. Goldfish	*Cariassius auratus*	O
16. Green sunfish	*Lepomis cyanellus*	I
17. Largescale sucker	*Catostomus macrocheilus*	O
18. Northern squawfish	*Ptychocheilus oregonensis*	P
19. Red shiner	*Cyrpinella lutrensis*	O
20. Redear sunfish	*Lepomis microlophus*	O
21. Reticulate sculpin	*Cottus perplexus*	I
22. Rudd	*Scardinius erythrophthalmus*	O
23. Silver carp	*Hypophthalmichthys molitrix*	O
24. Spotfin chub	*Cyprinella monacha*	I
25. Western mosquito fish	*Gambusia affinis*	O
26. White sucker	*Catostomus commersoni*	O
27. Yellow bullhead	*Ameiurus natalis*	I

* Trophic designations
G = Generalist; plant and animal material I = Insectivore; opportunistic predator on aquatic insects.
O = Omnivore; bottom feeder P = Piscivore; opportunistic predator on other fish.

aurocapillus) surely deserves to be placed in the urban survivor Hall of Fame. Another example of this resiliency is the ability of urban wildlife to adapt to novel and unnatural surroundings and to use human structures as they conduct their day-to-day activities, survive, and even thrive. We invite the reader to take advantage of the information sources on bird mortalities caused by human-built structures and mitigation procedures provided by the Fatal Light Awareness Program (FLAP) at (http://www.flap.org).

2.5.9 Wildlife assemblages in a city without people

From an ecological perspective, one of the most catastrophic human-induced disasters to occur in the twentieth century was the meltdown of the fourth reactor of the nuclear power plant in Chernobyl, Russia, on 26 April 1986. The explosion, and resulting widespread background radiation (3,000 times higher than areas outside the blast zone) caused the evacuation of 135,000 people. Unexpectedly, the event has provided an opportunity to watch the successional process of an anthropogenic ecosystem once the keystone species (*H. sapiens*) has been removed.

Over the last two decades, studies on vertebrate faunal diversity have revealed that animals inside the exclusion zone (a 30 km or 18.6 miles radius around the blast zone) were experiencing a population boom. The exclusion zone contained a higher diversity and greater abundance of wild vertebrates than the area outside this zone, where human activity and resultant impacts on wildlife persist. Researchers reported numerous sightings of moose (*Alces alces*), roe deer (*Capreolus capreolus*), Russian wild boar, foxes, river otter (*Lutra canadensis*), and rabbits (*Lepus eropaeus*) within, but not outside the zone (Flanary *et al*. 2008). Baker *et al*. (1996) found that within the exclusion zone, 355 specimens representing 11 species of small mammals were obtained, whereas 224 specimens representing 12 species were obtained from outside the exclusion zone. It was concluded that the diversity and abundance of the small-mammal fauna is not presently reduced at the most radioactive sites. Flanary *et al*. (2008) attributes this faunal diversity to the relocation of thousands of Ukranian citizens outside the exclusion zone, thus allowing the ecosystem within the exclusion zone to flourish (i.e. go through secondary succession) in the absence of human activity.

2.5.10 Anthropogenic ecosystems: are humans key?

Clearly humans have a large and ongoing influence on wildlife species within anthropogenic ecosystems but do urban wildlife communities need people to survive? Sure, human neighbours offer some perks, but do people play a fundamental role in maintaining the plants and animals in this ecosystem, as one would expect from a true keystone? At first glance, the Chernobyl research discussed above would seem to suggest the answer is no.

On closer scrutiny, however, the results are less than definitive. Researchers did find abundant wildlife in the exclusion area, but were the species present the same ones living in Chernobyl prior to the evacuation of its human residents? We were unable to find any published research on the urban wildlife assemblage of Chernobyl prior to the evacuation. Further review of the literature failed to uncover any before-and-after research on wildlife communities in anthropogenic ecosystems, although surely there are any number locations in the US alone that could serve as study sites (e.g. Superfund sites).

The problem with attempting a study of this kind is that shortly after the people leave one is no longer studying urban wildlife in an urban habitat. The transformation begins as previously limitless food resources disappear; the same is likely true for water resources, particularly in arid regions. Exotic landscaping plants and lawns no longer thrive in the absence of regular watering and fertilizer. Native successional plants move in, but are unable to provide the food resources necessary to support the large animal populations that grew from overflowing wildlife feeders, pet bowls, restaurant dumpsters, and landfills. Predator species may initially switch from garbage and housecats to weakening prey species, but if their numbers were artificially high before the humans left they will soon feel the effects as well. Shelter and nesting sites provided by urban infrastructure—attics, basements, chimneys, bridges—will take longer to disappear, but they will begin to degrade quickly once maintenance efforts end. Wild species that were intolerant of humans begin to move in from surrounding undeveloped areas, challenging the urban species for food, water, and space.

Observing this change in the wild residents of an urban settlement certainly has scientific merit, just as there is merit in observing the changes that occur when beaver are removed from a wetland habitat, or prairie dogs from a grassland. But for all practical purposes, once the people leave an anthropogenic ecosystem it almost immediately ceases to be an anthropogenic ecosystem. Could there be a clearer example of the influence of humans on urban wildlife and habitats, or a stronger argument for the designation of *Homo sapiens* as the ultimate urban keystone species?

Summary

Glenn R. Guntenspergen

Much remains to be learned about how vegetation and wildlife responds to and participate in the dynamics of urban landscapes. The forces associated with urban development result in complex gradients leading to whole-scale re-assortment and modification of the species pool. Recognizing if patterns of biodiversity result from this milieu is an essential part of understanding the structure and function of any ecosystem and the ability to predict how ecosystems and their dynamics may change (Guntenspergen & Levenson 1997). Urban settings are great havens for high levels of biodiversity that result because of the complex interactions among physical, biotic, and social drivers that occur there and that influence such fundamental processes as colonization, disturbance, the distribution of species along environmental gradients, and succession. The chapters in Section 2 identify general patterns occurring in urban habitats and begin to detail the relationships and interactions occurring in urban ecosystems.

In his classic essay on plant succession 'The fundamentals of vegetational change', the American ecologist W.S. Cooper argued that it was the complex interaction of a broad array of causes that resulted in vegetation change. His metaphor of vegetation as a braided stream and its causes suggested a 'complex whole governed by groups of causes, all of which influence its course'. 'The vegetational stream is governed and directed by the interaction of factors residing in the constituent organisms and their environment....The net change...is determined by the resultant of their activities'. It is not too far of a reach to see that this interactive multivariate view of vegetation can incorporate the role of humans as parts of ecosystems—the contemporary paradigm of urban ecosystems. This prevailing view that includes humans as part of the ecosystem explicitly recognizes the need to also incorporate human social institutions (and decisions) as well as the built environment to further our understanding of urban biodiversity.

The chapters in this section reflect this concept and recognize the need to understand the critical role of biotic and physical drivers as well as the physical infrastructure and social drivers in understanding urban biodiversity patterns. Although we can recognize the various factors that can influence biodiversity in cities and have begun to develop conceptual frameworks to explain these patterns, we need both comparative and experimental studies that test the importance of the broad array of causes which influence the course of the 'braided stream' and determine whether they generate recognizable patterns.

Because of the complexities of urban landscapes, it is imperative that we make progress in a number of areas. How one views the urban landscape is important. Wittig (2009) recognized the need to clearly differentiate between the boundaries of the urban landscape and urban and non-urban habitats, a theme that echos the comments of McIntyre *et al*. (2001) to clearly quantify urban characteristics. Lehvävirta and Kotze (2009) lay out clear guidelines and methodologies for conducting comparative research in urban settings. However, it is also clear that what we need to make progress is an approach that includes appropriate tools and methodologies that provides urban ecologists with a way to identify recognizable patterns of biodiversity, examine the impact and importance of those processes in generating these patterns, and an opportunity to examine the impact and importance of these patterns in the context of the entire system.

SECTION 3

Ecology in Cities: Processes Affecting Urban Biodiversity

SECTION EDITOR: **Nancy E. McIntyre**

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Introduction

Nancy E. McIntyre

Urbanization alters climate, resource quality and availability, the physical structure of a locality, and many other factors that in turn dictate the types, abundances, and behaviours of organisms within cities. The field of urban ecology was developed to understand the specialized ecosystems that are cities, so a book on urban ecology would not be complete without a discussion of how urbanization affects non-human organisms. Although many of the other chapters in this book include plant and animal examples, the five chapters in Section 3 adopt an explicitly taxonomic-centred view, focusing chiefly on birds, plants, arthropods, reptiles, and amphibians in urban settings. In contrast to the chapters in Section 2, which explore the ecology 'in' cities in terms of patterns of urban flora and fauna, the chapters in Section 3 serve as a transition from the ecology 'in' cities towards the ecology 'of' cities through an examination of how ecological processes are shaped by urbanization. Building on the habitat descriptions of Section 2, the chapters of Section 3 explore organism–environment relations within cities, whereby the inhabitants must adapt to the disturbance and fragmentation of habitat, living in sympatry with introduced species, and dealing with an unfamiliar series of selective pressures, including direct and indirect interactions with humans. These chapters lead into Section 4, which focuses on the ecology 'of' cities (full integration of built and ecological systems) and thence into Section 5, which explores urban design to maximize positive human–urban wildlife interactions and minimize negative ones.

The chapters in Section 3 distill our existing knowledge of how urban development affects organisms. The opening chapter by Barbara Clucas and John Marzluff sets a broad, evolutionary framework, exploring how urbanization is imposing novel selective pressures and resulting in unique adaptations to city life, with a primary focus on birds (but also with examples of plants and mammals). The review of urban plants by Sarel Cilliers and Stefan Siebert demonstrates how human decisions and environmental conditions combine to generate a uniquely urban botanical assemblage that serves biotic and aesthetic roles. Like the urban plant community, urban arthropod communities also exhibit the loss of certain species and gain of others; the chapter by Johan Kotze, Stephen Venn, Jari Niemelä, and John Spence looks at arthropods beyond their customary role as pests and instead focuses on their role as acutely sensitive environmental indicators of land-use change, the urban 'heat island' effect, and contaminant levels in the soil, air, and water. In contrast to other urban biota, the herpetofauna (reptiles and amphibians) in cities are typically depauperate relative to non-urbanized areas, exhibiting a net loss of biodiversity; the chapter by Bruce Grant, George Middendorf, Haseeb Amahdm, Mike Vogel, and Mike Colgan examines how some of the very traits that characterize herpetofauna—relatively limited dispersal capacity, high trophic level, and ectothermy—can constrain the ability of these ancient lineages to adapt to modern circumstances. Having these organism-based chapters under the same roof, as it were, should make this compilation a go-to reference for future work in urban ecology. These taxon-based chapters are followed by a concluding chapter by Chris Swan, Steward Pickett, Katalin Szlavecz, Paige Warren, and Tara Willey that provides an overall synthesis on the feedbacks that connect the human, non-human, and spatial components of urban communities.

The research covered in these chapters illustrates how socio-economic variables are just as important as biological ones, if not more so, in terms of urban biotic diversity, and how the design of the built environment influences which species future city-dwellers will encounter. Our increasingly urban society will share space with other urbanites, plants, and animals that (like their human counterparts) either tolerate or actually thrive in an ecosystem that was unknown until people constructed it.

CHAPTER 3.1

Coupled Relationships between Humans and other Organisms in Urban Areas

Barbara Clucas and John M. Marzluff

3.1.1 Introduction

Today's urban landscapes, from cities and suburbs to sparse rural and exurban settlements, are the stages for an incredible evolutionary play. Humans are the lead actors, affecting the genetic and cultural inheritance of other species. But non-human organisms are also important players, influencing our culture and genetic legacy. Humans, other species, and abiotic elements and processes are tightly coupled in urban ecosystems, perhaps more today than ever before. This coupling of human and natural systems (Liu *et al*. 2007a, b) has recently gained attention, with researchers expressing a need to investigate the mechanistic underpinnings and complexities of these interactions (Grimm *et al*. 2000; Alberti *et al*. 2003; Liu *et al*. 2007a, b). This need stems from both a purely theoretical point of view and an applied perspective. As the world becomes more urbanized, our increased well-being has come at a cost to nature, which in turn is now becoming costly to humans (Tarsitano 2006; Lui *et al*. 2007a).

Although we might appreciate the potential extent of human and natural system coupling as our urban footprint (Marzluff *et al*. 2008; Fig. 3.1.1), it is more difficult to imagine our ongoing evolution. It appears that we have distanced ourselves from the natural world, creating 'artificial' environments by altering nature in physical ways, with the removal of vegetation and building of man-made structures, and changing the way we obtain food (e.g. no longer hunting or gathering). However, as urban areas increase worldwide we are increasing both the number of our interactions with the natural world and the intensity of these interactions (Liu *et al*. 2007a, b), and these increases affect people and nature (Marzluff & Angell 2005a).

The influence of humans on other organisms, from wild to domesticated, is profound (see also Adams and Lindsay, Chapter 2.5). Humans affect species survival, population structure, reproduction, behaviour, and evolution in urban areas (e.g. Marzluff 2001, 2005; Chace & Walsh 2006; Ditchkoff *et al*. 2006; Brunzel *et al*. 2009). Changing landcover has had a major influence on wildlife populations and communities across the globe, and fragmentation, degradation, and pollution are also known to have negative and positive consequences for the survival of animal and plant populations (Donnelly & Marzluff 2006). However, less is known about the extent to which human behaviour towards other organisms (intentional or unintentional) can affect urban plants and wildlife and subsequently affect their population viability and evolution.

In this chapter we provide a brief background on the relationship between humans and natural systems and how complex feedback loops can emerge and form coupled systems. We will then focus on interactions between humans and other organisms, the selective forces they impose, and the potential outcomes of such relationships. We document interactions that began with early sedentary lifestyles up to present urban settings and discuss the potential for coevolution. We then describe in detail how a

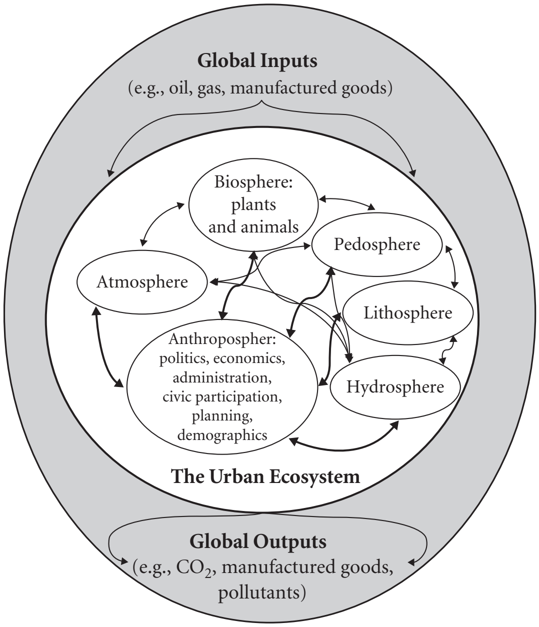

Figure 3.1.1 Components of the urban ecosystem (white area) and the relationships between humans (anthroposphere) and nature (other elements of the ecosystem) within the ecosystem. Coupling of human and natural systems is indicated by the dual-headed arrows linking components (arrow width indicates relative magnitude of linkage). The global footprint of the coupled human and natural system that is the urban ecosystem is indicated by the gray area. Reprinted from Marzluff *et al*. (2008)

focal group (birds) and humans interact in urban areas. To conclude, we will discuss how studying coupled relationships of humans and other organisms is a new direction of research for urban ecologists and some other potential areas to be explored.

3.1.2 Humans and natural systems

Humans have been important components of almost every ecosystem for thousands of years, but their influence on natural processes has increased dramatically in the recent past (Grimm *et al*. 2000; Liu *et al*. 2007a). Ecosystem dynamics are driven by patterns and processes associated with biotic and physical factors, as well as climate and element cycles. Humans contribute to these processes by using land and resources, producing waste, and changing communities of fauna and flora (Grimm *et al*. 2000). Although humans play a large role in these systems, the outcomes are not always favourable to humans themselves (e.g. pollution). Research on the complex ways humans and natural systems interact has been stressed as a means of understanding how we can work to decrease negative consequences of our actions (Liu *et al*. 2007a, b).

Interactions between humans and natural systems can form positive and negative feedback loops (Liu *et al*. 2007a). An example of a positive (unregulated) feedback loop is the over-exploitation of natural resources. Short-term economic gains for corporations can fuel the unsustainable degradation of natural systems at a large cost to the general populace. For instance, intensive farming of shrimp or salmon pollutes and depletes the aquatic environment for short-term profit but long-term loss of ecosystem services (Balmford *et al*. 2002). As natural production of salmon and shrimp collapse, aquaculture may increase because wild alternatives are extinct, further degrading the natural ecosystem and robbing society of ecological services. We suspect unregulated feedback loops to be rare, because of their great cost to society (Liu *et al*. 2007a). Perhaps more likely are negative, or self-regulating, feedback loops. For example, loss of iconic ecosystem elements in urban ecosystems, like salmon or redwood trees, may trigger policies to stem extinction (listing of salmon in Seattle, Washington, under the US Endangered Species Act, or reservation of redwood forests as US National Parks, for example) or the recognized economic benefits of ecosystem services may prompt watershed conservation over degradation and development of water treatment facilities. Self-regulation is often delayed until costs to society and nature are extreme.

Positive, mutually reinforcing, reciprocal feedback loops are also possible between humans and animals. For example, the Wolong Nature Reserve in China provides needed habitat for threatened giant pandas (*Ailuropoda melanoleuca*), but humans are also living in the reserve and they degrade the habitat by collecting firewood and setting up agriculture (An *et al*. 2006). These actions negatively affect both the panda population and the local economy because if pandas are not doing well, ecotourism suffers. An *et al*. (2006) hypothesize that if the quality of electricity is increased in the area (e.g. voltage increases), humans would collect less wood for fires and this would then support panda viability and increase profit from ecotourism.

These examples of reciprocal feedback loops between humans and nature demonstrate that to understand these interactions, one must integrate information about human socio-economic factors, animal and plant biology, as well as human behaviour (e.g. legislation).

3.1.3 Interactions between humans and other organisms

Humans have had an unavoidable link to animals in their environment. During human evolution we have in part transitioned from being prey to predators and then caretakers of other animals as our diet and lifestyles changed over time (e.g. nomadic to sedentary; O'Connor 1997, Heffner 1999). However, human–animal relationships go beyond those of food: we are tightly connected to numerous animal species, from microbial organisms in our intestines that aid digestion, to rodents living uninvited in our homes (Somerville 1999), to captive elephants that provide a work force.

To help distinguish what type of relationships humans have with certain species, it is useful to determine the effects each interaction has on each participant (positive, negative, or neutral). Most animals living with humans in urban areas are commensal: they benefit from living in proximity to humans without overtly harming them. In some cases, these relationships are very old and their origins coincide with human sedentism (O'Connor 1997). Indeed, the presence of rat (*Rattus* spp.) and mouse (*Mus domesticus*) remains can be used as a proxy of human settlement at archeological sites (Somerville 1999). These species were most likely attracted to the increase in foraging opportunities provided by human rubbish and warmth provided by human dwellings. The historical relationship of rats (*R. rattus* and *R. norvegicus*) and mice (*M. domesticus*) with humans is marked by changes in human behaviour. There is archaeological evidence for increased efficiency of food storage devices that is attributed to preventing pilferage by rodents (Somerville 1999). In addition, the negative effects (or at times negative perception) of rodents have led humans to tolerate the presence of rodent predators such as mustelids and felines (Somerville 1999).

Humans in urban areas often have negative perceptions of many other organisms sharing their environment (e.g. cockroaches, pigeons, raccoons, opossums, bats, dandelions and other weedy or poisonous plants). These organisms thrive on living in proximity with humans by foraging on human refuse or finding suitable habitat in gardens, housing, or attics. Urban pests have adapted to living with humans by adjusting their activity patterns (e.g. nocturnality) or habituating to humans. Sometimes, these organisms can have serious negative impacts on humans, such as the spread of rabies or other diseases, however in most instances they are simply nuisances.

Some urban pests are non-native species as urban areas seem to select for or attract exotic, invasive species. Native species, which are adapted to local habitats, are often replaced by these invasive species that tend to be generalists capable of quickly adjusting to an urban lifestyle (e.g. Marzluff 2001). Transition from native to exotic communities of organisms can be attributed to structural changes in habitat, temperature changes (urban areas tend to be warmer than outside areas), and, for certain species, availability of human refuse (as mentioned above) for foraging. However, humans can also directly contribute to these species changes. For instance, vegetation in cities consists of more exotics than native species due to humans planting exotics and the increased dispersal of seeds by human foot and automobile traffic (Sukopp 1973; Kowarik 1995; Brunzel *et al*. 2009).

Humans also share positive interactions with other organisms, whereby both parties benefit from forming a mutualistic relationship. Perhaps the best-known historical mutualism is that of humans and dogs, the 'domesticated' relative of wild canids. The exact origins and history of this relationship is a bit contentious (see Schleidt & Shalter 2003), but today dogs provide benefits to humans as companions, guards, and workers (hunting partners, seeing-eye, sheep herders, drug and people finders). In turn, dogs benefit from humans because they are provided with food, shelter, protection, and from the species level, have increased their population sizes and spread throughout the world (Heffner 1999). Although indirect, humans and urban animals share a positive interaction with trees and plants. Urban parks provide habitat and increase viability for many animal species while also providing a view of nature for humans which has positive health benefits (Maller *et al*. 2005).

Domestication is the extreme in symbiotic relationships between humans and animals. Domestication can loosely be defined as 'the capture and taming by man of animals of a species with particular behavioural characteristics, their removal from their natural living area and breeding community, and their maintenance under controlled breeding conditions for mutual benefits' (O'Connor 1997). Interestingly, as O'Connor (1997) points out, this definition ends by claiming animals benefit from being domesticated by humans. Indeed, domestication of animals by humans not only increased our fitness but also those of the animals, as they quickly became more numerous and widespread than their wild ancestors (Heffner 1999). Furthermore, this 'selection by man' created animals that became dependent on humans for survival and reproduction and in turn, humans developed a dependence on these animals for food, horsepower, and in some cases, companionship and protection (Heffner 1999). O'Connor (1997) continues by suggesting, 'domestication is unlikely to be one-sided' and that domestication can be seen as a form of coevolution between humans and animals (see also Schleidt & Shalter 2003).

3.1.4 Coevolution of humans and animals

Coevolution occurs in nature when two species are interlocked in evolutionary interactions that drive reciprocal selection between the species. When humans select for certain phenotypic traits in other species that, in turn, affect human phenotypes, we can call this coevolution. Two excellent examples of coevolution through positive interactions between humans and an animal species involve cooperation to gain a common food source. The first is between the Boran people of East Africa and a bird species, the Greater Honeyguide (*Indicator indicator*) (see Marzluff & Angell 2005a). Honeyguides eat wax and larvae of honeybees but cannot raid hives themselves due to the thickness of the hives. They can forage on honeycombs, however, after they lead humans to, and humans break open, the hive. Thus, working together, humans and honeyguides can efficiently locate and access a food source that is rarely obtained by either species working alone. The second example is between fishers in Laguna, Brazil, and bottlenose dolphins (*Tursiops truncates*) (Marzluff & Angell 2005b). Here again both parties have altered their behaviour to work with another species for a mutual benefit—dolphins chase fish into the fisher's nets. However, in this example the interaction between organisms is culturally coevolved (Marzluff & Angell 2005b). Dolphins learn this 'hunting behaviour' and likewise people must learn how to work with the dolphins.

Humans are known to have large evolutionary influences on other species in urban environments (Palumbi 2001) but we know less about how urban animals influence humans. Interactions between humans and animals in urban areas have the potential for forming feedback loops and leading to coevolution of behavioural (including cultural), physiological, or morphological traits. Our ancient and ongoing relationships with birds of the corvid family (crows, ravens, magpies, nutcrackers, and jays; hereafter 'crows') provide many examples, including arms races between crows and humans that are intent on scaring or hunting them, and mutually supportive interactions between crows and individual people or societies who worship them (Marzluff & Angell 2005a, b). Consider our propensity to import fruits, pave ground, and drive cars. These human cultures spawn unique corvid foraging activities, which include socially learned customs or cultures, which in turn feed back to reshape human culture. Carrion Crows (*Corvus corone*) in Sendai, Japan, harvest walnuts each autumn and carefully place them in front of cars stopped at traffic signals. When the cars move, the nuts are crushed, and the birds fly down to eat the nutritious nutmeat (Nihei & Higuchi 2001). This behaviour is spreading slowly from the place it was first observed 20 years ago, which is consistent with social learning. In accordance with cultural evolution, other populations of Carrion Crows do not use cars to crack nuts, but they do drop nuts to crack them. Drivers in Sendai are changing their habits, in part because of the interest the world has shown in their innovative crows: they now routinely and purposefully run over nuts (H. Higuchi, pers. comm.).

3.1.5 Humans and birds in urban areas

Ask an urban dweller what type of animal is seen on a daily basis and the response will likely be birds. The connection between birds and human settlements is not a novel one. The House Sparrow's (*Passer domesticus*) commensal relationship with humans is estimated to have begun between 400,000 and 10,000 years ago in the Middle East (Anderson 2006). This species is hypothesized to have become an obligate commensal species with humans because of year-round access to stored grain, and has also changed from a migratory to a sedentary lifestyle (Anderson 2006). Similar to human commensal rodent species, abundant remains of House Sparrows at archeological sites serves as a proxy of human settlement (Somerville 1999).

3.1.5.1 Effects humans have on birds

Due in part to their popularity as a study species and presence in cities, there are many studies on bird species in urban areas, addressing aspects from species compositions and abundances to life history and behaviour (see Chace & Walsh 2006; Robb *et al*. 2008; Chamberlain *et al*. 2009; Evans *et al*. 2009). The effects humans have on birds in urban areas are both positive and negative (Table 3.1.1a) and vary across species (see below; Marzluff 2001). Key factors negatively affecting bird species are habitat alteration (loss, fragmentation, small patch sizes, vegetation changes) and introduced/exotic species (predators and competitors; Chace & Walsh 2006; see also Table 3.1.1a). These factors, however, are mostly indirect. Direct negative effects of human behaviour in urban areas, such as disturbance, have received less attention (Evans *et al*. 2009, but see Fernandez-Juricic *et al*. 2003; Møller 2008; Schlesinger *et al*. 2008).

Humans may unintentionally negatively affect birds in urban areas simply by passing by a nest or walking in a foraging area (Fernandez-Juricic *et al*. 2003; Møller 2008). Human visitation to parks and other natural areas can disturb birds' foraging, breeding, and nesting behaviour (Chace & Walsh 2006). Although we have a considerable understanding about risk assessment in animals, most studies attempting to test effects of human disturbance on bird species were conducted in semi-natural areas (parks, forest remnants or fragments, or by lakes) within cities. Perhaps the assumption that birds in less natural areas in cities are more habituated to humans (see Evans *et al*. 2009) or that it is easier to conduct such research in semi-natural areas accounts for paucity of studies in less natural areas.

A major direct action by humans towards birds occurs with supplementary food. In fact, up to 43 per cent of households in the United States and 75 per cent in the United Kingdom feed birds (Robb *et al*. 2008), and 48 per cent of urban households in the UK provide food for birds (Evans *et al*. 2009). The effect of supplementary feeding on birds in urban areas has the potential to be substantial (Table 3.1.1a; Lepczyk *et al*. 2004; Chace & Walsh 2006; Fuller *et al*. 2008; Robb *et al*. 2008; Chamberlain *et al*. 2009). Most experimental studies on the influences of feeding birds, however, have been conducted in rural areas (see Evans *et al*. 2009). Nevertheless, feeding birds is likely to have an overall positive effect on birds in urban areas and is often cited as a large contributing factor to the increased density of certain species in cities.

3.1.5.2 Effects birds have on humans

Due to their abundance in urban areas, birds are often the most visible wildlife in human-dominated areas. The influences birds have on humans are outlined in Table 3.1.1b. Just the presence of birds can positively affect human health and well-being, as studies have shown that viewing nature can decrease recovery time after surgeries; improve blood pressure, cholesterol levels, and outlook on life; reduce stress and mental fatigue; and improve concentration (Bjerke & Ostdahl 2004; Maller *et al*. 2005). Birds can also perform certain ecological services for humans, such as decreasing pest arthropod numbers (Kepczyk *et al*. 2004).

The negative impacts of birds on humans are mostly indirect—damage to property (including homes, vegetable gardens, and fruit trees) and nesting and defecating in undesirable places; however, birds can transmit disease and some species will even attack humans (e.g. corvids and gulls; Marzluff *et al*. 1994). In fact, this propensity to attack people may be a cultural and genetic response of birds to the lack of hunting in cities (Knight 1984; Knight *et al*. 1987).

Table 3.1.1a Positive and negative effects of human actions on birds

Human action	Effect on birds
Direct	
Feeding	**Positive**: increased winter survival[1], population sizes[1], and prey base for raptors[1]
	Negative: decreased diet quality[2]; inadequate diet for nestlings[2]; increase disease transmission[1], predation risk[1], and exotics (negative for native species)[2]
Harassing	**Negative**: time wasted fleeing; disturbed from foraging, mating, nesting, and other activities
Unintentional disturbance	**Negative**: decrease in reproductive success (nest desertion; decrease in hatchling success, ability to feed young and parental attendance; increase in predation)[1]; disturbed from foraging, mating, nesting, and other activities decrease in species richness[3]
Intentional killing	**Positive**: decrease of exotics (positive for native species)
	Negative: decreased population sizes
Driving vehicles	**Positive:** provision of food for scavengers
	Negative: death or injury from collision[1]
Indirect	
Habitat destruction/alteration	**Negative**: loss of nesting and foraging sites; increased exposure to predators[4]; collisions with buildings[1]; decrease in species richness
Vegetation changes	**Positive**: population increases due to exotic vegetation (especially for exotic bird species)[1];
	Negative: loss of nesting and foraging sites; increased exposure to predators; decrease in species richness
Habitat fragmentation	**Positive**: increased proximity of diverse land covers and resources used by generalist species
	Negative: limiting dispersal
Pesticides	**Negative**: toxic and/or lethal[5]; decrease of arthropod pests[5]
Light pollution	**Positive**: advanced breeding[2]
	Negative: inappropriate behaviour (e.g. singing at night)
Noise pollution	**Negative**: decrease in communication efficiency[1]
Exotic species introduction	**Negative**: increased competition for food and nesting sites; increased predation
Road-kill	**Negative**: negative when a bird is killed
	Positive: positive for scavengers
Waste	**Positive**: increased winter survival and population sizes
	Negative: decreased diet quality[1]; inadequate diet for nestlings; increased exotics (negative for native species)[1]

[1] Chace & Walsh 2006
[2] Chamberlain *et al*. 2009
[3] Schlesinger *et al*. 2008
[4] Evans *et al*. 2009
[5] Lepczyk *et al*. 2004

3.1.5.3 Factors contributing to variation in human–animal interactions

The interactions between humans and birds in urban areas are influenced by various factors. First, characteristics of the urban environment, such as city age (Marzluff in press), size, geographic location, and habitat type can influence relationships. For example, in comparison to European cities, North American cities are relatively young and species compositions of birds may include species that are currently adapting to urban life or being driven to extinction, whereas older European cities may include more adapted species (Marzluff in press).

Human interactions with and attitudes towards animals can be influenced by human demographic, socio-economic, and cultural factors. For instance, age, gender, and education can affect whether landowners feed birds, and occupation and relative house size can influence if they use pesticides or herbicides, which can harm birds (Lepczyk *et al*. 2004). In addition, people in deprived areas are less likely to feed birds (e.g. Fuller *et al*. 2008). Human interest and concern for animals have been shown

Table 3.1.1b Positive and negative effects of bird actions on humans

Bird action	Effect on humans
Visiting feeder	**Positive**: reward for feeding[1], health aspects of viewing nature[3], growth of bird feeding industry
Foraging on crops and/or fruit trees	**Negative**: loss of crops or fruit
Foraging on insects	**Positive**: decrease of insect pests
Raptors foraging	**Positive**: decrease of rodent pests[1]
Defecating	**Negative**: damage to property, possible disease transmission
Nesting or roosting on property	**Positive**: healthy aspects of view nature
	Negative: damage, increased noise, fecal droppings
Damage to property	**Negative**: time and cost of repair
Visual presence	**Positive**: health aspects of viewing nature[2]; education value[3]; increasing conservation awareness[3]
Increased exotics	**Negative**: negative perception of urban animals (e.g. pigeons)[3]

[1] Chace & Walsh 2006
[2] Maller *et al*. 2005
[3] Dunn *et al*. 2006

to vary with age, gender, and education level (see review in Bjerke & Ostdahl 2004). An increase in education corresponds to an increase in positive attitudes towards animals, and there are subtle species preference differences between children and adults and between females and males (Bjerke & Ostdahl 2004). Cultural differences also exist among countries where human attitudes and actions towards animals were surveyed (e.g. USA, Germany, and Japan; Bjerke & Ostdahl 2004).

Birds' relationship with humans can vary according to life history and morphological, physiological and behavioural traits. Habitat use, degree of specialization (e.g. diet), local history (native or exotic), activity patterns (e.g. nocturnal vs. diurnal), migratory and reproductive behaviour, intelligence (e.g. degree of innovative behaviour), physiological tolerance, and personality (e.g. risk adverse or explorative) have all been shown to effect whether bird species are urban adaptors or avoiders (Jerzak 2001; Chance & Walsh 2006; Croci *et al*. 2008; Møller 2008; Marzluff in press).

3.1.5.4 Evolutionary changes

Most of the direct and indirect human actions towards birds and the subsequent effects on bird species, populations, and individuals described above and in Table 3.1.1a are proximate in nature. However, our introductions of species and industrial lifestyles often place urban birds into environments that cause rapid genetic responses (Marzluff in press). Europeans seeking to acclimate to new lands, for example, enriched native avifaunas with birds from their homeland. Since introduction, their morphology, colouration, and basic migratory habits evolved in response to novel climatic conditions. In another example, soot from urban factories in eighteenth- and nineteenth-century Europe darkened the substrates animals lived among and allowed predators to drive the evolution of colouration in many insects and at least one bird, the Feral Pigeon (*Columba livia*). Initially, heavy soot provided an advantage to dark (melanic) forms of pigeons whose better match to the dark background reduced predation and provided a selective advantage over lighter morphs (Bishop & Cook 1980). This classic demonstration of natural selection leading to 'industrial melanism' is continuing to respond to urban policies. With increased pollution control in urban areas in the latter half of the twentieth century, the lighter-coloured trees, rocks, and buildings provided safe resting sites for paler morphs and natural selection has adjusted downward the relative frequency of melanic forms (Cook *et al*. 1986).

Birds in urban environments have learned to eat a wide variety of exotic and prepared foods (Marzluff & Angell 2005a). Some of these innovations have spread through local populations as culturally evolved traditions. The cultural evolution of novel feeding behaviour has been experimentally demonstrated within urban Feral Pigeons (Lefebvre 1986). It has been implicated in the removal of milk bottle lids by tits and corvids in Britain (Lefebvre 1995) and in the use of cars as nutcrackers by Carrion Crows in Japan as discussed above (Nihei & Higuchi 2001).

Traffic, industry, and built surfaces that characterize urban areas have profoundly altered the acoustic environments that many birds rely on. Low frequency noise, disruption of sound waves by large, flat surfaces, and altered sound channels have selected for shorter, higher frequency, louder, faster, and novel songs (e.g. Slabbekoorn & den Boer-Visser 2006). Such adjustments benefit singing birds because they enable potential mates and territorial intruders to hear vocal advertisements in noisy environments (Slabbekoorn & Smith 2002). In oscine songbirds, social learning is allowing such changes to proceed rapidly by cultural evolution, which may initially constrain, but later promote, speciation (Slabbekoorn & Smith 2002).

The Great Tit (*Parus major*) is a model species illustrating adaptive, contemporary, cultural evolution of song in urban environments. Loud, low frequency noise in major European cities favours short, fast, and high frequency song (Slabbekoorn & den Boer-Visser 2006). Phenotypic plasticity in singing enables tits to vary song characters. Learning may rapidly fit songs to the acoustic environment because songs that can be heard can also be copied and those that do not elicit responses can be dropped (Slabbekoorn & den Boer-Visser 2006). Because tits learn in part through interactions with neighbours on their breeding territory, urban tit songs are consistent and different from rural tit songs (Slabbekoorn & den Boer-Visser 2006). Assortative mating for habitat-dependent song characteristics and their genetic basis may lead to reproductive isolation and possible speciation of urban tits (Slabbekoorn & Smith 2002). But currently, phenotypic plasticity and cultural evolution are simply allowing tits of various genotypes to persist in urban environments, thereby softening the force of natural selection on genetic variation.

Some evolution in urban populations is neither directional nor clearly an adaptive response to natural selection. The culturally derived unique songs of eastern House Finches (Pytte 1997), for example, may reflect the random action of genetic drift or the chance occurrence of unique individuals in small founding populations.

The detailed work needed to understand the relative contributions of phenotypic plasticity, genetic drift, and adaptive responses to natural selection is exemplified in Yeh's (2004) studies of plumage evolution in urban Dark-eyed Juncos (*Junco hyemalis*). By raising juncos from urban and wildland nests in a common environment, determining the effective population size, and quantifying the rate of evolution, Yeh concluded that the observed reduction in the proportion of white in the tails of urban (San Diego, California, USA) juncos was an adaptive response to natural selection. A variety of aspects of the San Diego environment may have favoured less conspicuous tails, including a lengthened breeding season that favours greater parental investment over aggressive defense of a territory, increased risk of predation, and novel climate, food, and vegetation.

Similar experimentation is also documenting the interplay between phenotypic plasticity and genetic differentiation in urban Blackbirds (*Turdus merula*). In European cities, lack of persecution by people, artificial light, and supplemental food have enabled Blackbirds to attain unprecedented densities. Urban Blackbirds are tamer, less migratory, breed and moult earlier, and suppress stress responses relative to rural Blackbirds (Partecke *et al*. 2006). Changes in stress response, migratory behaviour, and timing of reproduction are mainly a result of phenotypic plasticity with some genetic change in adaptive characters (Partecke *et al*. 2006). Despite adaptive genetic microevolution, divergence in neutral alleles has not been detected (Partecke *et al*. 2006). Experiments on Blackbirds and Dark-eyed Juncos show how phenotypic plasticity, behavioural innovation, and the effects of small founding populations may set the initial range of responses that birds exhibit in novel urban environments, and how heritable traits quickly evolve in adaptive responses to natural selection.

3.1.5.5 Coupled relationships

As described above, humans can form reciprocal relationships with natural systems, including animals (An *et al*. 2006). Due to the high densities and close contact of humans and animals in urban areas, these are ideal places to look for such coupled relationships. A good example is that of humans and House Sparrows (Anderson 2006).

The relationship between humans and House Sparrows is ancient (Anderson 2006). House Sparrows are considered a commensal human species that thrives in urban areas. Recently, however, declines of native populations in Europe have called attention to the understanding of the mechanisms by which House Sparrows have evolved such an obligate relationship (De Laet & Summers-Smith 2007). House Sparrow populations had decreased in the past (early 1900s) due to changes in human lifestyles: the exchange of horse transport for automobiles removed a major food source for the sparrows, as they would feed on oats that spilled out of the horses' nosebags (De Laet & Summers-Smith 2007). Therefore, it might be likely that current declines in urban centres are due to additional changes in human actions. Indeed, a recent review suggests a link between the decline and human socio-economic status (Shaw *et al*. 2008).

Habitat structure in urban areas can differ depending on the socio-economic status of humans. For example, in the UK deprived areas are characterized by older houses with gardens that contain native shrub species, whereas affluent areas are characterized by newer houses with gardens that contain more pavement and exotic shrubbery (Shaw *et al*. 2008). House Sparrows are likely to be affected by the increase of affluent areas for several reasons (Shaw *et al*. 2008; see Fig. 3.1.2). First, House Sparrows nest in roof cavities of older houses; newer houses removed these nesting possibilities. Second, replacement of native shrubs with exotic plants and pavement can decrease the abundance of arthropods that House Sparrows need to feed their young. Finally, decreasing vegetation can increase predation risk of visual predators, such as raptors and domestic or feral cats. These compounding factors can affect adult and nestling survival, reproduction, and foraging behaviour, all of which can lead to population declines.

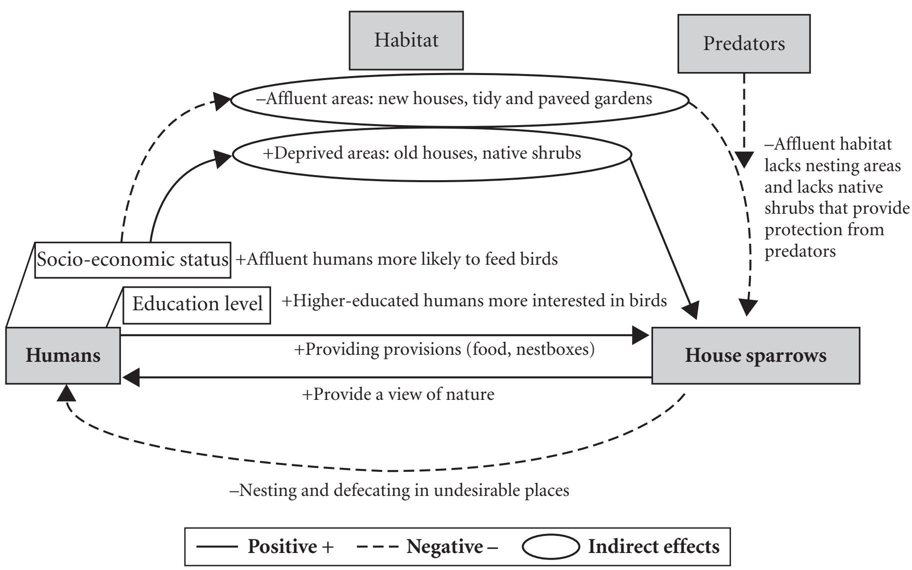

Figure 3.1.2 Interactive relationship between humans and house sparrows in urban areas. House sparrows are urban/human commensal species that profit from supplementary food provided by humans. In return, House Sparrows provide humans with a view of nature which can provide health benefits (e.g. reduce stress). Recent declines in House Sparrows in urban areas are in part due to habitat change from old houses (preferential nest sites) and native shrubs (protection from predators) to new houses with more paved and tidy gardens. However, humans in affluent areas are more likely to feed birds, and higher educated humans are more likely to be interested and concerned with birds. Therefore, combined interactions and effects of socio-economic status and education in humans and habitat selection and foraging behaviour in House Sparrows may be quite complex (see Shaw *et al*. 2008)

Due to their habituation to humans and foraging behaviour, House Sparrows provide humans with an up-close view of nature in urban areas. Any restaurant or café with outdoor seating and leftovers attract sparrows and allow for intimate interactions. Such viewing of, and interaction with, wildlife are known to positively influence human health and well-being. As shown in Fig. 3.1.2, the combined interactions and effects of socio-economic status and education in humans and habitat selection and foraging behaviour in House Sparrows may be quite complex, and it is only through an understanding of these couple interactions that we will be able to modify our actions to aid in the conservation of this species and others that share tight connections with humans in urban areas.

3.1.5.6 Potential for coevolution

A unique aspect of contemporary evolution (evolution that occurs in less than a few hundred generations) in urban ecosystems is the potential coevolutionary relationship between people and other organisms. We summarize here what we have explored elsewhere (Marzluff & Angell 2005a; Marzluff in press) of this intriguing relationship by extending the theory of niche construction (Odling-Smee *et al*. 2003) to explicitly consider how people influence and in turn can be influenced by aspects of their environment. We suggest that when humans interact with other social species, who themselves have the ability to evolve culture, then simple feedbacks from a culturally evolving 'environment' can stimulate rapid cultural evolution in humans. Exploring this 'cultural coevolution' (Marzluff & Angell 2005a, b) may expand our understanding of evolution in social urban birds and increase our awareness of the important cultural services people obtain from nature.

Humans and other social animals have a dual inheritance system whereby gene frequencies change through time in response to mutation, natural and cultural selection, and drift (genetic inheritance); and meme frequencies change through time in response to innovation, natural and cultural selection, learning, and drift (cultural inheritance; Fig. 3.1.3. At any point in time, human culture is composed of memes that reflect genetic, individually learned, and socially transmitted information (Boxes in Fig. 3.1.4; Laland *et al*. 2000). We follow the more mechanistic definition of culture—variation acquired and maintained by indirect and direct social learning (Boyd & Richerson 1985).

Humans are potent agents of natural selection affecting the microevolution of organisms and modifying the configuration and functioning of the physical environment (Fig. 3.1.4). Environmental responses to these effects can force cultural and natural selection on humans, affecting an individual's inclusive fitness and the cultural fitness of their memes, as originally postulated by gene-culture and niche construction theory. Niche construction theory recognizes that the environment changes in response to human natural and cultural selection so that humans 'inherit' a change in ecology as well as a change in gene and meme frequency (Laland *et al*. 2000). This 'ecological inheritance' is not only the physical and ecological change wrought by people, but also the cultural change in response to human activity by animals capable of social learning. Many animals, especially social, long-lived, and intelligent ones, acquire information through social learning and develop traditions that meet the social-learning based definition of culture (Avital & Jablonka 2000). Where human activity results in differential cultural fitness of another animal's memes (Fig. 1.3.3; cultural selection from humans to the environment) and the resulting cultural evolution in the animal affects the cultural fitness of human memes (cultural selection from the environment to humans), then human and animal memes may become coevolved. This cultural coevolution is analogous to traditional genetic coevolution where reciprocal natural selection between organisms drives mutual change in genes.

Corvids, a common component of urban bird communities throughout the world, provide many examples of cultural coevolution with people (Marzluff & Angell 2005a). Interacting with people favours cultural adjustment by corvids because human attitudes and important resources regularly and rapidly change. Three aspects of human culture appear especially important to corvid culture: persecution, provision of new food, and creation of new opportunities. The power of persecution is evident in the nest defense culture of crows and ravens

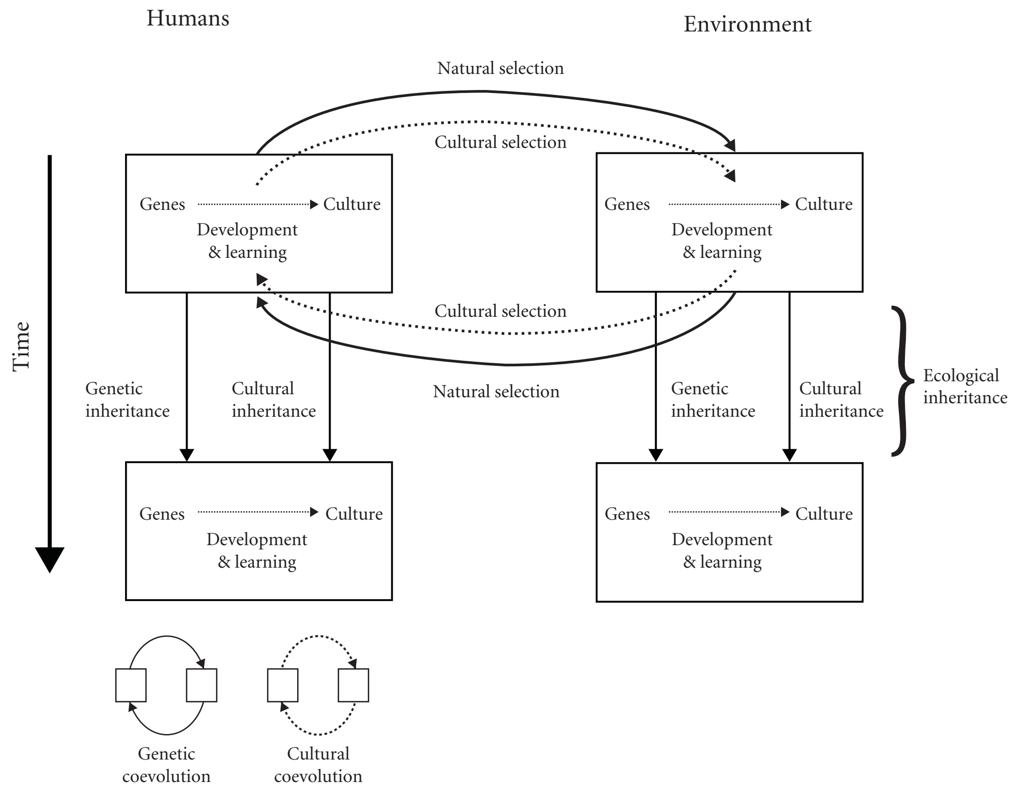

Figure 3.1.3 Integration and expansion of gene-culture coevolution and niche construction models to illustrate genetic and cultural coevolution between two interacting species (here humans and another social animal in their environment, after Laland *et al*. 2000). The collection of phenotypes within a population (boxes) emerge as a joint product of genetic, individually-acquired, and socially learned (culture) information. Populations evolve as genetic and learned information is transferred through time by genetic and cultural inheritance. As people interact with their environment they cause changes in other organisms' inclusive fitness (natural selection) or the cultural fitness of their memes (cultural selection). Reciprocal changes in humans caused by organisms in their environment can lead to coevolved genes (—) or cultures (…). Redrawn from Marzluff and Angell (2005b)

alluded to earlier. American Crows (*Corvus brachyrhynchos*) and Common Ravens (*Corvus corax*) in the western USA aggressively defend their nests in cities and towns where shooting is outlawed and in general where persecution is frowned upon, but quietly retreat out of gun range in rural areas where an aggressive bird would be wounded or killed (Knight 1984; Knight *et al*. 1987). We do not know if these cultural changes in crows and ravens affect human culture, but annoyance with aggressive city crows is common and responded to with control efforts in some cities (e.g. Lancaster, Pennsylvania, USA) which may diminish future aggressive tendencies of city crows. In other settings, people bond with crows almost as closely as they bond with their pets, feeding them daily. This favours tameness and solicitation by crows that recognize the individuals who provide for them (Marzluff *et al*. in review). The response of crows to people, be it aggressive or solicitous, has stimulated a radiation of popular culture from the naming of sports teams and rock bands to the myriad trappings and tales traditional in American Halloween celebrations.

The cultural responses of people to nature often depend on the frequency and effect of the interaction. This may be an important factor in the likelihood of coevolution. When birds like corvids are rare, people often ignore or revere them, but once

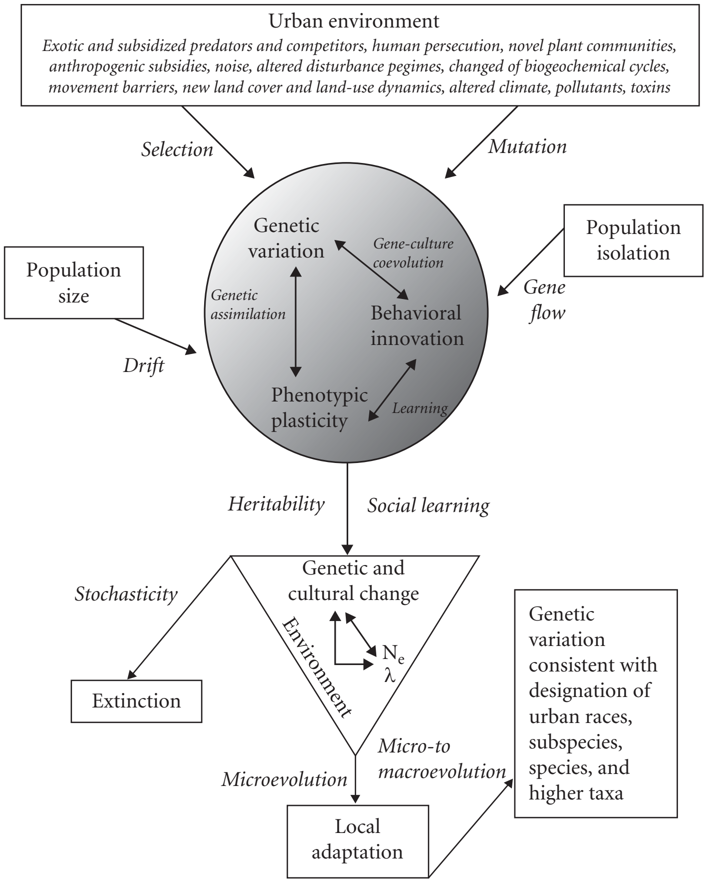

Figure 3.1.4 Key processes and interactions of contemporary evolution in urban ecosystems. Many novel and challenging aspects of urban environments affect phenotypic and genotypic variability (coloured circle) of an urban animal population, but they do so in two fundamental ways: as agents of natural selection and as mutagenic toxins and pollutants. Genotypes (lighter portion of circle) and phenotypes (darker portion of circle) interact in complex ways within and between generations to determine phenotypic variation that is exposed to selection. Responses of a population to selection may be mediated by small population size (drift) and gene flow but genetic traits are inherited and cultural traits are passed on by social learning to change the genetic and cultural composition of future generations. This change, or evolutionary adjustment to the environment, interacts with environmental and demographic stochasticity (triangle) in such a way that small populations are at risk of extinction as they are adapting to the novel urban environment. Larger or rapidly growing populations are at less risk of stochastic extinction and evolve local adaptations that may be deemed sufficient to warrant taxonomic designations of urban populations as races, subspecies, or full species. Redrawn from Marzluff (in press)

they become competitors they are viewed as pests and despised, harassed, and perhaps controlled. Our interaction with urban nature thus has a built in negative feedback mechanism that favours novel cultures in people and birds as the frequency and type of their interaction changes. When fewer in numbers and less a rival for resources, ravens, for example, were birds of the gods, even gods themselves, and useful guardians, navigators for mortals, and efficient sanitation engineers. When their abundance was thought to reduce valuable game or their consumption of human flesh evoked horror, guns and control policies were used to reduce their numbers and change their culture. Persecuted ravens became rare and shy around people. This contemporary rareness brought mystery, wonder, and concern from enough people that a culture of restoration developed (Glandt

2003). Understanding our role in such cyclical cultural phenomena may be important to conservation and restoration efforts in urban settings.

3.1.6 Conclusions

Most efforts in urban ecology are independent investigations of either how humans degrade nature or, to a lesser extent, how nature affects humans. As when studying any ecosystem, a fuller understanding of urban ecology can be gained by explicitly studying the degree to which humans and nature are linked in reciprocal feedback loops, what we refer to as coupled systems. In the case of understanding the roles of animals in urban ecosystems, we not only need to interest ourselves in the biology of non-human animals, but the effects of human behaviour on these other species and the consequences of such interactions on people. Therefore, we might find that incorporating sociological data provides insight into the adaptability of wildlife to urban areas. Indeed, as urbanization increases worldwide, future wildlife conservation efforts will increasingly broaden from focusing on natural areas to working to conserve ecological processes and species in cities (e.g. Dunn *et al*. 2006).

CHAPTER 3.2

Urban Flora and Vegetation: Patterns and Processes

Sarel S. Cilliers and Stefan J. Siebert

3.2.1 Introduction

Vegetation is the most palpable representation of any ecosystem and will determine the composition and abundance of other biota occurring within that system. Plant diversity is, therefore, an important determinant of ecosystem biodiversity in general, which is regarded as an important indicator for sustainability. Urban development, often seen as unsustainable, is a highly intensive land-use modifier and has caused drastic changes in vegetation patterns in terms of diversity, composition, and cover. Plant communities in urban areas are generally characterized by species adapted to anthropogenic disturbances, an abundance of species in eutrophic habitats, and a high diversity of non-native species (Pyšek 1998; Godefroid & Koedam 2007; Vallet *et al*. 2008). On the other hand, vegetation fulfills an important role in several ecosystem services of urban areas, such as amelioration of the urban heat island effect, removal of carbon from the atmosphere, mitigation of stormwater runoff, and flood control, to mention a few, but urban vegetation also provides important social functions in terms of aesthetic values and community well-being (Bolund & Hunhammar 1999; Alberti 2005; Hope *et al*. 2006; Cilliers *et al*. in press).

Several studies of changes in the diversity, composition, and cover of urban flora and vegetation have been undertaken. These indicate that most cities have a higher total plant species richness than natural areas, but non-native species tend to dominate while native species are declining as land-uses intensify (Kent *et al*. 1999). Kühn and Klotz (2006) mention several studies on various scales around the world that indicate similar patterns of positive correlations between human population density and species richness, and showed that native plant species may also contribute to higher plant species richness in urban areas. Increasing urbanization also affects land cover and generally leads to an increase in impervious surface area and lawn cover replacing natural vegetation (Raciti *et al*. 2008), as well as a decrease in vegetation cover in city centres (Clarkson *et al*. 2007), in parks with an increase in visitor pressure (Sarah & Zhevelev 2007), and in areas with a lower mean household income (Iverson & Cook 2000).

Descriptions of distinct vegetation patterns, gradients, and trends in urban areas are achievable (see also Section 2 of this book) but an understanding of the underlying ecological processes is difficult (Kent *et al*. 1999) although it is essential for the sustainable management of vegetated urban open spaces (Lehvävirta & Rita 2002). These processes are, however, variable on different scales and between the different components of urban vegetation and also pose a challenge to urban ecologists to compare cities across continents to better understand the general patterns.

The main objective of this chapter is, therefore, to explore the different processes that are driving vegetation patterns in urban areas. It is, however, important to explain what is understood by the term 'urban vegetation', what the current state of research on urban vegetation is, and also to discuss the way forward in terms of urban vegetation research.

3.2.2 Urban vegetation: definitions and the current state of research

An analysis of literature addressing patterns and processes of urban flora and vegetation has revealed quite a variety of approaches in terms of types and origin of species and habitats included in studies of urban environments. Most of the studies have been undertaken in northern-hemisphere countries (Hahs & McDonnell 2007), with very little urban ecological research in developing countries. Several studies have included all habitats, namely remnant natural as well as novel habitats such as parks and roadside verges. These studies referred to urban floras as a combination of indigenous species originally present in the area (native species) and indigenous species colonizing from other areas into novel habitats formed by urbanization, and non-native species (also called aliens or exotics) introduced by humans that may have the ability to establish wild populations outside parks and gardens (Dana *et al*. 2002; Chocholoušková & Pyšek 2003; Kühn *et al*. 2004; Fanelli *et al*. 2006; Williams *et al*. 2009). These studies often treat urban areas as homogenous and combine all anthropogenic factors into a single environmental variable, but essentially different types of built-up areas have different influences on plant species distribution, making it possible to describe indicator species for the different areas (Godefroid & Koedam 2007).

The term 'semi-natural vegetation' is often used in Europe to describe urban plant communities as existing somewhere between totally natural and totally artificial, acknowledging the inevitable role of human influences and 'natural factors' in the establishment of vegetation (Millard 2004, 2008). Indigenous and spontaneous vegetation are regarded as subsets of semi-natural vegetation that only differ in terms of time of development, where indigenous vegetation forms communities that have endured over several centuries and spontaneous vegetation has naturally colonized derelict urban sites over the last century (Millard 2004). In Europe spontaneous vegetation is often subdivided into those which colonized before 1500 AD (archaeophytes) and those after 1500 AD (neophytes) (Millard 2008). In studies on urban plant diversity neophytes are usually regarded as non-natives, but there is a variable approach in terms of the native or non-native status of archaeophytes in different studies. Archaeophytes are sometimes regarded as native species in European studies and this could lead to a perception of higher native species diversity in those cities than in surrounding natural areas.

Several other studies focused on plant diversity patterns in specific areas inside urban environments, especially remnant natural areas such as urban forests (Kostel-Hughes *et al*. 1998; Iverson & Cook 2000; Stewart *et al*. 2004; Clarkson *et al*. 2007; Vallet *et al*. 2008; Heckmann *et al*. 2008; Hamberg *et al*. 2009; see also Dunn and Heneghan, Chapter 2.4), but also urban wetlands and riparian areas (Ehrenfeld 2000a, 2008; Maskell *et al*. 2006) and urban grasslands (Cilliers *et al*. 2008). The emphasis in these studies was mainly on protection of indigenous vegetation, as remnant natural areas are usually targeted in conservation policies, but invasion by non-native species was also addressed.

Novel manmade habitats such as urban parks (De Candido 2004; Sarah & Zhevelev 2007), urban lawns (Raciti *et al*. 2008), and urban domestic gardens (Head *et al*. 2004; Kirkpatrick *et al*. 2006; Loram *et al*. 2007) are also studied. Domestic gardens as part of urban green infrastructure have to a large extent been ignored in urban ecological studies, and the studies mentioned here are only from a few cities worldwide. Although gardens cover more than 20 per cent of the urban area in many cities (Loram *et al*. 2007), they were not included in the studies of urban floras mentioned earlier. Spontaneous and indigenous vegetation are included in garden studies (see Dunn and Heneghan, Chapter 2.4), but more emphasis is placed on planted flora, which is also of historical importance according to Sullivan *et al*. (2005), as they form the original source of non-native species invasions.

Another urban habitat usually neglected in conservation of urban open spaces is wasteland, defined as abandoned lands where plant species grow without human control (Muratet *et al*. 2007). The established vegetation is regarded as spontaneous, also known as ruderals, as they often develop in immensely disturbed areas. Several studies have focused on wasteland flora, mainly in Europe (Godefroid *et al*. 2007; Muratet *et al*. 2007).

3.2.3 Processes affecting plant diversity patterns

The urban environment is a powerful selective force that changes biodiversity patterns but also alters the structure, function, and behaviour of city-dwelling organisms (Shochat *et al*. 2006; Grimm *et al*. 2008). A combination of mechanisms or driving forces or processes, each representing specific selection pressures, is shaping the floras of urban areas. Williams *et al*. (2009) grouped these processes into four environmental filters, with two of them (habitat transformation and fragmentation) present in most ecosystems, whereas human preferences and specific urban environmental conditions (see Section 1 of this book) are unique to cities. Although these processes are acting simultaneously and it is difficult to isolate single drivers of individual species losses or gains (Williams *et al*. 2009), a number of processes affecting plant diversity patterns will be described further in an attempt to increase our understanding of plant diversity patterns of urban ecosystems.

3.2.3.1 Habitat transformation and fragmentation

There is a general consensus that transformation of habitats into other land-uses (including urban development), and the additional loss of habitat due to fragmentation, have devastating effects on indigenous plant diversity (De Candido 2004; Pauchard *et al*. 2006; Williams *et al*. 2009). Studies from Europe have indicated that cities were preferentially placed in pre-existing biodiversity hot spots (Kühn *et al*. 2004), emphasizing the importance of understanding habitat transformation and fragmentation as powerful drivers of urban biodiversity.

Landscape transformation in terms of land-use and land cover changes from natural to urban land do not only drive biodiversity changes but also aspects such as climate change on local (urban heat island), regional, and even global levels (Grimm *et al*. 2008). Global warming may also influence plant diversity in a different manner in that it enhances gardening and urban landscaping, which in turn may have negative (source of plant invasions) and positive (increase of carbon sequestration) consequences (Niinemets & Peňuelas 2008). There are several examples of the effect of urbanization on different types of natural habitats. Wetlands are often transformed through draining and filling for residential and industrial development; for example, Pauchard *et al*. (2006) have indicated that 23 per cent of the original wetlands in Concepción, Chile, have been lost. The relative ease of development on grasslands in combination with an ignorance of the importance of the conservation of these highly threatened ecosystems (Cilliers *et al*. 2008) have also seen an extensive transformation of grasslands mainly into residential areas, as indicated for two cities in the Grassland Biome of South Africa over a 25-year period (Table 3.2.1). Even formal urban open spaces are affected by habitat transformation, as indicated by 'upgrading' of an urban park in New York City (USA), which resulted in the extirpation of over 25 per cent of the indigenous species over a 50-year period, a loss of 2.9 species per year (De Candido 2004).

In general, habitat fragmentation is an extensive topic and according to Lindenmayer and Fischer (2006) 'it is frequently used as an umbrella term for many ecological processes, patterns of vegetation cover, and biotic responses that accompany alteration of landscapes by humans'. The term 'habitat fragmentation' is often used so widely that it

Table 3.2.1 Transformation of natural grasslands into different land-use areas over a period of 25 years in two cities in the grassland biome of South Africa.

	% Transformation	Land-use replacing natural grasslands (% of total transformed areas)			
		Residential	Industrial	Commercial	Other
Potchefstroom	13.27	84.02	3.75	0.09	12.14
Klerksdorp	26.18	81.91	7.80	1.36	8.93

becomes vague and ambiguous. Many studies, for example, do not distinguish between 'habitat loss' and 'habitat isolation' or between 'habitat loss' and 'indigenous vegetation loss', all different processes with unique impacts (Lindenmayer & Fischer 2006). Patch dynamics, connectivity, and edge effects are themes often discussed under the umbrella of habitat fragmentation, including in urban areas. It was shown in several studies that urbanization leads to a decrease in patch fragment sizes but an increase in the number of patches, which is expressed as a high proportion of edge habitats (Godefroid & Koedam 2003; Grimm *et al*. 2008; Hamberg *et al*. 2009). Alberti (2005) referred to several studies indicating that the size and shape of patches and their edges can affect the habitat, resource availability, and competition of species. It was further reported that indigenous plant species richness in isolated patches declines with patch size due to habitat loss and interspecific interactions (see Dunn and Heneghan, Chapter 2.4).

The edges of fragments may experience different environmental conditions, such as microclimatic differences and higher human impacts, than the interior of these fragments, which may cause changes in plant species composition and abundance. Processes such as competition, herbivory, and seed dispersal may also be different at the edges, and all these changes form a zone of influence that are collectively called the edge effect (Godefroid & Koedam 2003; Hamberg *et al*. 2009). Edge effects are one of the most studied concepts in ecology as they form areas of transition, contact, or separation between contrasting landscape elements (Cadenasso *et al*. 2003). In urban areas, edge effects are not often studied, despite the fact that natural remnant edges are very abrupt in these areas and are often bordered by artificial structures such as buildings or asphalt (Hamberg *et al*. 2009), creating specific environmental conditions. Studies on edge effects in small fragments of urban forests have shown that the structure of forest edges (closed or open) and their composition (broad-leaved or coniferous) influences the composition of the understorey vegetation (Hamberg *et al*. 2009). Godefroid and Koedam (2003) have also identified several edge-oriented species in an urban forest in Brussels, but failed to describe any species that were interior oriented. In a rare comparative study on linear native grassland remnants in urban and rural areas in two countries in the southern hemisphere, Cilliers *et al*. (2008) have indicated that these grasslands respond differently to fragmentation in urban and rural areas. This has important implications for the management of fragmented grasslands in urban areas as the various exogenous disturbances could require additional landscape management practices such as removal of above ground biomass by increasing the frequency of mowing and burning (Cilliers *et al*. 2008).

Connectivity between different fragments in the form of dispersal corridors is a highly controversial topic and Lindenmayer and Fisher (2006) have proposed that a distinction should be made among habitat, landscape (physical), and ecological (processes) connectivity of patches. Several studies have, however, indicated that species composition and structure of urban fragments do not differ that much from those in natural areas, despite a lack of connectivity or maybe because of some subtle degree of connectivity. Heckmann *et al*. (2008), for example, found no decline in native plant species richness or significant change in forest structure in urban forest fragments in the Sierra Nevada of California (USA), but noted that non-native species loads could increase and tree density and cover could decline in future as disturbances increase with growing urban development. Hahs and McDonnell (2007) have shown by means of ordination that only small differences exist in species composition of remnant patches of grassy woodland communities in urban areas and in surrounding natural areas in Australia, indicating that these communities are relatively resilient to landscape changes associated with urbanization. The opposite scenario was found in grassland fragments in South Africa, where plant species composition was distinctly different among urban, suburban, and rural grassland fragments (Fig. 3.2.1). The main determining factors for these differences are management, as urban and to a lesser degree suburban grassland remnants are intensively managed in South Africa. Mowing and trampling are, therefore, important anthropogenic disturbances affecting plant species composition, while a natural disturbance of grasslands such as burning is highly restricted in urban areas (Cilliers *et al*. 2008). The

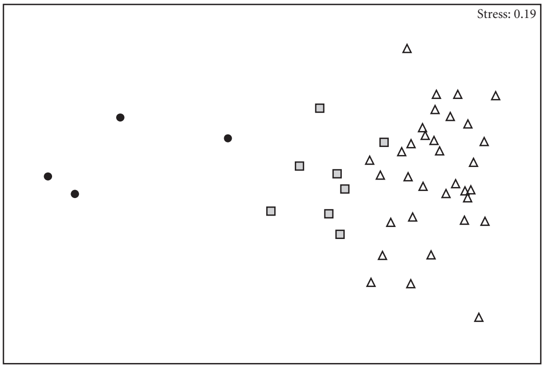

Figure 3.2.1 Composition of all plant species for each sample plot within the two dimensional NMDS ordination space of fragmented grasslands in Klerksdorp, South Africa. Each point within the ordination space represents a sample plot within a remnant grassland patch in urban (dark circles), suburban (shaded squares), and rural (open triangles)

complex effect of disturbances on urban plant diversity will be discussed later.

In exploring the effects of habitat fragmentation on vegetation, the focus should not only be on assemblages of species and aspects such as species diversity (as indicated above) but also on single species (Lindenmayer & Fischer 2006). Plant species that will typically be under threat through fragmentation are those with limited dispersal capacity, low seed production, or no seed bank, but also those dependent on mutualisms, such as specialized pollinators and those with high inbreeding depression (Williams *et al*. 2009). Very little is known, however, about the effect of habitat fragmentation on the species level in urban areas. This is surprising, considering the huge effect of urban sprawl (excessive urban development) on natural plant populations (Roberts *et al*. 2007). Pollinators may adapt their foraging behaviour in patchy plant populations in urban areas, in response to the amount of available resources through lower visitation rates but longer flower visits, than in large plant populations in natural areas (Andrieu *et al*. 2009). Consequences of these modifications in plant–pollinator interactions may include lower plant fecundity, reduced plant fitness, and eventually plant trait evolution, and should be explored further (Andrieu *et al*. 2009). Additionally, studies on 'urban forms' of endangered plant species may enhance long-term conservation and management of these species. Urban plants could contain genetic variations that were lost from protected areas and could therefore 'increase genetically effective population sizes, provide stepping-stones for gene flow between nearby populations, and increase the spatial extent and size of populations', according to Roberts *et al*. (2007). Conversely, though, urban plants from other parts of the species range or from nursery-propagated individuals or from hybrids could threaten the genetic integrity of surrounding populations (Roberts *et al*. 2007).

Most of the studies mentioned in this section have also emphasized the importance of historical factors on biodiversity, such as different land-uses from the past. These and other legacy effects are important drivers of urban plant diversity and will be discussed later.

3.2.3.2 Urban environmental conditions: disturbances and adaptations to city life

Anthropogenic disturbances are an inevitable part of urban ecosystems, as was mentioned earlier. Some of the most obvious types of disturbance regimes in urban areas include trampling of vegetation and soil compaction (Sarah & Zhevelev 2007; Hill & Pickering 2009), soil pollution (Madrid *et al*. 2004), alien plant invasions (Sullivan *et al*. 2005), and mowing/cutting of managed landscapes (Greller *et al*. 2000). Additionally, urban development may reduce or increase the magnitude, frequency, and intensity of natural disturbances, affecting important ecosystem processes (Alberti 2005).

Disturbance frequency and intensity are major parameters that determine species composition in an urban environment (Dana *et al*. 2002). The overall effect of human impact, sometimes described as the hemeroby concept, probably has the biggest effect on species responses (Chocholoušková & Pyšek 2003). Disturbance intensity is important and Zerbe *et al*. (2003) have indicated that in a mosaic of land-use patterns, the areas with moderate frequencies or intensities of disturbance have positive effects on habitat and overall species diversity in accordance with the intermediate disturbance hypothesis (IDH). Kowarik (1995) has, however, stressed that the IDH is only supported if native species are considered, because aliens increase in sites subjected to high disturbance levels.

Disturbance, in combination with soil characters and microclimate, shapes the composition of plant communities, has an influence on species richness, and determines which life forms are present (Fanelli *et al.* 2006; Maskell *et al*. 2006). It is, therefore, not only disturbances but also stress factors to the plants, especially in terms of restriction of photosynthesis, which are important. Environments that induce stress and disturbance are therefore inaccessible to plants. However, this exclusion is overcome through different adaptations to such harsh environments and is best described by different plant life strategies: competitive (C), ruderal (R), and stress-tolerant (S) species. Depending on the environment, different types of life strategies can be defined (CS, CSR, RS, CR) (Grime 2002). Many parts of urban areas are considered as harsh environments and spontaneous vegetation in these areas will therefore consist of a species composition dominated by a specific set of life strategies. Domestic gardens will contain many R-strategists, as this strategy will favour quick seed set before weeding. Roadsides with high concentrations of salt will be characterized by a flora dominated by S-strategists able to tolerate physiological stress. Water channels will again be dominated by C-strategists that have the ability to grow fast and out-compete other species for water and light.

In urban ecosystems the multiple ecological factors that shape the composition of communities on a small scale (e.g. parks, gardens, walls, fragmented natural areas, and wastelands) are interacting, not independent variables (Dana *et al*. 2002; Fanelli *et al*. 2006; Lundholm & Marlin 2006; Bornkamm 2007; Ehrenfeld 2008). At a finer scale these variables can be grouped into specific interactive combinations to explain plant diversity patterns (Fig. 3.2.2) that may differ from the general patterns described earlier for urban floras. Soil characteristics and microclimate are major drivers of plant diversity and will be discussed further. Inherently the habitat template determines soil and microclimate and it is therefore important to consider the replication of habitat analogues by built forms and the creation of ecologically novel habitat to predict the species composition of urban communities (Dos Reis *et al*. 2006; Lundholm & Marlin 2006). The Urban Cliff Hypothesis predicts that plant species that spontaneously colonize rock outcrop habitat types, also have the ability to colonize habitat analogues with similar environmental conditions in the built environment such as walls, pavings, and shallow and compacted soils (Larson *et al*. 2004, quoted by Lundholm & Marlin 2006).

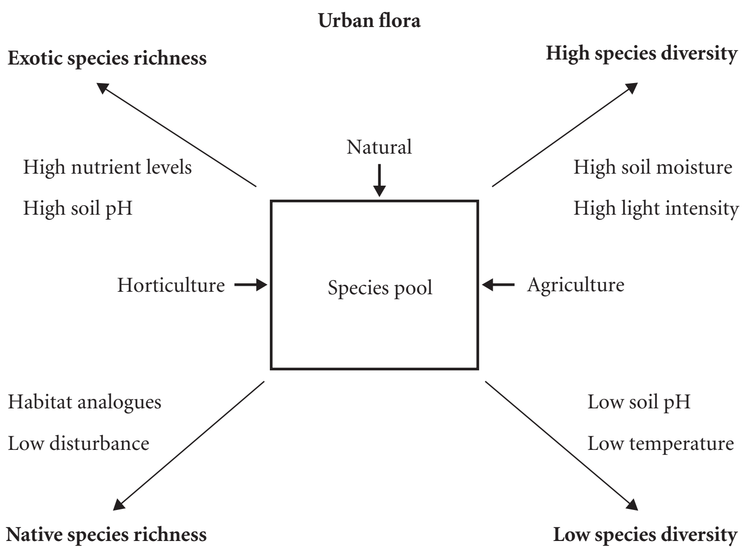

Figure 3.2.2 Schematic representation of patterns in plant diversity as an outcome of the main environmental processes at a fine scale

When soil characteristics are considered, then nutrient content, moisture, pH, and pollutant concentrations of soils are the major factors correlated with long-term changes in plant species distributions in urban environments (Van der Veken *et al*. 2004; Fanelli *et al*. 2006; Godefroid *et al*. 2007). In terms of nutrient status, the higher levels in urban environments (Shaw & Reeve 2008) favour generalists (Vallet *et al*. 2008), often exotics that have higher demands for nitrogen especially (Chocholouškova & Pyšek 2003). In contrast, species diversity of indigenous habitat specialists declines with increasing eutrophication (Van der Veken *et al*. 2004). Higher nutrient status sees the establishment of vigorous herbaceous species that out-compete tree seedlings and could slow the process of succession towards forest in the northern hemisphere (Bornkamm 2007). Vegetation strata in wetlands also respond strongly to nutrients (Ehrenfeld 2008).

Soil moisture, in combination with other factors, also shapes the composition of plant communities in different patches, has an influence on species richness, and determines which life forms are present (Fanelli *et al*. 2006; Lundholm & Marlin 2006; Godefroid *et al*. 2007). Soil water content is negatively affected by pressures of human movement (e.g. soil compaction) (Sarah & Zhevelev 2007) and hydrogeomorphic alterations of water resources (e.g. draining of wetlands) (Ehrenfeld 2000a). Although it has been shown that species in the city have lower demands for moisture (Chocholouškova & Pyšek 2003), water availability is a major parameter that determines species composition in an urban environment (Dana *et al*. 2002). Indeed, Fanelli *et al*. (2006) established that species diversity increases as soil moisture increases.

Species composition in urban environments such as wastelands and wetlands respond strongly to soil pH (Godefroid *et al*. 2007; Ehrenfeld 2008). Soils near intensively used roads are less acidic than adjacent pristine soils (Shaw & Reeve 2008). Urban forests have a higher soil pH than rural ones along an urban–rural gradient (Vallet *et al*. 2008). Variations in pH on a small scale are usually linked to pollution. Although large amounts of pollutants are present in urban soil, they do not readily affect plants, as they are accumulated by organic material (Madrid *et al*. 2004). The effects of pollutants could, however, become more pronounced in the future as microenvironments with high human visitor pressure have significantly lower soil organic matter (Sarah & Zhevelev 2007).

Microclimate is probably best understood as the urban heat island effect, which could, for instance, result in the advancement in flowering time, which in turn could block or limit gene flow among the metapopulation and urban communities and enhance species polarization (Neil & Wu 2006; Luo *et al*. 2007). The microclimatic effect is even further complicated as compositional changes in communities may correspond to a microclimatic gradient based on adjacent land-use (Godefroid & Koedam 2003). Differences in vegetation among patch types are often related in part to the microclimate created by habitat analogues (Lundholm & Marlin 2006). The best understood microclimatic factors that influence species composition are temperature (Fanelli *et al*. 2006) and light intensity (Lundholm & Marlin 2006; Godefroid *et al*. 2007), although air humidity also shows a significant influence on the presence of some species (Godefroid *et al*. 2007).

Temperature as a case in point shapes the composition of plant communities, in combination with other factors, and has an influence on species richness to determine which life forms are present (Fanelli *et al*. 2006). Species in the city have higher demands, especially for temperature (Chocholouškova & Pyšek 2003; Godefroid *et al*. 2007), and the results of recent long-term studies have shown that frost selectively suppresses or even removes species from communities in urban environments (Bornkamm 2007). Temperature is therefore a highly significant predictor for the number of total aliens, reflecting the origin of aliens from warmer areas (Pyšek 1998).

3.2.3.3 The human factor

The unique environmental conditions in urban areas, as described above, are important for the establishment of plants, as they govern resource availability. It was, however, indicated in several studies that human choices and management in urban areas have modified the resource availability–diversity relationships derived from classical ecological theory, through the removal of limitations in natural resources (e.g. water, nutrients, temperature)

(Martin *et al*. 2004; Kinzig *et al. 2005*; Hope *et al*. 2006). Humans have created a whole new set of anthropogenic habitats (parks, pavements, gardens, lawns, roads, railways, roofs, etc.) for plants in a combination of horticultural plantings and non-native invasives (Williams *et al*. 2009). Human choices in terms of plants in urban areas do, however, not only include the creation of vegetated landscapes (what and where to plant) but also the preservation of existing vegetated areas (what kind and how much of existing fragments should be conserved) and, equally importantly, indicate that the choices can be personal or institutional (Martin *et al*. 2004; Hope *et al*. 2006). Personal choices refer to management of private green spaces by urban residents (called bottom-up factors by Kinzig *et al*. 2005) as well as the impacts of different role players on municipal green spaces, which are largely governed by institutional choices through policy issues enforced by local government (called top-down factors by Kinzig *et al*. 2005). South African studies comparing diversity of urban domestic gardens with other land-uses, including natural areas, emphasized the importance of the heterogeneity of cultivation preferences and habitat types in that gardens have a low alpha-diversity, but high gamma-diversity, resulting in a high turnover of species between gardens (beta-diversity) (Cilliers *et al*. in press). Additionally, the public, planners, managers, scientists, and policy-makers mostly have different perceptions in terms of the value of urban open spaces. This variability in the approaches of different stakeholders poses a tremendous challenge to the design, planning, and management of urban areas that will be discussed in Section 5 of this book.

Understanding the relationships between social structure and vegetation in urban areas is extremely important for the development of sustainable management plans for urban open spaces. Studies exploring these relationships have focused on vegetation cover (Iverson & Cook 2000), plant diversity (Hope *et al*. 2006), and vegetation structure (Grove *et al*. 2006). Household income and household density were found to correlate with urban forest cover, with wealthier neighbourhoods having higher tree cover than poorer areas (Iverson & Cook 2000). Hope *et al*. (2006) confirmed the link between socio-economic status and vegetation in residential areas by describing the so-called 'luxury effect' as the tendency of wealthier people to inhabit areas with higher plant diversity, either by creating them or by selecting neighbourhoods with naturally high diversity. Using remotely sensed data and an urban land cover classification system, Grove *et al*. (2006) have shown that vegetation structure and vegetation management vary in different neighbourhoods. They discussed the complexity of explaining the processes behind these relationships, and also mentioned the differences in vegetation management by specific lifestyle groups driven by human capital (access to private financial resources) and social capital (ability to work collectively or access to government resources) (Grove *et al*. 2006).

Removal of limitations from natural resources for the establishment of plants is nowhere as evident in the different land-uses in urban areas as in gardens. Common garden practices are aimed at the facilitation, establishment, growth, and dispersal of plants by 'diminishing dispersal limitations, amelioration of environmental and biological stresses and improving winter survival' according to Niinemets and Peñuelas (2008). In an overview on urban domestic garden research, Cilliers *et al*. (in press) indicate that research in developed countries focuses on gardens as resources for biodiversity and ecosystem functioning, whereas the emphasis in developing countries is more on subsistence in terms of urban agriculture and agroforestry. There is, however, limited information on specific processes driving species diversity of gardens. Cultural background seems to be an important driver for selecting a specific garden type, such as ornamental or food gardens. Head *et al*. (2004) indicates that gardens of migrants to Australia are clearly different as traditions from their homelands were carried on, but a focus on food productivity seems to lose emphasis in the next generation. Socio-economic status of urban residents, expressed in terms of housing density, household income, educational level, and unemployment rate are positively correlated with total plant diversity, indigenous plant diversity, and tree cover (Kirkpatrick *et al*. 2007; Cilliers *et al*. in press). In a study in a rural settlement in South Africa an opposite trend was, however, described, with residents with a lower socio-economic status living in areas with a higher native plant diversity (Cilliers *et al*. in press). This emphasizes the importance of a combination of management issues (clearing of land before construction or not), sociocultural issues (lack of resources to clear the land and use of native plants for medicinal purposes and grazing), and the effect of legacy (history of land-use in agricultural or natural areas) in determining urban plant diversity.

Legacy effects, described by Hope *et al*. (2006) as the inherited present day situation due to past events, form an important template in determining urban plant diversity. Two large-scale historical events that determined urban plant diversity were described by Hope *et al*. (2006) in Phoenix, Arizona, USA; namely past land-use, as parts of the city were developed on former agricultural land, and the policy on water distribution. Concerns over the amount of water available to residents for gardening purposes led to a change in water policy in 1980s in Arizona. Flood-irrigation was stopped and incentives were provided to convert to drip-irrigation, restricting the amount and type of vegetation used in gardens (Hope *et al*. 2006). The change in policy also initiated a tendency towards 'xeriscaping'—use of drought-adapted plants that are smaller, allowing more species per yard, and less emphasis on lawns (Hope *et al*. 2006).

Another example of the effects of legacy on urban plant diversity comes from the pre-1994 apartheid era of racial segregation in South Africa. The legacy of town development during that period was characterized by large numbers of alien plant species entering parks, nurseries, and government-managed grounds in cities, towns, and suburbs inhabited by people from European origin. Gardening in these areas was also driven by European ideas, which was in stark contrast to the home garden system of the native African cultures. The gardens of the latter were based on useful plants (medicinal, food, spiritual, hedging) and did not necessarily address any aesthetical issues. Figure 3.2.3 shows a historical gradient of exposure of settled areas to European garden culture; it is clear that the legacy of the mentioned segregation is still evident in the floras of different settlements. Mbazwana is regarded as deep rural, a somewhat isolated town in the heart of the Zulu Kingdom. Esikhawini is a rural village but lies in the vicinity of a large industrial city and is less than 30 years old. This level of exposure to European culture is the point where the percentages of native and exotic species are equal. Ganyesa is a town almost exclusively occupied by the Batswana ethnic group, that has been subjected to extensive development, despite being rural, and has a rich history of European farming influence. The percentage of indigenous and exotic species is nearly equal at this level of exposure to European culture. The black township, Ikageng, lies on the outskirts of a city, with most inhabitants working in a city which is dominated by people of European origin, explaining the increase in alien and decrease in native and indigenous species. Potchefstroom is an old (120 years), typical colonial city, with decades of active landscaping and introduction of alien species. Although the patterns are clear, more detailed studies are needed focusing on specific aspects of the different cultures in the same or in different settlements,

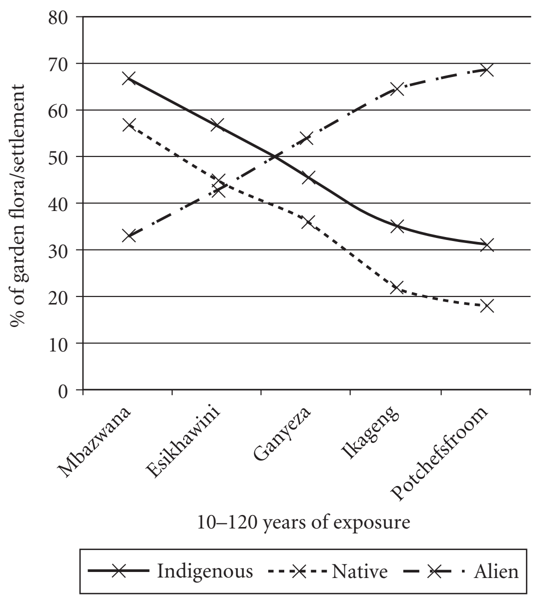

Figure 3.2.3 Comparison of plant diversity patterns along a historical gradient representing exposure of different settled areas in South Africa to European gardening culture (indigenous refers to species occurring naturally in South Africa, native refers to indigenous species occurring in natural areas surrounding the settled areas, alien refers to species introduced from other countries)

and a more refined characterization of their socio-economic status as a driver of plant diversity.

3.2.3.4 Introduced species

All the processes discussed earlier may promote invasions of non-native species, but this should also be discussed under a separate heading due to the complex nature of species invasions. Ehrenfeld (2008) has, for example, shown that the invasions of non-native species in urban wetlands are due to a complex combination of intrinsic site properties (e.g. soil) and various direct and indirect human impacts (inside and around the wetland) as well as the individualistic response of each species to all the different factors. In a study in urban riparian habitats, Maskell *et al*. (2006) emphasized that extrinsic factors such as eutrophication and disturbance can also be associated with certain native species that may also invade and dominate urban riparian areas. Two processes describing the diversity of non-native species in urban riparian habitats have been described by Maskell *et al*. (2006); namely invasion of disturbed fertile habitats by native and non-native species to form communities dominated by weedy annuals, and invasion and domination by competitive non-native species in less disturbed areas where native species diversity declines.

As indicated earlier, it is generally accepted that urban areas form the most important centres of spread of non-native species (see Dunn and Heneghan, Chapter 2.4). Propagules of non-native species are intentionally introduced as horticultural objects, crop plants, or medicinal herbs, but are also unintentionally dispersed by the increase of vehicle transport, by animals, and by dumping of garden waste (Kent *et al*. 1999; Godefroid 2001; Sullivan *et al*. 2005). In Brussels, Godefroid (2001) described anthropogenic influences, such as eutrophication, drainage, deforestation, soil enrichment with construction rubble, and the urban heat island, that create environmental conditions favouring non-native species. Studies in South Africa have shown that non-native species have the potential to become naturalized with time, even after decades in domestic gardens, as they only require a change in human behaviour and access to favourable habitat conditions (Siebert 2009). Urban conditions in general do not favour the spread of most native species, and some studies have indicated the dominance of non-native species over native species in soil seed banks in urban areas (Kostel-Hughes *et al*. 1998; Stewart *et al*. 2004).

The replacement of native plant diversity by generalist non-natives may also increase biotic similarity between plant communities in different cities; this is termed biotic homogenization and has been studied at different organizational, spatial, and temporal scales (McKinney 2006). Fears of complete globalization of the Earth's biota have for now been augmented because, according to La Sorte *et al*. (2007), biogeographically defined anthropogenic and historical factors are still driving the ability of non-native species to become widespread. The specific ecological and evolutionary consequences of biotic homogenization are, however, difficult to describe. Olden *et al*. (2004) proposed studies on three distinct forms of homogenization, namely genetic (threat to the integrity of endemic gene pools and reduced levels of genetic variability), taxonomic (increase in compositional similarity due to invasion of winning species and extirpation of losing species), and functional homogenization (winners and losers are largely determined by interactions between environmental characteristics and species traits).

3.2.4 Conclusions

In this chapter various processes have been described that may determine the distinct patterns of urban plant diversity. Emphasis was placed on the complexity of the processes as well as the problem of linking certain processes to specific patterns, indicated by the several cross-references between the discussions of the different processes. Williams *et al*. (2009) also warned against broad generalizations in terms of processes, as the regional setting of a specific urban area may also affect urban floras. Nevertheless it is important to move our research attention from mere descriptions of plant diversity patterns to more mechanistic studies (Shochat *et al*. 2006) to determine what, why, and how plants are affected and also explain the ecological and environmental consequences of these effects (Neil & Wu 2006). Although the processes are complex, Grimm *et al*. (2008) clearly state that cities do not only present the problems, 'but also the solutions to the sustainability challenges of an increasingly urbanized world'. Williams *et al*. (2009) argue that a more mechanistic understanding of urban floras would depend on more extensive and more representative studies of the selection pressures of specific environmental filters. These filters could lead to species losses and gains, alterations of plant functional traits, changes to phylogenetic distributions of species within the urban flora, and could also eventually act as agents of natural selection of plant populations in urban environments (Williams *et al*. 2009).

To be able to better understand the underlying processes in determining biodiversity in an increasingly urbanized world, plant ecologists should focus their attentions more on urban vegetation, but then within an integrated natural–human system, according to Pickett and Cadenasso (2008), following a spatially explicit landscape ecological approach (Neil & Wu 2006) and recognizing the importance of scale on the effects of the different processes (Alberti 2005). Understanding these processes contributes to ecological theory, even changing the discipline of ecology, but it also leads to a more sensible integration among plant ecology, other environmental sciences, and urban design (architecture, landscape architecture, civil engineering, and urban planning) (Grimm *et al*. 2008; Pickett & Cadenasso 2008). The implementation of ecological urban design, based on plant ecological studies, has several benefits, especially improving the quality of life and human health of urban residents.

CHAPTER 3.3

Effects of Urbanization on the Ecology and Evolution of Arthropods

Johan Kotze, Stephen Venn, Jari Niemelä, and John Spence

3.3.1 Introduction

The role of the 'little things that run the world' (Wilson 1987) is challenged by the human enterprise in urban areas. High-density human habitation is associated with loss, fragmentation, isolation, and pollution of natural habitats, introduction and spread of exotic species, accumulation of waste products, and changes in climatic, edaphic, and hydrological processes (Frankie & Ehler 1978; McIntyre *et al*. 2001). All of these impacts affect arthropod communities and their contributions to ecosystem function. On the other hand, this diversity of human actions also creates opportunities for arthropods to thrive in cities.

Because they fill important functional roles in ecosystems, including decomposition, pollination, nutrient cycling, foodweb interactions, and biological control, arthropod communities reflect urban environmental quality. Their short generation times and rapid responses to environmental change make arthropods especially well-suited for studying the biotic effects of urbanization. In addition, many arthropods have educational and aesthetic values to urban residents (Andersson *et al*. 2007) that increasingly balance negative perceptions of arthropods as pests in well-educated societies.

Frankie and Ehler (1978) called for the establishment of 'urban entomology' as an independent discipline, but the field has remained incompletely developed (McIntyre 2000), although its importance is recognized, for example for urban pest management (Robinson 1996). Nonetheless, urbanization reduces biodiversity (Davis 1978; Connor *et al*. 2002) and thus has major implications for ecosystem function and the provision of ecological services, including a sense of human well-being. In hoping to stimulate research on non-pestiferous urban arthropods through this chapter, we review how urban arthropods are affected by: 1) habitat fragmentation, 2) habitat changes along the urban–rural gradient, 3) uniquely urban environments (such as domestic gardens, ruderal land, etc.), and 4) natural selection in urban environments. Based on this review, we consider major issues relevant to conservation of urban arthropods. Because of our collective experience with ground beetles, we draw extensively on that literature for examples, using studies of other arthropod species to amplify the generality of conclusions.

3.3.2 Arthropods in the fragmented urban landscape

Greenspaces exist as fragments in urban landscapes and, as such, are small, isolated, and disproportionately composed of edge habitat (see Dunn and Heneghan, Chapter 2.4). As predicted by the theory of island biogeography, the number of arthropod species decreases in such habitats as their area shrinks in urban landscapes (Faeth & Kane 1978; Helden & Leather 2004), and thus simple habitat loss accounts for elimination of many species in urban landscapes (Connor *et al*. 2002). However, for carabid beetles, habitat area predicted species richness more poorly than did distance to the city centre (Weller & Ganzhorn 2004) or habitat quality (Hartley *et al*. 2007). In fact, urban fragments may be characterized by high populations of some taxa (e.g. Niemelä & Spence 1991; Bolger *et al*. 2008). These observations suggest that the urban greenspace is rich in biodiversity but could be better managed to achieve conservation objectives as the arthropod assemblages in small habitat fragments are greatly altered.

Arthropod species with different ecological requirements, life histories, and morphological traits respond differently to habitat fragmentation. Most species that flourish in urban settings are habitat and dietary generalists and highly dispersive (e.g. Šustek 1987; Kotze & O'Hara 2003). Species strongly associated with particular natural habitats decline in urban environments, but those associated with edge habitat thrive (Venn *et al*. 2003; Christie & Hochuli 2005). Thus, urban assemblages are homogenized, mainly through the loss of specialists (Hartley *et al*. 2007; Niemelä & Kotze 2009). Many habitat specialists depend on patchily distributed resources, the occurrence of which is reduced, altered in scale, or absent altogether in urban areas. For example, saproxylic species are scarce in urban forests that lack dead and decaying wood (Fayt *et al*. 2006; Sörensson 2008). Also, open habitat arthropod assemblages include specialist species, which are dependent on the implementation of appropriate management strategies (Hartley *et al*. 2007), as do those of aquatic habitats (Moore & Palmer 2005).

Monophagous and univoltine species are more susceptible to habitat loss, while polyphagous and multivoltine species tend to be more prevalent in urban assemblages (Blair & Launer 1997; Sharma & Amritphale 2007). For example, the number of host plant-specialist and univoltine butterfly species was positively correlated with area of oak woodland, whereas multivoltine and host plant-generalist species were more common in smaller fragments with lower percentages of oak woodland (Niell *et al*. 2007). Such shifts in pollinator communities could lead to failure of seed set, affecting plant communities in the urban matrix (Pauw 2007). The greater observed abundance of generalist predators in urban environments may affect predator–prey interactions (Shrewsbury & Raupp 2006) and consequently pest populations, although the general implications of such effects remain poorly understood (Symondson *et al*. 2003).

Arthropod larvae frequently require different habitats than adults. Understanding the consequences of such effects in urban areas is particularly important for managing human disease vectors (Lounibos *et al*. 2002) but can also have important implications for overall biodiversity. For example, hoverfly assemblages in urban areas are predominantly composed of species whose larvae are aquatic or predatory, whereas species with saproxylic or sap-feeding larvae are scarce or absent (Bankowska 1980).

High dispersal ability is an important prerequisite for persistence in urban environments (Sharma & Amritphale 2007). Non-flying specialist carabid beetles inhabiting brownfields, for example, are negatively affected by habitat isolation, especially if nearby sites are in early successional stages or otherwise unsuitable (Small *et al*. 2006). Wing-dimorphic species persist better in highly altered environments, apparently because long-winged individuals are able to disperse among suitable habitat patches while short-winged individuals survive and reproduce well once a population is established (Kotze & O'Hara 2003; Hartley *et al*. 2007). The wing-dimorphic European carabid *Pterostichus melanarius*, for example, has successfully invaded urban habitats across Canada, and long-winged individuals are disproportionately common at the edges of local distribution ranges (Niemelä & Spence 1991).

3.3.3 The urban–rural gradient

Arthropod assemblages of urban habitats have special characteristics. Some taxa thrive in urban habitats (Frankie & Ehler 1978), whereas others are virtually absent. Community responses have been studied across urban–suburban–rural gradients, ranging from highly fragmented, disturbed, and isolated urban core habitats, through suburban areas, to relatively undisturbed rural environments (McDonnell & Hahs 2008). This approach has been particularly well developed using forest-dwelling carabid beetles. In a number of cities (Debrecen, Hungary; Helsinki, Finland; Hiroshima, Japan; Sofia, Bulgaria), the abundance of beetles increased from the city centres to the rural surroundings, but this pattern was less consistent in Birmingham (England) and Sorø (Denmark) and varied from year to year in Helsinki (Niemelä & Kotze 2009). In contrast, overall carabid abundance in Edmonton, Canada, was highest in the city centre, reflecting large populations of introduced species, especially the above-mentioned *P. melanarius* (Niemelä & Kotze 2009). In most of these cities, carabid species richness also increased from urban to rural areas. However, the pattern again varied temporally in Helsinki and was not observed in either Debrecen or Sorø (Niemelä & Kotze 2009). Thus, it appears that there are some generalities in the responses of carabid beetles to urbanization along urban-to-rural gradients but also factors not consistently associated with position on the urban–rural gradient affect arthropod assemblages.

Other arthropod groups have shown similar responses to urbanization. For example, gall-inhabiting moths had generally lower species richness, larval density, and larval abundance at sites closest to the city centre than at sites further away (Rosch *et al*. 2001). Species richness and abundance of arthropods were lower in patches of desert within the urban core than in patches outside Phoenix, USA (Rango 2005). As in other stressed and disturbed systems, urban assemblages show reduced evenness, and certain species gain dominance. For instance, dominance of the carabid species *Pterostichus madidus* increased from less than 10 per cent in rural sites to 60–85 per cent in the most urban sites within Birmingham, UK (Sadler *et al*. 2006), and exotic species drive the same pattern in Edmonton (Hartley *et al*. 2007; Niemelä & Kotze 2009). However, opportunistic species were not more dominant in Brussels, Belgium (Niemelä & Kotze 2009).

Body size, habitat affinity, and flight ability appear to be related to arthropod responses to urbanization. Small-bodied carabid species dominated urban sites in several European cities whereas large-bodied species were more common at rural sites; however, in Hiroshima and Edmonton no such differences were documented (see Niemelä & Kotze 2009). Body size patterns of Polish carrion beetles decreased towards the city centre whereas those of predatory beetles remained unchanged (Ulrich *et al*. 2008). Carabid species associated with forests are more common in suburban and rural than in urban sites (Niemelä & Kotze 2009), whereas generalist and open-habitat species occur more frequently at urban sites. Urban assemblages are characterized by species capable of flight (either macropterous or wing-dimorphic), whereas strictly flightless species tend to be more common in rural areas (Niemelä & Kotze 2009).

In conclusion, studies on arthropods, especially carabid beetles from cities across the world, suggest reasonably consistent community-level responses to urbanization. Abundance and richness of indigenous species tend to increase from city centres to the rural surroundings. Urban environments are characterized by a few dominant species and, in many cases, these are invasive, exotic species. The proportion of large-bodied species and those requiring forest habitat usually decreases towards city centres, and thus effective biodiversity management in the urban forest will differ from that in the less-developed landscape. Most species of urban and/or suburban environments are capable of flight, whereas species unable to fly are more common in rural environments.

An appropriate challenge for future work will be to unravel the processes that generate the species responses discussed above. These processes include habitat loss and fragmentation, leading to variation in size, location, and disturbance of urban habitats (Sadler *et al*. 2006; Hartley *et al*. 2007). For example, forest remnants smaller than 8 ha were insufficient to support populations of large-bodied woodland carabids (Sadler *et al*. 2006), and this limitation will affect the composition of assemblages when smaller, albeit reasonably natural, sites are set aside to meet biodiversity objectives. Of course, species vary with respect to habitat affinity (e.g. forest species, open habitat species, etc.), and this can have interesting biodiversity impacts along the urban–rural gradient (Niemelä & Kotze 2009). Large, flightless, and specialist woodland species are especially susceptible to urbanization because of their long life spans, lower reproductive rates, specialized niches, and limited dispersal potential (Kotze & O'Hara 2003). Some species, such as the carabid *P. melanarius*, may become dominant in disturbed urban forests (Niemelä & Spence 1991; Niemelä & Kotze 2009), whereas others, such as *Platynus mannerheimii*, virtually disappear from these forests (Venn *et al*. 2003), presumably due to lack of suitable conditions.

We suggest that additional work to quantify the concepts of 'disturbance' and 'urbanization' along the gradient (e.g. Hartley *et al*. 2007) would be useful. The gradient could be defined, for example, in terms of the proportion area covered by buildings, pavement, and various habitat types. The number of vehicles, residents, and pedestrians, and direct human effects such as trampling and waste, could also be quantified and used analytically with more traditional variables, such as climate, vegetation, and soil variables to explain urban faunal patterns. For instance, an index developed by Weeks *et al*. in 2003, and tested by McDonnell and Hahs (2008), is based on combining (1) the density of people working in non-agricultural industries obtained from census information, and (2) the proportion of the landscape covered by impervious surfaces obtained from satellite imagery. The advantage of such a broad but quantifiable index of urbanization is that it provides a common measure for comparing patterns of urbanization among cities across the world. Development and use of landscape indices (e.g. size and shape of various habitat patches) to characterize and quantify urban environments might further illuminate how urbanization affects urban species distributions. Although habitat isolation appears to influence the structure of urban species assemblages (Sadler *et al*. 2006), comparative studies based on common metrics of isolation are too rare to explore the generality of this effect.

3.3.4 Unique urban habitats

Cities include unique habitats (e.g. brownfields, domestic gardens, parks, roadside verges) that are subject to high levels of anthropogenic effects (e.g. fertilizers, pesticides, trampling, noise, and light) (see Pauleit and Breuste, Chapter 1.1; Dunn and Heneghan, Chapter 2.4). Virtually all core urban habitats are surrounded by a matrix of impervious surfaces and highly modified vegetation. Intensive management commonly results in a predominance of habitats in early successional stages, with these supporting particular arthropod assemblages, including high proportions of phytophages and species with high population growth rates and winged adults (Strauss & Biedermann 2008). With advancing succession, generalists give way to specialists and average niche breadth decreases. However, the majority of urban habitats do not proceed beyond early successional stages and remain predominantly generalist dominated. Urban areas may also contain unique natural habitats that are rare or threatened (Connor *et al*. 2002) in addition to elements of the cultural landscape, such as historical hedgerows, mediaeval lawns, and patches of ancient forest that have been virtually eliminated outside urban regions (see Gilbert 1989). In the following section, we summarize information about anthropogenic urban habitats that have special significance for arthropod biodiversity.

Ruderal habitats (including brownfields, wastelands, and unmanaged grasslands) are open habitat complexes that develop on disturbed sites with nutrient-poor soils. Ruderal vegetation establishes through succession from various starting points, including bare rubble, with accompanying changes in the arthropod fauna. Successional stages of brownfields are characterized by assemblages of leafhoppers with distinct habitat preferences and life histories (Strauss & Biedermann 2008). Carabid assemblages of such highly disturbed urban sites may contain rare and threatened species, which are predominantly thermophilic (Schwerk 2000). These potentially benefit from the higher temperatures of urban habitats due to the heat island effect. Ruderal habitats maintain considerable biodiversity that increases through succession. Furthermore, ruderal habitats host unique assemblages of insects while requiring little management to maintain their ecological value. In fact, too much effort to ‘tidy them up’ can reduce the biodiversity values of ruderal habitats (Angold *et al*. 2006; Hartley *et al*. 2007).

Domestic gardens constitute a large portion of heterogeneous greenspace, with potential for enhancing local arthropod diversity by providing host plants and microsites required by various insects (Tallamy 2009). For example, private gardens comprise 23 per cent of the urban land area of Sheffield, UK; the invertebrate fauna of these gardens varies in abundance and richness with the abundance and canopy cover of trees (Smith *et al*. 2006). From a 15-year study of a Leicester garden, Owen (1991) concluded that gardens can provide suitable habitat for a third of the UK insect fauna. Nonetheless, gardens are connected to the wider landscape, and garden assemblages become less diverse over time when isolated in an urban area (see also Davis 1978). Active management of domestic gardens for enhancement of insect diversity benefits some taxa (Smith *et al*. 2006;

Tallamy 2009), but more information is required about what management activities actually achieve desired results. Gardeners have an incentive for supporting biodiversity, especially populations of pollinating insects (Matteson *et al*. 2008) and predators as they benefit directly from improved fertilization, seed set, and control of pest species (Symondson *et al*. 2003). Opportunities for nature experiences, ecological education, and social interaction flow from community and allotment gardens (Andersson *et al*. 2007; Matteson *et al*. 2008), and these contribute positively to the human sense of well-being (see below).

Parks and other recreational areas comprise highly managed woodland, lawns, and ornamental shrub and flower beds. Urban habitat management leads to unique habitats by simultaneously favouring early successional plants and mature trees (Sörensson 2008). The latter may, in turn, provide habitats for some saproxylic arthropods, although little coarse woody debris and leaf litter exist in these habitats. Mulches applied to flower beds in parks, however, do provide habitat for some invertebrate taxa, and irrigation of urban gardens and parks ensure favourable conditions for others (Cook & Faeth 2006). Although ornamental plantings provide supplementary resources for nectar-feeding insects, these mainly benefit generalist species (Frankie *et al*. 2005). Urban parks may also host a more diverse ant fauna than other urban habitats, and this can be significant in biomes threatened by development (Pacheco & Vasconcelos 2007).

Roadside green comprises areas such as roundabouts and verges that are rough grasslands, maintained by regular mowing. Whilst road verges represent a relatively large surface area in many urban areas, they are narrow, bordered on at least one side by inhospitable habitat, and subject to deposition of heavy metals and other pollutants (Kayhanian *et al*. 2008). The width of the vegetation strip and the abundance of nectar-providing plants in verges are associated with butterfly and moth species richness (Saarinen *et al*. 2005), and these habitats can support uncommon carabids, such as *Amara equestris*, even adjacent to major transport corridors (Koivula *et al*. 2005). Roadside verges also seem to provide movement corridors for some arthropods (Niemelä & Spence 1991), but clearly if corridors do not possess the features required by vulnerable and threatened species, they will only be used by abundant and generalist species, for which corridors are unnecessary. Management of verges can have biodiversity implications. For example, management intensity in grassy verges determined the ability of isolated verges to support both grass-dwelling and arboreal hemipterans (Helden & Leather 2004).

Built surfaces that abound in urban areas may provide habitat for a number of taxa, and may also contribute to the 'heat island effect' in cities, which favours thermophilic species. For example, the introduced spider *Tegenaria agrestis* has become established on cement structures beneath motorway intersections (Gilbert 1989) and the warm, dry, and sheltered conditions makes these habitats particularly suitable for species from warmer climates. The interiors of buildings also provide suitable conditions for a diverse range of arthropod taxa, most of which are regarded as pests and which have, unfortunately, become the most visible domain of urban entomology (e.g. Robinson 1996). This indoor fauna includes a variety of beetles, flies, termites, spiders, ants, and cockroaches, to name a few (Frankie & Ehler 1978). Many of these species feed on dry stored products, such as grain, seeds, fur, and cloth (Gilbert 1989), and have long histories of association with humans. In fact, many such species are known only from urban populations and appear to be synanthropic, and in many instances we know little about their original habitat preferences and food sources.

Refuse heaps are commonly associated with human habitation, especially in and around cities. These provide resources such as carrion, decaying food, and dung to communities of detritivorous taxa, such as carrion and dung beetles and flies (Perez *et al*. 2005). Modern Western cities, whilst being by no means sterile, are relatively hygienic and support species-poor faunas of detrivores (Ulrich *et al*. 2008) but to our knowledge, these faunas have not been studied beyond the above-mentioned carrion study (Perez *et al*. 2005). These arthropods are generally considered undesirable and are therefore excluded by removal of their resources and breeding habitats and via direct control. From an ecological point of view, however, these taxa presumably also include vulnerable and threatened species, so the maintenance of complex detrivore-based ecosystems might be important to ecosystem integrity.

3.3.5 Arthropod adaptation to urban environments

Urban development affects many species negatively (Davis 1978; Connor *et al*. 2002; McKinney 2002), but urban environments can also foster unique interactions among arthropod species and between these species and their environment. Furthermore, as mentioned above, urban environments can select for life history and morphological changes. Frankie and Ehler (1978) argued that insects that survive in urban environments do so either through genetic adjustment and/or preadaptation for life in urban environments. Clearly many arthropod species have adapted to exploit urban habitats (McKinney 2002), and these include species that are characteristically early successional, omnivorous, oligophagous or polyphagous, generalist, and highly dispersive (see above).

Urban arthropod populations may be morphologically or physiologically distinct from their rural counterparts. For example, the carabid beetles *P. madidus* and *Abax parallelepipedus* were larger in urban Birmingham than in nearby rural sites (Sadler *et al*. 2006); however, in contrast, the carabid *Carabus nemoralis* declined in body length towards the city centre of Hamburg (Weller & Ganzhorn 2004). Urban leaf-cutter ants (*Atta sexdens rubropilosa*) are more tolerant to heat than are those from rural populations around São Paulo, Brazil (Angilletta *et al*. 2007). Research about the basis of such morphological and physiological changes will improve understanding of selective forces at work on urban insects.

Industrial melanism in the peppered moth, *Biston betularia*, provides one of the best-known examples of genetically mediated responses of arthropods to urban environments. Over a recent 30-year period (1969–99) the dark *carbonaria* morph decreased from the most common to the rarest in the Netherlands, while the light *typica* morph increased both in observed morph and allele frequencies, presumably the result of reversal in air pollution (Brakefield & Liebert 2000). Interestingly, the darkest of the intermediate *insularia* morphs increased in the 1980s but decreased again in the 1990s, while the mid-*insularia* morph increased during the 1990s. The peppered moth story illustrates how genetics and predator–prey relationships can be altered in a complex, interactive way, mediated by variation in urban features like air pollution affecting the colour of tree trunks on which the species rests.

Byrne and Nichols (1999) concluded that the *molestus* form of the mosquito *Culex pipens* found in the London Underground railway system was genetically distinct from the aboveground populations of this same mosquito species. They argue that a single underground colonization event, followed by adaptation to the underground environment (through strong selection and possible genetic drift), best explains these genetic differences. Interestingly, however, allele frequencies of underground *C. pipiens* populations of Britain and Germany clustered with those from North American, Middle Eastern, Japanese, and Australian populations and not with their British and German aboveground populations. Thus, it is possible that underground mosquitoes from northern Europe are genetically distinct from their aboveground counterparts and more likely derived from a southern mosquito species (Fonseca *et al*. 2004).

The genetic structure of populations of the German cockroach, *Blattella germanica*, was similar in two French cities located 900 km apart, but there was strong genetic substructuring within the cities. Cloarec *et al*. (1999) suggest that this pattern is due to frequent long-distance transport of cockroaches between the cities, whereas within cities a poorly understood combination of restricted gene flow, founder effects, and small effective population size shape the genetic structure of populations. However, long-distance gene flow among five central European cities isolated up to 917 km was insufficient to compensate for the effects of genetic drift between buildings within cities in the cellar spider, *Pholcus phalangioides* (Schäfer *et al*. 2001). The small and demographically unstable mating units in urban populations of this spider suggest that gene flow within a city is limited by the low probability that spiders will enter buildings and not by a low dispersal capacity (Schäfer *et al*. 2001). Thus, specific features of urban areas can alter important evolutionary processes. However, there is much to be learnt about the myriad ways in which these features structure the population genetics of urban arthropod populations.

3.3.6 Arthropod conservation in urban environments

The 'biophilia hypothesis' (Wilson 1984) suggests that human beings are innately connected with nature and that our sense of well-being and behaviour are influenced by the extent to which we connect to biotic elements and functional ecosystems. Thus, it may be argued that reasons to conserve the biota in our surroundings extend beyond the utilitarian aspects of ecosystem services and disease vector management. The maintenance of biodiversity on the anthropogenic 'cultural landscape' garners considerable attention in Western cultures (e.g. Foster 2002; Hunter & Hunter 2008), and urban environments are the developed extreme of cultural landscapes. Thus, present work toward designing cities more effectively to conserve the huge fraction of biodiversity represented by arthropods deserves our attention and will likely have benefits for those that inherit our cityscapes (Connor *et al*. 2002). Those benefits can be counted both in terms of ecosystem services (see McIntyre 2000 for a comprehensive account) and increased connection to nature that is thought to be associated with a sense of human well-being. Therefore, conservation of indigenous arthropods is both an appropriate and practically achievable goal in urban environments.

So, with this goal in mind, how should we proceed? It would be wise to remember that many adult humans do not like 'bugs' (Hunter & Hunter 2008), although few do not find beauty in butterflies. In our experience, these attitudes are mainly learned and so we can collectively 'unlearn' them. In fact, there is evidence that efforts to conserve and enhance populations of interesting insects in urban environments actually improve human empathy for these creatures (Primack *et al*. 2000) and, thus, presumably fosters additional conservation effort. The first and extremely important part of an effective strategy to conserve nature everywhere, including urban areas, is to remember that humans will more highly value nature and the organisms that give it less anthropocentric flavour if given incentives to do so. Progress toward arthropod conservation can be facilitated through education and working conservation projects in urban landscapes. In many places in the world, it will also require progress toward reducing angst about basic human needs so that people may view their environment as more than a base for provision of immediate human needs.

Preserving and enhancing insect populations in urban settings, as in parallel to conservation planning everywhere, will require reserves of natural habitat, effective management of urban greenspace, innovative design of cityscapes developed as living and working areas for humans, and especially, integration of potential arthropod habitats with the urban matrix. Much research, the general themes of which are summarized above, has been conducted with respect to the first issue. Urban greenspace comprises a spectrum of development, ranging from more-or-less disturbed fragments of original cover-type, to areas developed for specific human activities such as road verges, parks, and golf courses. Natural areas are critical for conserving indigenous species and their associated ecological functions (Connor *et al*. 2002), whereas other kinds of greenspace can be designed to support overall biodiversity objectives (see above).

As summarized by Hunter and Hunter (2008), landscape architects have much experience with designing the urban matrix and, by working with them constructively, urban ecologists and entomologists can likely achieve much that will support insect conservation in urban areas. Our growing ecological understanding of urban greenspace can be put to work in design of cityscapes that are effective for their human inhabitants and, at the same time, meet reasonable insect biodiversity goals, mainly by ensuring that potential insect habitats are considered during urban development (Connor *et al*. 2002). Of course, this will go hand in hand with design and sustainable management of urban plant communities. Variety is the key. If urban greenspaces become mainly homogenized plant communities maintained in unnatural configurations through regular chemical and mechanical intervention, insect diversity will invariably diminish over time (Davis 1978; Connor *et al*. 2002).

An emerging challenge for those who design cities is in the 'big picture' integration of urban and suburban landscapes. Conservation planning will be required at the whole city level and must include connections to suburban landscapes (see Connor *et al*. 2002). Of course, urban planners have recognized many problems associated with urban sprawl, but as we seriously incorporate biodiversity concerns into our plans, new kinds of problems directly related to human welfare will emerge. For example, Allan *et al*. (2003) suggest that fragmentation of forests in urban and suburban areas leads to increases in mice populations, through reduction in local populations and foraging pressure from larger avian and mammalian predators. This, in turn, increases the success of nymphal ticks at finding their first meals, and concomitantly the transmission rates of the bacterium responsible for Lyme disease. As a result, human exposure to Lyme disease is increased and fragmentation has indirect effects on human health mediated through complex trophic interactions involving arthropods. This example illustrates that urban design problems related to sustaining biodiversity will require integration of greenspaces and the urban matrix into functional ecological units that foster consequences acceptable to humans. We can do considerably more than we have done in this regard if we shift our planning focus to whole urban landscapes.

Exotic species are generally well-represented in urban assemblages, and urban environments have been a common insertion point for introductions. For example, the innocuous carabid beetle, *P. melanarius*, has become established in many Canadian cities and rapidly colonized large areas (Niemelä & Spence 1991). Of course, species more highly visible as pests (e.g. Gypsy Moth, Asian Longhorned Beetle and the Emerald Ash Borer) have also got their 'feet in the door' through urban centres. The Argentine ant, *Linepithema humile*, is a particularly well-studied example of an unappreciated species that has invaded many cities across the subtropics. This large-bodied and aggressively territorial species competitively excludes all other surface-forging ant species from moist environments (Holway & Suarez 2006), and this may send biodiversity ripples through whole ecosystems via indirect effects. For example, the ants that tend lycaenid butterflies are displaced and as a result the butterflies may be locally at risk (Connor *et al*. 2002). Two things are clear: 1) many exotic species become pests in urban areas, remaining relatively innocuous elsewhere; and 2) exotic species, both known pests and more innocuous species, will increasingly become part of urban biodiversity around the world. Learning to embrace, manage, and minimize the impacts of such species is a goal worthy of attention from urban ecologists.

3.3.7 Future research

Cities can be thought of as a collection of ecological islands with dynamics that are still developing. These result in a number of unique environmental configurations and associated management challenges. Phenomena with special biodiversity implications include 1) rock islands, providing stone-like walls with crevices representing ideal habitat for many species; 2) dry islands, providing suitable climatic conditions for xerophilic taxa, although in arid regions, cities may constitute moist islands; 3) nutrient islands, with levels of soil nutrients increasing towards the city core; 4) food islands, for granivores and species utilizing garbage and dung; 5) heat islands favouring exotic species from warmer climates; 6) disturbance islands supporting early successional species; 7) light islands, which can pose problems for nocturnal species and taxa that rely on light sources for orientation and navigation; and 8) islands of remnant, natural habitats which may or may not be able to support species of the natural environment replaced by the urban landscape. Clearly, urban arthropod faunas comprise a diverse array of species, including specialists and generalists, indigenous and introduced species, and those classified by humans as pest, beneficial, or innocuous species. A major contemporary challenge for urban ecologists seeking to sustain arthropod biodiversity is to go beyond the corroboration of positive or negative impacts of urban development on particular species or assemblages. Instead, we should now turn attention to the effects of urbanization on ecological processes and ecosystem function, and to integrating social and ecological priorities in management strategies that include insect conservation in urban environments (Connor *et al*. 2002; Hunter & Hunter 2008).

Acknowledgements

We thank the Academy of Finland (project number 126915) for financial support.

CHAPTER 3.4

Ecology of Urban Amphibians and Reptiles: Urbanophiles, Urbanophobes, and the Urbanoblivious

Bruce W. Grant, George Middendorf, Michael J. Colgan, Haseeb Ahmad, and Michael B. Vogel

3.4.1 Introduction

'Hello, Simichidas! Where are you dragging your feet to at mid-day?
Even the lizard is snoozing the noontide away in the dry walls,
And the funeral larks are not flitting about in the sunshine.'

—Theocritus (ca. 275 BC), *Idyll* 7.21–24

The admonition above by the goat-herder Lycidas to his city-dweller friend Simichidas for his reckless thermoregulation on a hot afternoon while wiser reptiles and birds seek shelter is not only one of the first recorded observations of vertebrate thermal ecology, but stands as a 2,200 year-old portent of how urban life may be out of step with the natural systems that sustain us. Today, urban ecosystems are the fastest growing ecosystem type on Earth, expanding at rates of over 3 per cent per year at the expense of other ecosystems. More than half of our 6.8 billion plus population live in cities, and this ever-increasing population is both compacting in the urban core while at the same time expanding, via saltational metastasis-like sprawl, to engulf surrounding habitats (Hern 2008). Importantly, the ecological impact of our urban ecosystems, being almost entirely heterotrophic, extends well beyond the urban core and places far reaching demands on agriculture, watersheds, and global capacities of Earth systems to remediate our wastes (Vitousek *et al*. 1997; Grimm *et al*. 2000)—whether these are molecular wastes, such as greenhouse gases, ecotoxins, acidification, or mountains of solid waste (Grant 2003). Worldwide, species of relatively immobile animals, such as reptiles and amphibians (herpetofauna, or herps), are marooned at best and obliterated at worst (Mitchell *et al*. 2008). Herps in today's cities lack the luxury of herps in Theocritus's time which could merely wait out the heat of the day for urbanites to pass by.

The cities of Philadelphia and Washington, DC, which are both major East Coast metropoli, are perfect examples of a syndrome of urbanization and herpetofaunal decline. Table 3.4.1 lists the herps found in Philadelphia from wildlife censuses dating back to collections by John Bartram in 1789 and from results of an extensive, more recent survey undertaken by the Philadelphia Academy of Natural Sciences in 1998–99 (Fairmont Park Commission 1999). Over the past two centuries, Philadelphia amphibians declined from 16 to 10 species and reptiles from 25 to 11.

Surveys of the herpetofauna in Washington, DC, based on historical collections housed in the National Museum of Natural History of the Smithsonian Institution, data from the 1996 and 2007 BioBlitz in Kenilworth and Rock Creek Parks, and photographs of herpetofauna posted on DCNature.com (Table 3.4.2) reveal a remarkably similar decrease in amphibian and reptilian biodiversity. After 1950, amphibian species declined from 20 to 13 and reptiles from 29 to 17. Thus, both Philadelphia (n = 21/41) and

Table 3.4.1 Reptiles and amphibians found within the city limits of Philadelphia from censuses in 1789–1960 and 1998–99 (data compiled in Fairmont Park Commission 1999).

All taxa listed were censused from 1798–1960	found in 1998–99?
FROGS & TOADS	
Eastern American Toad	rare
Anaxyrus americanus	
Fowler's Toad	no
Bufo fowleri	
Gray Treefrog	no
Hyla versicolor	
Northern Spring Peeper	rare
Pseudacris crucifer	
Bullfrog	common
Rana catesbeiana	
Green Frog	common
Rana clamitans	
Pickerel Frog	rare
Rana palustris	
Coastal Plain Leopard Frog	no
Rana sphenocephala	
Wood Frog	no
Rana sylvatica	
SALAMANDERS	
Spotted Salamander	no
Ambystoma maculatum	
Marbled Salamander	no
Ambystoma opacum	
Dusky Salamander	rare
Desmognathus fuscus	
Northern 2-lined Salamander	common
Eurycea bislineata	
Longtail Salamander	rare
Eurycea longicauda	
Redback Salamander	common
Plethodon cinereus	
Northern Red Salamander	rare
Pseudotriton ruber	
LIZARDS	
Five-lined Skink	no
Plestiodon fasciatus	
European Wall Lizard	no
Podarcris murialis (sicula)	
{non-native}	
TURTLES	
Snapping Turtle	common
Chelydra serpentina	
Eastern Painted Turtle	common
Chrysemys picta	
Spotted Turtle	no
Clemmys guttata	
Wood Turtle	rare
Glyptemys insculpta	
Bog Turtle	no
Glyptemys muhlenbergii	
Eastern Mud Turtle	no
Kinosternon subrubrum	
Red-bellied Turtle	common
Pseudemys rubriventris	
Stinkpot	rare
Sternotherus odoratus	
Eastern Box Turtle	common
Terrapene carolina	
Red-Eared Slider	common
Trachemys scripta	
{non-native}	
SNAKES	
Northern Copperhead	no
Agkistrodon contortrix	
Worm Snake	no
Carphophis amoenus	
Northern Black Racer	rare
Coluber constrictor	
Ringneck Snake	no
Diadophis punctatus	
Eastern Ratsnake	no
Elaphe obsoleta	
Eastern Hognose Snake	no
Heterodon platirhinos	
Eastern Milksnake	no
Lampropeltis triangulum	
Northern Water Snake	common
Nerodia sipedon	
Queen Snake	no
Regina septemvittata	
Northern Brown Snake	common
Storeria dekayi	
Northern Redbelly Snake	no
Storeria occipitomaculata	
Eastern Ribbon Snake	no
Thamnophis sauritus	
Eastern Garter Snake	common
Thamnophis sirtalis	

Table 3.4.2 Reptiles and amphibians found within Washington, DC (data from the Natural Museum of Natural History, Smithsonian Institution, 1996 BioBlitz, www.pwrc.usgs.gov/blitz.html and DC Nature, www.dcnature.com)

FROGS & TOADS	**before 1950?**	**after 1950?**
Eastern American Toad *Anaxyrus americanus*	yes	yes
Woodhouse's Toad *Bufo woodhousii*	no	yes
Northern Cricket Frog *Acris crepitans*	yes	yes
Southern cricket frog *Acris gryllus*	yes	no
Cope's Grey Tree Frog *Hyla chrysoscelis*	yes	no
Green Treefrog *Hyla cinerea*	yes	no
Northern Spring Peeper *Pseudacris crucifer*	yes	yes
Upland Chorus Frog *Pseudacris feriarum*	yes	no
Gray Treefrog *Hyla versicolor*	no	yes
Bullfrog *Rana catesbeiana*	yes	yes
Green Frog *Rana clamitans*	yes	yes
Coastal Plain Leopard Frog *Rana sphenocephala*	yes	no
Wood Frog *Rana sylvatica*	yes	yes
Pickerel Frog *Rana palustris*	no	yes
SALAMANDERS		
Spotted Salamander *Ambystoma maculatum*	yes	yes
Dusky Salamander *Desmognathus fuscus*	yes	yes
Northern 2-lined Salamander *Eurycea bislineata*	yes	yes
Longtail Salamander *Eurycea longicauda*	yes	no
Northern Slimy Salamander *Plethodon glutinosus*	yes	no
Northern Red Salamander *Pseudotriton ruber*	yes	no
Redback Salamander *Plethodon cinereus*	yes	yes
Eastern Newt *Notophthalmus viridescens*	yes	no
Greater Siren *Siren lacertina*	yes	no

(*Continued*)

Table 3.4.2 Continued

FROGS & TOADS	before 1950?	after 1950?
LIZARDS		
Eastern Fence Lizard *Sceloporus undulatus*	yes	yes
Little Brown Skink *Scincella lateralis*	yes	no
Five-lined Skink *Plestiodon fasciatus*	yes	yes
Six-lined Racerunner *Aspidoscelis sexlineata*	yes	no
TURTLES		
Snapping Turtle *Chelydra serpentina*	yes	yes
Eastern Painted Turtle *Chrysemys picta*	yes	yes
Spotted Turtle *Clemmys guttata*	yes	no
Wood Turtle *Glyptemys insculpt*a	yes	no
Eastern Chicken Turtle *Deirochelys reticularia*	yes	no
False Map Turtle *Graptemys pseudogeographica*	yes	no
Eastern Box Turtle *Terrapene carolina*	yes	yes
Red-Eared Slider *Trachemys scripta* {non-native}	yes	yes
Eastern Mud Turtle *Kinosternon subrubrum*	yes	yes
Stinkpot Turtle *Sternotherus odoratus*	yes	yes
Red-bellied Turtle *Pseudemys rubriventris*	yes	no
Diamondback Terrapin *Malaclemys terrapin*	yes	no
SNAKES		
Northern Copperhead *Agkistrodon contortrix*	no	yes
Worm Snake *Carphophis amoenus*	no	yes
Northern Black Racer *Coluber constrictor*	no	yes
Ringneck Snake *Diadophis punctatus*	no	yes
Eastern Ratsnake *Elaphe obsoleta*	yes	yes
Corn Snake *Elaphe guttata*	yes	no
Eastern Hognose Snake *Heterodon platirhinos*	yes	no

Common Kingsnake *Lampropeltis getula*	no	yes
Prairie King Snake *Lampropeltis calligaster*	yes	no
Eastern Milksnake *Lampropeltis triangulum*	yes	no
Northern Water Snake *Nerodia sipedon*	yes	yes
Plain-bellied Water Snake *Nerodia erythrogaster*	yes	no
Rough Green Snake *Opheodrys aestivus*	yes	no
Queen Snake *Regina septemvittata*	yes	no
Northern Brown Snake *Storeria dekayi*	yes	yes
Eastern Ribbon Snake *Thamnophis sauritus*	yes	no
Eastern Garter Snake *Thamnophis sirtalis*	yes	yes
Smooth Earth Snake *Virginia valeriae*	yes	no

Washington, DC, (n = 29/49) show declines of about 40–50 per cent in herp taxa once found within the city limits.

Why is this? What are the attributes of some herp taxa that enable them to persist and/or thrive in urban ecosystems while others cannot sustain populations? Are there attributes of the urban ecosystem type that match with the 'native' criteria for some species while not for others? Are there particular urban habitat features that are pre-aligned with the evolutionary adaptations of successful urban taxa? And for the herp taxa found in cities, what roles do these taxa play in urban ecosystem food webs, energy flow, and biogeochemical cycles? What roles do they play in stabilizing ecosystem function amidst the disturbance pervasive in urban environs? Can these taxa serve as indicators of environmental stress as well as indicators of successful urban ecosystem re-engineering for sustainability? How effectively can we use these taxa to educate our urban populations about the value of biodiversity and engender curiosity, care, respect, and responsibility for our hybrid natural–built world?

Our goal with this chapter is to lay out an ecological science–policy–educational framework (Lubchenco *et al*. 1991) to 1) better understand herpetofaunal adaptation to the urban environment, 2) suggest a research and educational agenda for using urban herpetology to better understand urban ecosystem structure, and 3) to inform decision-making about our urban ecosystems (Zipperer *et al*. 2000; Berkowitz *et al*. 2003). By understanding how herpetofauna adapt, accommodate, tolerate, and function in urban ecosystems we may be better able to evaluate herpetofaunal conservation efforts and to evaluate urban ecosystem design for the sustainability of all urban dwellers. Because herps are closely tied to biophysical parameters, urban herps are model organisms to study the effects of global environmental change on our cities and thus, urban herps provide both metaphor and medium for urban sustainability science and education.

3.4.2 Herps *in* cities

Anthropogenic influences in urban ecosystems result in unique biophysical and environmental stressors, constraints, and opportunities on herpetofauna at both individual and population levels. The transformation of indigenous herpetofaunal communities can be broadly categorized as the

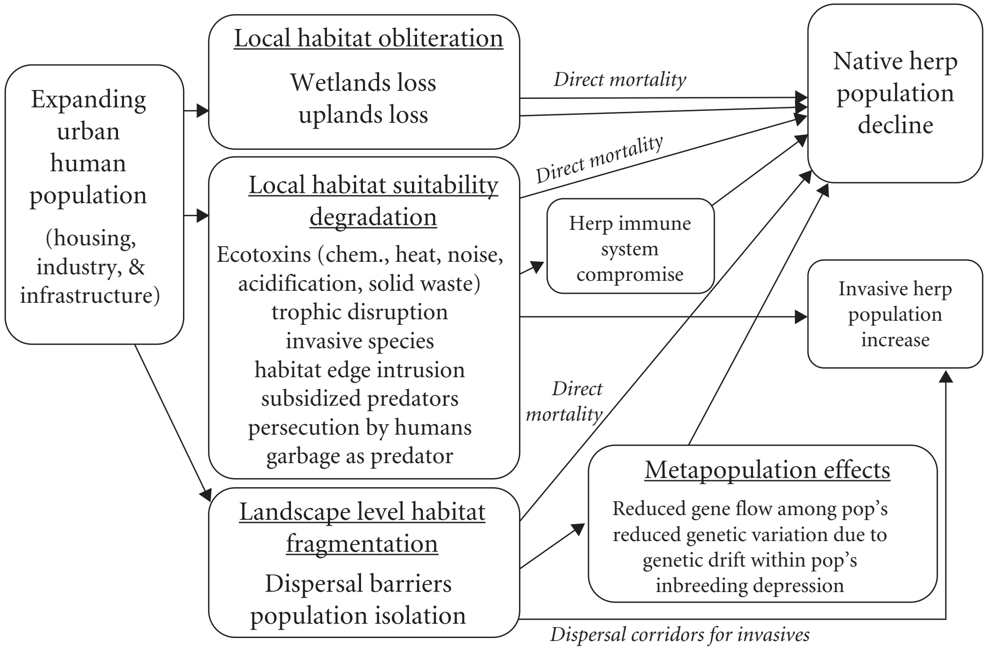

Figure 3.4.1 The transformation of indigenous to urban herpetofaunal communities as the result of local habitat obliteration, local habitat suitability degradation, and landscape-level habitat fragmentation

result of: (1) local habitat obliteration, (2) local habitat suitability degradation, and (3) landscape level habitat fragmentation (see Fig. 3.4.1).

3.4.2.1 Local habitat obliteration

The single most significant loss of herpetofauna in urban areas is due to habitat destruction (Mitchell *et al*. 2008). The conversion of naturally occurring forests or wetlands into housing, businesses, transpor infrastructure, and other industries of our urban matrix can result in direct herpetofaunal death by development—most herps cannot escape bulldozers (Tilman *et al*. 1994; Gibbs 1998; Wilcove *et al*. 1998). In addition, many herps (and most amphibians) require dual habitats. Loss of wetlands, riparian, or upland terrestrial breeding or living habitats can lead to population loss (Gibbons 2003; Herrmann *et al*. 2005; Skidds *et al*. 2007). Loss of critical breeding habitat (generally, wetlands for amphibians and terrestrial nesting sites for reptiles) (Semlitsch & Bodie 2003; Rubbo & Kiesecker 2005) will eventually affect even the longest lived herps. Historically, this is a not a new form of encroachment. For millennia, cities have been located on major rivers and/or continental margins with wetland-rich habitat and biotic diversity. However, over the past century, large scale changes in wetlands by draining, backfilling, development of waterway margins, stormwater mitigation by channelization, and impoundment construction have dramatically reduced the wetland habitat needed by and suitable for herpetofauna in urban areas.

3.4.2.2 Local habitat suitability degradation

In contrast to outright habitat devastation, urbanization can also affect the integrity of resident natural populations by more subtly altering the habitat. This may occur via ecotoxicological impacts, changes in the availability of preferred food, and alterations in species with which the herps interact. Ecotoxicological impacts include airborne toxins, acidified soils and wetlands, pesticide, herbicide, de-icing and hydrocarbon contamination of soil and groundwater, and direct industrial dumping (Sparling *et al*. 2000; Karraker 2008; Snodgrass *et al*. 2008). In addition, artificial night lighting can disrupt nesting and nocturnal activity (Perry *et al*. 2008), noise can affect anuran breeding choruses, and urban heat plumes can alter active and breeding season phenologies and bioenergetics, for example by causing greater over-winter lipid depletion and less storage for survival and breeding, a process comparable to that outlined by Sinervo *et al*. (2010) for global warming.

Several studies suggest that trophic top-down and bottom-up effects are likely to be important. Increases in urban predators (especially cats, dogs, and people), with some predators being unintentionally subsidized by people (e.g. raccoons; Mitchell & Klemens 2000; Ner & Burke 2008), can wipe out undefended taxa such as lizards, nesting turtles and their nests, and so on. To some degree these effects may be offset by the loss of other, naturally occurring predators of herps (especially raptors and large-bodied snakes), which are sensitive to urban stresses. Of course, we must also mention deliberate human persecution of larger bodied predatory reptiles as well. Lastly, and although not technically a trophic effect, plastic garbage, especially netting, 6-pack can rings, bags, and so on, can act as an important ‘urban predator’, most significantly affecting aquatic turtles and snakes. Of course, a similar argument holds for vehicle traffic, which we will address separately below.

Although a great deal is known about the urban-induced changes in community composition (McKinney 2002), the effects of the changes and those of invasive species on ecosystem structure are lesser known. The progression of urban-induced change is likely to impact herps through enforced algivorous and invertivorous dietary shifts. Many terrestrial herps, particularly frogs and salamanders, are major predators of soil and understorey invertebrates and many of these herps also depend upon algae as their primary food source during early stages of development. Anthropogenic disruption of habitat integrity and water quality facilitates species invasions, which in turn alter the ecosystem in largely unknown ways. It would seem likely that as invasive plants and insects change the composition and structure of urban forest remnants, cascading trophic effects on herbivore and decomposer communities would be sure to follow. One would expect dramatic changes in food availability and diet for both algivorous and herbivorous herps. However, descriptive data are lacking to compare diets of herp species along urban–rural gradients to assess these basic trophic ecological effects. The impact of invasive competitors and predators on herpetofauna is little understood and remains to be investigated.

In addition to these direct effects, there are numerous other indirect, or at least somewhat less direct, possibilities. For instance, impacts resulting from stress, hormones, and compromised immune systems induced by changes in habitat, diet, and species interactions are known, for example ecotoxin effects, as Hayes *et al*. (2002, 2006) have shown for the herbicide atrazine. Further, these are likely to be exacerbated by trophic effects where habitat disturbance-induced stress further increases susceptibility to disease and parasite-induced developmental anomalies. This is especially likely in urban recreational areas where chemicals toxic to herps are routinely used, such as parks and golf courses (but see Song & Knapp 2003; Scott *et al*. 2008). Recent studies of salt and other chemicals used in de-icing also suggest roadside and downstream effects (Forman & Alexander 1998; Sanzo & Hecnar 2006; Karraker 2008).

3.4.2.3 Landscape level habitat fragmentation

The effects of habitat loss, particularly those mediated by habitat fragmentation, are well-documented for a wide variety of ecosystems (Rosenzweig 1995). Most profound are the impacts of reduced population size and constraints on dispersal (Mitchell *et al*. 2008). Small population sizes lead to genetic drift, allele loss, inbreeding depression, and within-patch extinction (Berven & Grudzien 1990; Parris 2006). Dispersal constraints include not only that movement through unsuitable habitat as forests and wetlands change with development, but the extraordinary barriers that roads, rails, and the like pose. The lethality of such barriers largely results from a combination of the slow speeds of herps and the fact that the habitat is completely open and absent of any protective cover (Funk *et al*. 2005). Road traffic is particularly lethal for many taxa. Hopping or crawling speeds are simply too slow to allow safe passage for even roads with low traffic flow. Major highways are major kill zones (Andrews *et al*. 2008). Railways, too, can be impenetrable to nesting adult turtles, and railbeds often lack gaps at ground level to allow juvenile dispersal. Fencing of railways, roads, and industrial and housing developments can further impede dispersal. Increased predation pressure in the resulting fragmented areas creates small population sizes within patches, and when combined with dispersal barriers to gene flow (Andrews *et al*. 2008) prevents metapopulation rescue effects of both numbers and genetic variation. (Rowe *et al*. 2000).

3.4.2.4 Ecological mechanisms

Although the effects and impacts of urban change are widely known (Mitchell *et al*. 2008), almost completely missing in the literature are the nuts and bolts mechanistic science of *why* particular taxa do or do not persist as urban species. What exactly are the selection pressures and ecological contexts that produce the evolutionary physiological, behavioural, and population genetic responses that enable some herps to adapt to urban ecosystems, whereas others are excluded?

Examination of herpetofaunal responses to urban areas might profitably be preceded with a look at least one other group—plants—for which much is known. So-called weed or ruderal plants, those that occupy disturbed and waste sites (Garber 1987; Vessel & Wong 1987), typify 'wild' urban flora (where 'wild' refers to native, naturally occurring species). Disturbed areas are most naturally colonized by 'r-selected' species which tend to be short-lived, small, and good dispersers—in other words, weedy species. Although much of the urban environment is best characterized as disturbed, there are often areas that remain relatively undisturbed, for instance parks and forested areas. Plants in these areas tend to be longer-lived (often pre-dating the establishment of the parkland in which they remain), large, and often poor dispersers. The ecology of the effects of frequency and severity of habitat disturbance on community structure has been studied for decades (Pickett & White 1985; McCabe & Gotelli 2000) and provides an interesting and useful framework for understanding herpetofaunal associations with urban environments.

Understanding urban herp distributions requires knowledge of how island biogeographic and disturbance theory can be used to predict the abundance, frequency, and longevity of urban herp populations. Biogeographic analysis (MacArthur & Wilson 1967) should incorporate the perspective of each particular species with respect to habitat grain, dispersal ability, and island size. To this we would add species fecundity and the degree of urban isolation imposed. Most of these factors have been well-studied (see the recent review in Laurance 2008). However, one parameter, the degree of isolation, considers the severity of barriers which will vary among species and will require examination and testing. For instance, we expect that patches surrounded by highways will be effectively more isolated than those surrounded by housing developments. Disturbance impacts on urban herps have also not yet received much attention. We suggest the need to examine both the impact of frequency and severity of disturbance and the sensitivity of species to types of disturbance (habitat disruption, fire, toxins, etc.). We suspect that many herps may well be as influenced by the availability of vacant lots as the size of natural and semi-natural parks (Vessel & Wong 1987). Thus, urban herp communities would be influenced by the biogeography of vacant lots (number and distribution) and also be affected by disturbance (the longevity of lots as driven by urban

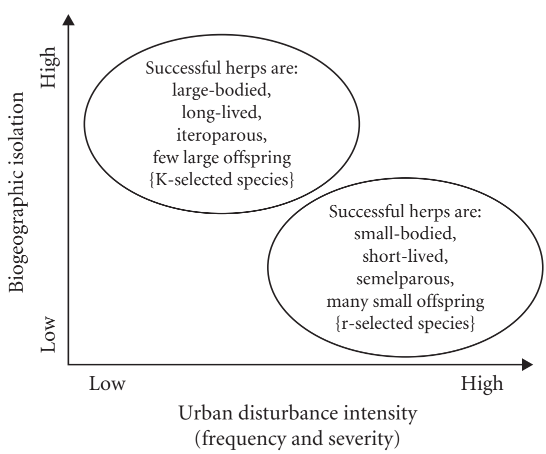

Figure 3.4.2 A generalised role of biogeographic and disturbance factors on urban herps as mediated by life history influences.

renewal, revitalization, and smart growth where infill replaces sprawl).

We might thus predict that herps species displaying life history characteristics associated with 'r' strategies (small-bodied, short-lived, semelparous, many offspring) would typically survive in small, highly disturbed sites nearby to other similar sites, such as vacant lots (Fig. 3.4.2). In contrast, large-bodied, longer-lived, iteroparous species ought to be associated with large undisturbed urban areas, such as parks. But these features alone are probably insufficient to delineate all possibilities.

The influences of a number of other issues affecting species remain to be examined, such as habitat generality, minimal habitat requirements, sensitivity to change, and so on. Also in need of investigation are how these same factors might effect herp communities—the interactions and connections among herp species in urban areas. Given the astonishing fact that salamander biomass in north-eastern US forests is equal to that of birds (Burton & Likens 1975), we might further note that the ecological roles of herps in urban ecosystems have the potential to be greater than expected. Thus, the effects of these same communities in urban areas are largely unknown, as are the potential impacts of the loss of species on the structure and dynamics of the urban ecosystem (Ellison *et al*. 2005).

3.4.3 Herps *of* cities

In this section we ask, is there a unique herpetofauna of the urban ecosystem type, and if so what are their characteristics? What are the rules of the structure, urban trophic relationships, community assembly, and evolutionary synecology of urban herpetofaunal communities?

Mitchell *et al*. 2008 (and references therein, especially Rodda & Tyrell 2008) characterize two distinct types of urban herpetofauna: urbanophiles and urbanophobes. We summarize these in Table 3.4.3 and add a third category—urbanoblivious.

Urbanophiles—basically, urbanophiles are either uniquely adapted to specific urban habitats or act as

Table 3.4.3 Summary characteristics of urbanophiles, urbanophobes (after Mitchell *et al*. 2008), and of the urbanoblivious.

Urbanophiles	**Urbanophobes**
♦ broad habitat tolerances, which could include managed lawns and parks, buildings, storm drains, impoundments	♦ habitat specialists, with low tolerance to built habitats, wetlands disturbance, habitat fragmentation
♦ dietary generalists	♦ dietary specialists
♦ high mobility and dispersal ability despite urban habitat fragmentation and barriers	♦ lower mobility and poor dispersal ability—sensitive to urban dispersal barriers and habitat fragmentation
♦ high reproductive output	♦ lower reproductive output, especially in fragmented subpopulations
♦ small body size	♦ intolerance by humans
♦ tolerance of humans	

Urbanoblivious

- ♦ dietary generalists
- ♦ cryptic habits
- ♦ capable of replacement reproduction in relative isolation
- ♦ high tolerance to a diversity of urban insults (biophysical, ecotoxicological, and solid wastes, and hydrological, trophic, and landscape-level disturbances)
- ♦ largely unnoticed by humans

♦ Temporally Urbanoblivious	♦ Spatially Urbanoblivious
◊ habitat generalists but restricted to obscure urban refugia of diverse characteristics but frequently riparian and post-industrial	◊ habitat specialists, but their habitat was captured intact by saltatory urban expansion and then overlooked by, or set aside from, further development
◊ higher mobility and dispersal ability, but tend not to disperse as adults	◊ low mobility and dispersal ability
◊ larger body size	◊ small body size
◊ long-lived with huge demographic lag in population growth rate	◊ small home range sizes

mobile, habitat generalists. Their ability to use and show preference for certain urban habitat features, coupled with their dispersal abilities, may enable them to move freely among urban subhabitats as expansion and redevelopment proceeds. Much of the thinking about urbanophiles stems from the conventional wisdom of the island biogeography model of MacArthur and Wilson (1967), applied to a disturbed and fragmented urban landscape colonized by mobile, generalist taxa. Urbanophiles fit this bill since they are defined to be good dispersers for which specific 'built-in' attributes of the urban environment match some particular habitat requirement they have in their native habitat—House Sparrows and Rock Doves are classic avian examples, whereas wall lizards, anoles, and geckos are good herpetofaunal examples. In this view, urban patch diversity results from the superposition of ecological processes enabling patch colonization and driving local extinction.

Urbanophobes—in contrast, urbanophobes are sensitive to the myriad of stresses and disturbances urban ecosystems impose and are summarily driven to exclusion—by definition (McKinney 2008). They may be found in cities currently, but are most likely not to persist. One might think of them as simply occurring in cities either because their populations pre-dated urban expansion into their habitat and so far they have managed to persist, or they dispersed or were transported into urban areas from elsewhere.

Urbanoblivious—upon examination of the identities of the taxa who have persisted in Philadelphia and Washington, DC (e.g. Tables 3.4.1 and 3.4.2, and elsewhere cited above), we offer a third category of urban herps. These are herps that exhibit a mixture of urbanophilic and urbanophobic life history characteristics, but the reasons for their persistence are quite different than those of urbanophiles. We call these taxa 'urbanoblivious' because they live obscure and cyptic lives within relatively isolated refugia that were surrounded by urban saltational expansion. These taxa may be quite specialized in habitat types but are generalists in terms of diet and, more importantly, they are generalists in terms of being able to physiologically tolerate a wide variety of urban insults (biophysical, ecotoxicological, and solid wastes, as well as hydrological, trophic, and landscape level disturbances) (Moll 1980; Gasith & Sidis 1984). Of particular note would be their capacity for immunological resistance to toxic compromise. In sum, they do not really notice that they are in an urban environment, are often hidden from sight, and persist tenaciously in the 'slow lane'. Among the urbanoblivious, two distinct groups emerge. Firstly, 'temporally urbanoblivious' taxa are long-lived, large bodied, cryptic, and typically restricted to riparian refugia that range from the margins of post-industrial brownfields to urban impoundments and other artificial managed or abandoned wetlands. For these species, the key facet of persistence is their long-term pattern of local recruitment. Good examples of this type are Snapping Turtles, *Chelydra serpentina*.

The second group is the 'spatially urbanoblivious' for which taxa are relatively small, short-lived, have low mobility, low dispersal ability, and are much more specialized in their habitat requirements. What enables them to persist is their short list of habitat requirements (feeding, reproduction, over-wintering) that can be entirely met within a small woodlot, vernal wetland, or other such isolated patch on the order of a few hectares in size. To these taxa, the isolated urban parks where they live are not really disturbed regimes—urban encroachment has bypassed their small spatial scale of existence. These taxa differ from urbanophiles in that the spatially urbanoblivious are not adapted to live in any specific attributes of the built world and merely persist because they live at such a small scale while urban expansion has leapt over their locally relatively undisturbed patch. Good examples of this type are Redback Salamanders, *Plethodon cinereus*, which exhibit dense populations in small isolated parks throughout Philadelphia. For both groups of urbanoblivious taxa, a distinct suite of key properties of their life histories and demographies facilitate their long term local persistence—albeit a precarious one. The next bulldozer, industrial spill, or disease, could permanently wipe them out in any given patch thus reducing the number of viable subpopulations and inching their metapopulation closer to extinction across the urban landscape.

3.4.4 Urban herps as indicators

Given the association of herps with certain urban areas, for example see the discussion of life history traits associated with the occupation of vacant lots and parklands, along with an understanding of how species react environmentally, such as sensitivity to toxins and thermal change, we postulate that herps can serve as very effective indicators of environmental change. To a degree, this has already been achieved. Focusing mostly on biogeographic variables, that is fragment size and shape, woodlot size and shape within the fragment, but also considering the presence of water, distance from the city centre, and legal protection, Vignoli *et al*. (2009) examined herp distributions in Rome. They reported that species distributions were significantly affected by a number of these parameters (fragment size and shape, woodlot size, water) in predictable fashions. Thus, these results could then be used to monitor areas by noting the presence and absence of expected species.

In fact, given the sensitivity of amphibians to chemicals and toxins and of reptiles to thermal conditions, using herps as indicator species should not only provide general information concerning environment change, but also specific information concerning the type and degree of change—and if not specific information per se, should serve to direct attention to particular parameters for further investigation.

3.4.5 Urban herps as educators

The use of amphibians and reptiles in education is longstanding and commonplace (e.g. 'Reptiles and amphibians in the classroom zoo' in Netting 1948). Undoubtedly, the choice to feature a frog on the cover of Richard Louv's *Last Child in the Woods* (2008) is because most children find herps fascinating. Opportunities for herps in education abound—for instance, a suggested treatment for nature-deficit disorder, not an illness but a description of the human costs of alienation from nature, uses frogs as an opportunity to reconnect children to the natural world, and other herp connections include using critter-focused hikes (including herps) to address childhood obesity (Louv 2008).

An Internet search of 'amphibians and reptiles in the classroom' yielded over 44,000 returns—ranging from incorporating herps into curriculum and lesson plans, websites with general information aimed at children, and websites dealing with specific herp issues such as understanding frog deformity, making classroom posters, and tips on keeping herps. In the US, millions of elementary school children are familiar with herps kept as classroom pets. Thus, many children have some classroom herp experience, however, this practice is far more common in more affluent schools than in poorer urban districts, which raises important issues in environmental justice in ecology education (Middendorf & Grant 2003).

Although fascinating by all accounts, this classroom experience is limited and quite different from encountering a herp in the 'wild'—even if the 'wild' is the vacant lot or woodlot adjacent to a school. Absent the protective enclosure, herps may take on a far more sinister and dangerous aura, and wild exposure is helped by gentile and expert guidance by trained environmental educators. Most urban areas, and Philadelphia is no exception, have remarkable facilities in city parks, preserves, or refuges, dedicated to outdoor environmental education. The important goal of urban herpetofaunal conservation requires the essential step of better connecting regional centres of informal environmental education to school-based field projects in herpetological education, so as to get beyond the 'one field trip per year per grade' modality. In addition, through recent advances in instructional technology, millions of children could supplement their actual field time with virtual field trips using real time weblinks and streaming data from remote sensing equipment. Urban school children could log on to a field station website nearby and participate by video uplink with volunteers searching for herps beneath an array of coverboards (plywood or some other durable object, placed in areas where herps live, see Grant *et al*. 1992). Transects and other quantitative survey techniques provide teachers with opportunities to include numerical and statistical skills development using real herp populations as part of standards-based science education. Our point is that we need to better connect field sites to classrooms to make the natural world more meaningful to young children. This is needed now more than ever because present day social forces are slashing school budgets, axing field trip opportunities, and restricting our children's exposure and learning about our natural world to indoor classroom curricular confines. These actions serve to undermine exactly the kinds of experiences that would promote responsible urban wildlife management and conservation and the development of ecological thinking (Berkowitz *et al*. 2003).

In addition to promoting ecological literacy, issues in herp conservation provide excellent examples in teaching students about the complexities of natural resources management and urban design. For example, Fig. 3.4.3 shows a detail of a model-building

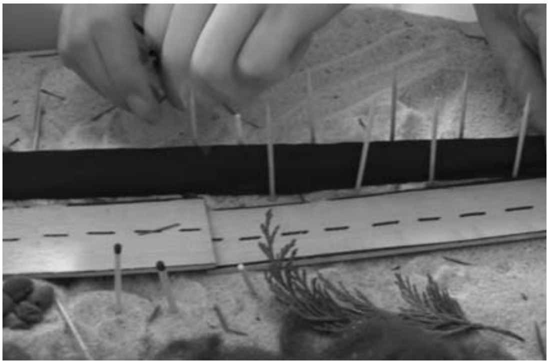

Figure 3.4.3 Detail of a model-building project for 7th-graders in which students design roadside barriers to prevent herps from being killed by vehicular traffic and yet allowing herps access to habitat on both sides of the road. These students came up with the design (with teacher guidance) to place drift fencing, roadside culverts, and a herp tunnel under the roadbed to facilitate orthogonal herp and human traffic coexistence. Photo by Bruce W. Grant

project for 7th graders in which students are given a box of sand, toothpicks, and other items, and tasked with designing roadside barriers to prevent herps from being squashed and yet giving them access to habitat on both sides of the road. Note the drift fencing, roadside culverts, and herp tunnel under the roadbed to facilitate orthogonal herp and human traffic coexistence. These kinds of 'sustainability science' projects provide far more rich curricula than simply reciting that we have to save the frogs. Students learn the importance of studying and understanding the natural history and ecology of living organisms, and that we can use our technology to intentionally design our built world to express our care about coexisting with the natural world around us.

3.4.6 Concluding comments

While the debate continues regarding the causes of herpetofaunal decline in relatively pristine natural areas (i.e. local effects of global perturbations in natural support systems that have occurred in recent decades due to anthropogenic causes), there is little doubt about the causes of the majority of herp declines in the urban areas where the majority of the human population currently resides. Simply put and as a matter of fact, urbanization generally kills herps (Mitchell *et al*. 2008). But having established that allegation, it is essential to follow with the admission that we are fundamentally an urban species and, over the past 6 plus millennia, our historical cultural identity and the majority of our gifts in innovation in the arts, sciences, and technologies are the fruits of our urban communities, broadly defined. Possibly, one can argue that our urban environmental development and its associated locally ecocidal activities over the many centuries of our development were performed in ignorance—an 'ends justifies the means' argument.

But such is not the case today. Evidence from urban herpetofaunal communities clearly shows the challenges that a haphazardly built world poses to life on Earth. We now know a great deal about the consequences of biodiversity loss in terms of reduced capacity of the natural world to perform critical ecosystem regulatory or remediation services for us, or in terms of reduced availability of raw biotic materials for new pharmaceuticals, foods, or fuels for us. In sum, we now know that urban herps are sentinels of declines in the ecosystems upon which we depend—herp declines foretell challenges to our sustainability. As we grow our urban ecosystems, we must also redesign their internal functioning to attain ecological sustainability. Cities are our ecosystems—it is up to us to make them livable or lethal. Herps call to us and thereby relay indicate which of the two we are doing. We have to listen. We need to redesign our house to make them welcome, or we will not be welcome ourselves. We need to design our world less like Theocritus's goatherder, Lycidas, exemplified with his waltz in the noonday sun, and more like the lizards patiently in tune with a measured economy of nature.

Acknowledgements

For their aid in gathering data used in this paper, we would like to thank BioBlitz, National Park Service, Virginia Herpetological Society, and the Division of Amphibians and Reptiles at the National Museum of Natural History of the Smithsonian Institution—and especially John Kleopfer, Ken Ferebee, John White, and Ken Tighe. GAM is a Research Associate in Division of Amphibians and Reptiles at the National Museum of Natural History of the Smithsonian Institution. We thank the monograph editors Jari Niemelä and Nancy McIntyre for their encouragement, patience, and important suggestions in improving this chapter.

CHAPTER 3.5

Biodiversity and Community Composition in Urban Ecosystems: Coupled Human, Spatial, and Metacommunity Processes

Christopher M. Swan, Steward T. A. Pickett, Katalin Szlavecz, Paige Warren, and K. Tara Willey

3.5.1 Introduction

Human activities globally have caused widespread disturbance resulting in habitat loss and change, movement and displacement of species, and the assembly of novel ecological communities (Berry 1990; Turner *et al*. 1990; McIntyre *et al*. 2000). Community ecologists seek to understand the underlying causes of patterns of species richness and composition, and there exists a rich literature for non-urban habitats. Urban ecosystems present community ecology with new opportunities to expand on, or perhaps develop new, ecological theory to explain patterns in species diversity. Some basic theory is being applied in the urban context, including the relationships among species richness and productivity, disturbance, and stress, whereby humans are viewed as imposing some gradient of change that basic theory tells us should result in changes in species richness (McKinney 2002; Faeth *et al*. 2005; Shochat *et al*. 2006). While human behaviour as a general category adds much explanatory power to urban models, in reality, human behaviour and decision-making consist of a complex set of socio-economic processes that influence not only the number of species in urban ecosystems, but how they disperse, and under what conditions they can coexist. That is, the relative role of local versus regional processes in regulating species assembly and community structure, itself with a rich history of ecological inquiry, is subject to interaction with humans, creating an opportunity for the development of a strong socio-ecological theory in community ecology.

Species diversity, which includes richness and evenness, is expected to change with human influence. Often, ecological analyses rely on simple variables such as impervious land cover and human population size (Arnold & Gibbons 1996). The interaction of humans with species diversity may in fact be more complex than the simple relationships imply. Three theories posed to explained patterns in diversity—the productivity hypothesis (Gaston 2005), the ecosystem-stress hypothesis (Menge & Sutherland 1987), and the intermediate disturbance hypothesis (Connell 1978)—all of which derive from basic ecological theory, may be useful in recognizing appropriate mechanistic complexity in the relationship of human behaviour and species diversity. Although patterns vary across scales, there is support for each of these theories, which have been addressed elsewhere (e.g. Lepczyk 2008). However, shifts in patterns of species composition are obvious in urban ecosystems and include changes in top carnivores (e.g. cats and coyotes; Crooks & Soulé 1999; Lepczyk in press), the introduction of exotic species (Clemants & Moore 2003), pets and other aesthetic species (Baker *et al*. 2008), eruptive species (e.g. white-tailed deer; Storm *et al*. 2007), invasive species (Rapoport

1993), and pests (Wetterer 1999; Morgan *et al*. 2009). In addition, there is mounting evidence that created habitats in the urban matrix are harbouring thriving populations of native species (e.g. amphibians in stormwater retention ponds, Simon *et al*. 2009; riparian reforestation, Breuste 2004). These species span taxonomic orders and exhibit substantial variety in life-history patterns (Blair 2001; Yli-Pelkonen & Niemelä 2005). In urban ecosystems, though, standard modes of dispersal and patterns in coexistence can be superseded by human facilitation of species movement and the purposeful placement (or elimination) of species on the landscape. This factor in species assembly is perhaps in contrast to the view of communities that become established as a consequence of human alteration of the landscape (e.g. nutrient enrichment; Shochat *et al*. 2006). These two perspectives certainly play a role in what local versus regional processes shape local community composition.

Metacommunity theory challenges the view that communities are localized and isolated, by placing them on a landscape and connecting them via a common regional species pool (Leibold *et al*. 2004). It organizes hypotheses on patterns in alpha and beta diversity and in species composition that arises and/or vary across scales. Conceptually, metacommunity theory distinguishes between local and regional effects. Local effects comprise environmental constraints and species interactions, and regional effects are driven by dispersal and factors influencing dispersal (Cottenie *et al*. 2003; Cottenie & De Meester 2004; Urban 2004; Chase *et al*. 2005; Werner *et al*. 2007). Metacommunity theory distinguishes the relative role of local vs. regional factors as structuring processes into four categories of metacommunity structure (Leibold *et al*. 2004). While not necessarily mutually exclusive, each hypothesizes mechanisms responsible for species coexistence.

1) Species sorting: spatial niche separation and resource gradients and/or different patch types are strong enough to influence local demography and species interactions. These factors shape local community composition more strongly than do spatial dynamics.
2) Neutral: local community composition is balanced by extinction, speciation, and dispersal. All species are similar in terms of competitive ability, dispersal, and fitness (Hubbell 2005).
3) Mass effects: resource gradients, differential patch types, and species interactions are important in shaping local community composition, but dispersal plays a much stronger role. Local species can be rescued from competitive exclusion through immigration from locations where they are competitively superior.
4) Patch dynamics: patches may not be identical and can harbour any species, but local composition is controlled mainly by local extinction and colonization. See Wu and Loucks (1995), Pickett *et al*. (2000), and McGrath *et al*. (2007) for broader definitions not necessarily constrained by metacommunity theory.

Integral to the metacommunity concept are the explicit roles of space and dispersal relative to local environmental conditions and interspecific interactions. These factors are influenced by human activity in urban ecosystems. As such, the metacommunity concept provides a useful framework for integrating social science theory to account for patterns of diversity in urban ecosystems.

3.5.2 Constraints on metacommunity properties in urban systems

At first glance, the most prominent features of urban ecosystems are a substantial disaggregation of habitat and an increase in spatial heterogeneity. Over very small scales, patterns in natural and remnant vegetation and human infrastructure can change, resulting in complex spatial patterns not present in pristine environments (Rebele 1994; Cadenasso *et al*. 2007). These new patterns require new tools for their measurement in addition to presenting species with vastly different perceptions of the environment (Cadenasso *et al*. 2007). In general, alterations in habitat area and edge features are known to influence both life-history patterns and interspecific interactions, and certainly emerge as important spatial factors in urban community ecology (McKinney 2002). Human activity creates more patches of smaller sizes and greater edge length, benefiting those species that thrive at edges (e.g. white tailed deer, Alverson *et al*. 1988). However, reduced patch size is detrimental for organisms that require larger interior habitats (Donnelly & Marzuff 2004).

As human activities create a more fragmented natural environment, the linkages among natural areas become more important for maintaining populations and their genetic diversity. Furthermore, humans may intervene in the dispersal of plants and animals. Urban development results in patches of habitat with limited connectivity, critical for dispersal. Patches may be connected by parks, greenspaces along roads, median strips, or riparian zones (Fernandez-Juricic 2000). Those patches that have less connectivity or greater distances between patches may suffer from lower species diversity and be dominated by organisms that have greater dispersal capabilities (Bierwagen 2007). Humans might further modify dispersal patterns by selecting species by placing preferred taxa in yards, parks, and gardens (see below). For example, humans may transplant a wild shrub from a forest to a suburban yard, or humans may trap nuisance animals in urban areas, such as beaver, and relocate them to a forest farther from urban development (DeStefano & DeGraaf 2003). The metacommunity concept is based on spatial distribution of communities and their linkages (Leibold *et al*. 2004). Humans make it possible for an organism to be transported farther than it ever could move on its own and also actively select some species over others to support in urban environments.

3.5.3 Social dimensions of biodiversity in urban ecosystems

Biodiversity in urban ecosystems depends not only on the familiar biotic and physical factors that are used to explain species distribution in non-urban systems, but also on social patterns and processes (Low *et al*. 2000). Perhaps the primary social driver of biodiversity pattern is control over land (Alig *et al*. 2004). This is summarized in the concept of property regime (Ostrom 1990; Grove *et al*. 2005). A particular parcel of land is either free for all, called open access, a community-controlled commons, a private person or institution, or the state. It is not simply ownership that indicates who has control, as owners may sometimes exercise little control over a particular plot of land, as in the case of abandoned lots. One consequence of this variation in property regimes is that human population density clearly is not the sole driver of biodiversity in a city or neighbourhood. Instead, the organization of people into institutions, such as households, neighbourhood associations, or interest groups, determines how control over land is exercised and how decisions affecting biodiversity are made (Berkes & Folke 1994; Burch *et al*. 1997; Liu *et al*. 2005).

A second social factor is wealth or income (Logan & Molotch 1987). This factor determines who—again recognizing that the 'who' can be individuals, households, or other kinds of social institutions—has access to resources that enable them to change habitats, import species, or manage land in various ways (see also Cilliers & Siebert, Chapter 3.2). Varying affluence may be associated, for example, with varying use of landscaping services, fertilizer, nursery plants, or bird seed (Zhou *et al*. 2008). In other cases, it is not the wealth per se of the residents of an area, but their influence on the disposition of government resources that is the lever affecting biodiversity (Troy *et al*. 2007). The 'life cycle' of households may influence whether that wealth is actually spent on actions that affect biodiversity. Households with children or supporting college-age students may spend their wealth in different ways than 'empty nesters', with different impacts on biodiversity (Grove *et al*. 2006).

A variety of other social factors—education, culture, and lifestyle—determine the vision or expectation that people or institutions have about the landscapes they control (*Grove et al*. 2006). People with different levels of education, both formal and informal, may have knowledge about the environment and the role of biodiversity in it. Formal education may make people aware of different options for management, or suggest access to power that may not be apparent to those with less formal education. Culture, sometimes associated with ethnicity, also influences the species that people value in both public and private spaces (Fraser & Kenney 2000; Cilliers & Siebert, Chapter 3.2). For example, in Baltimore (a city in Maryland in the United States), the cultural habit of swept yards that some African-Americans imported from the rural South (Westmacott 2002), or the cement-covered, green-painted yards common to some Eastern European groups, would be expected to have very different ecological impacts to the rustic garden ideal expressed by some other American cultures (Schroeder 1993; Bhatti & Church 2004). Finally, lifestyle, the tendency of people to establish and exercise a social group identity through dress, commercial consumption, architecture, cars, home decoration, landscaping, leisure activities, and so on, is a key driver of environmental decisions and impacts on urban biodiversity (Grove *et al*. 2006). Lifestyle differentiation is such a powerful feature of contemporary society that in the United States commercial, charitable, and political marketing strategies are devised based on the spatial distribution of different lifestyle groups. For example, the marketing firm Claritas (1999) recognizes 62 distinct lifestyle groupings in the United States, and these have been useful ecological predictors (Troy *et al*. 2007).

This survey only introduces the complexity and multiplicity of social factors that can directly or indirectly affect urban biodiversity. However, it does suggest that it is social factors that determine 1) the access to and control of the land resource that biodiversity requires in urban areas, 2) the financial resources and social dynamics underlying actions that are relevant to biodiversity, and 3) provide the knowledge and shape the vision urban landowners and residents employ in designing and managing urban land cover.

3.5.4 Urban metacommunity properties

Ecological theory tells us that spatial heterogeneity and habitat connectivity are important for successful dispersal between habitats. Furthermore, environmental filtering via disturbance or productivity gradients influence species richness and community assembly (Pickett *et al*. 2010). Such factors are fundamentally altered in the urban environment for multiple reasons, including physical, biological, and socio-economic. Recognition of these emerging processes in urban ecosystems suggests that there are new, fundamentally different mechanisms by which local communities are assembled. We suggest one new area of inquiry is how patterns in species composition in urban ecosystems might be of equal or more importance than patterns in species diversity. The strength of the metacommunity approach is that it focuses on mechanisms driving community assembly from regional species pools, and the factors influencing species turnover in space. Here, we outline two extreme sets of conditions: communities structured primarily by the distinct environment filters associated with anthropogenic alteration of the urban landscape, a process we call 'self-assembly', and communities structured primarily by humans intentionally encouraging or assembling species at a location, a process we call 'facilitated assembly'. The two extremes reflect a widely recognized contrast between direct and indirect effects of urban physical and human environments (Effland & Pouyat 1997; Pickett & Cadenasso 2009). From these two extreme endpoints, we can further refine basic ecological and socio-economic theory to develop specific, testable predictions that allow us to diagnose the mechanisms structuring urban biotic communities.

3.5.4.1 Self-assembly of urban ecological communities

Urban ecosystems present species with a fundamentally unique geophysical template (Zipperer *et al*. 2000; Pickett *et al*. 2001). The environment can be viewed as harsh, especially in areas not directly managed for species composition. We term the process shaping the communities responding to such human disturbance 'self-assembly'. Here, humans influence assembly through indirect effects of an environmental template set by their actions, not through direct movement, intentional manipulation of species, or preparation/management of habitat. Vacant lots, abandoned property, roadsides, streams and rivers, and stormwater retention ponds occur as unique habitats or are impacted as a consequence of human alteration to the environment, but are not directly managed for species composition. These habitats have a strong environmental filter, excluding species that cannot maintain themselves in the highly disturbed and often polluted urban ecosystem. Species composition is a consequence of human presence and the larger-scale decisions made about how the urban environment is physically structured. Indeed, this perspective has much in common with the ecosystem-stress hypothesis (Menge & Sutherland 1987) and the intermediate disturbance hypothesis (Connell 1964) that are often invoked to explain patterns in biodiversity in urban places (McKinney 2002), or even in other anthropogenically stressed places (e.g. agroecosystems). However, metacommunity theory and the broad concept of patch dynamics emphasizes that these familiar hypotheses must be considered in a spatial context (Pickett *et al*. 2000; Wu & Hobbs 2007). This set of theory predicts a convergence of species that, among other traits, tend to disperse well, are weak competitors, and have high population growth rates (Blair 2001; Schwartz *et al*. 2006). Such communities might be prone to dominance by a single species and susceptible to invasion by non-native species (Shochat *et al*. in press). The socio-economic context for understanding self-assembled communities lies in the decision-making process that drives the patterns of environmental constraints across the urban landscape.

3.5.4.2 Facilitated assembly of urban ecological communities

Humans facilitate the establishment of species for many different reasons. We term the process shaping these communities 'facilitated assembly', with 'facilitated' meaning humans directly manipulate the presence (or absence, e.g. of pests) of some species over others. Perhaps the most obvious example is the garden. People plant species in various combinations based on basic habitat needs, and further enhance the habitat to either maintain the aesthetically pleasing nature of their garden, or to boost productivity of harvestable species (Bhatti & Church 2004; Daniels & Kirkpatrick 2006a). Although the privately owned garden is certainly a prime example, plant species are also assembled and maintained on corporate and public landscapes. In almost all cases, the species assembled in such communities did not disperse there naturally, nor would they likely thrive without human intervention (e.g. fertilizer, weeding). They are, in part, disconnected from their native life-history requirements. Natural dispersal needs are sidestepped, populations need not be maintained via local reproduction, trophic interactions are relaxed, and even death rates may play no role in species demography. Such assemblages remain a persistent pattern of biodiversity in the urban ecosystem, yet basic ecological theory cannot explain these patterns without taking a socio-economic perspective (e.g. Martin *et al*. 2004; Kinzig *et al*. 2005; Daniels & Kirkpatrick 2006b; Kirkpatrick *et al*. 2007).

Human facilitation of species establishment at a particular location includes various aspects of management. Parks and greenspaces, for example, might be preserved and not necessarily planted, but without human action they would not persist. Restoration is an excellent example of the human intention to facilitate assembly, if habitat rehabilitation or improvement is a goal. For example, geomorphic restructuring of a streambed might be designed to increase the proportion of sensitive fish and insect species (e.g. Groffman *et al*. 2003). Humans may not actually move species, but purposely create circumstances by which desirable species should be attracted. In the metacommunity context, where dispersal and local environmental factors interact to influence community assembly, the dispersal mechanism is different because people are moving species around. Environmental filtering, and thus the mode of species sorting, is mediated by human manipulation of habitat to support desirable species. Essentially, metacommunity processes that depend on life-history attributes (e.g. reproduction) become disconnected from the pattern we see in urban ecosystems due to human-mediated assembly.

3.5.4.3 Self- versus facilitated assembly as a gradient

We make the distinction between self- and facilitated assembly processes based on (1) the extent of habitat degradation caused by human modification of the geophysical environment, imposing a strong environmental filter on naturally colonizing species; and (2) the degree of human facilitation of desirable species, via either assisted dispersal and/or maintenance of ideal habitat. However, there are many examples of species from the extremes proposed that might occur on both habitats. Indeed, humans have done a very good job of facilitating the introduction of non-native, exotic, and eruptive species from self-assembled communities into nearby habitats. Furthermore, humans go to considerable lengths to eliminate undesirable species from highly managed habitats, via focused use of pesticides/herbicides and weeding, or jurisdictional management plans to rid the landscape of pests (e.g. mosquitoes). Our goal here is not to highlight the obvious fact that undesirable species occur in urban ecosystems, but that a certain amount of 'leakiness' via natural dispersal and exploitation of habitat by opportunistic species occurs between the extremes we identify. Armed with this observation, biodiversity and species turnover in particular can be better understood and predictions made about how species composition varies in space in urban places.

3.5.5 A conceptual model of urban metacommunities

The metacommunity concept characterizes communities as embedded in a landscape where a balance of local (e.g. environmental gradients) and regional (dispersal) factors explains patterns in both species composition and turnover across localities. In urban ecosystems, environmental gradients are created and/or changed either purposely to facilitate local species composition, or as a consequence of human modification of the geophysical template. Interspecific dispersal patterns are also altered by humans, such as by planting species together or via management/restoration techniques, or by imposing a strong environmental filter so as to eliminate species from thriving in a location. Taken together, local versus regional processes known to be important in governing species assembly in anthropogenically benign habitats need to be revised to aid in explaining the mechanisms driving patterns in urban biodiversity.

Chase *et al*. (2005) proposed a graphical approach to understanding local versus regional processes in governing species composition across the landscape. If regional effects (i.e. dispersal) are an overriding factor, then there should be a negative relationship between community similarity at two locations and the distance between those two locations. This is the distance decay relationship used commonly in biogeography. The slope of the line is beta diversity, or species turnover from one location to the next. The second approach is to estimate the relationship between community similarity between two communities and the environmental similarity of the habitats. A positive relationship suggests local effects, such as environmental gradients or species interactions, are influencing species composition. Taken together, the two approaches can paint a picture of the role of local versus regional processes in shaping community composition in space. Furthermore, the relationships are empirically tractable, and basic statistical methods (linear and nonlinear regression) can be used to test them.

We organize the conceptual relationship between local community composition, mode of assembly, and the feedback to the socio-economic template (Fig. 3.5.1). For the communities we term facilitated, local species composition is a non-random subset of the regional species pool. Species arrive at a location because they were physically transported by humans, as in the case of gardens, or their establishment facilitated via management or restoration activity. We propose a strong feedback to how social groups both create, value, and persist in maintaining species composition on the landscape. As such, we predict high species turnover or beta diversity among these communities due to varying mechanisms that facilitate the coexistence of species due to diversity in the underlying socio-economic drivers (positive relationship between community similarity and environmental similarity; Fig. 3.5.2a, bottom panel). Furthermore, it is likely that facilitated communities are predominately, but not exclusively, controlled locally by private landowners, tenants, or community-based organizations. A realistic exception would be management of parks and greenspace by local and regional governments. Dispersal limitation, or a negative relationship between community similarity and distance, should not be evident unless there is a complementary correlation between distance and mode of human facilitation of assembly (Fig. 3.5.2.a, top panel).

Self-assembled communities occur because strong environmental filtering selects species that can deal with the unique urban environment (Sukopp *et al*. 1990; Arnfield 2003). The highly modified geophysical template driven by regional-level infrastructural policies leads to the establishment of communities with very similar traits (e.g. high dispersal ability;

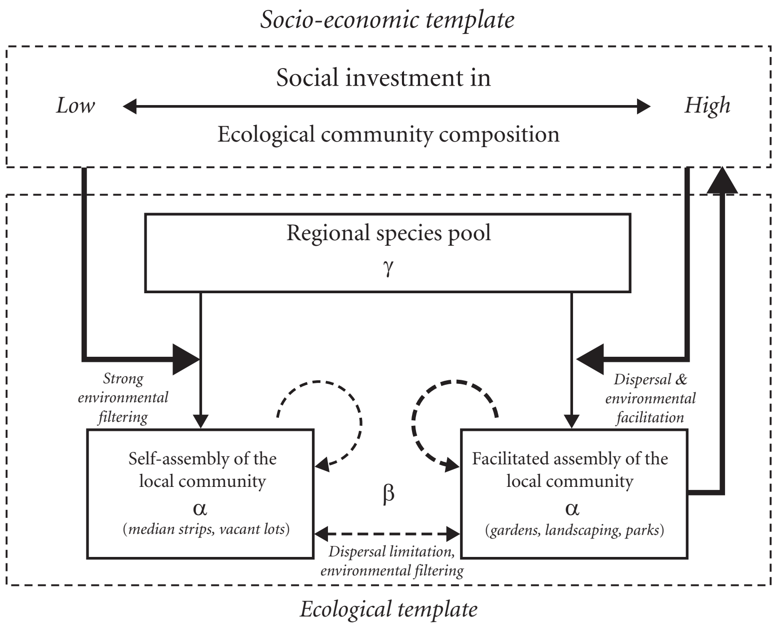

Figure 3.5.1 A conceptual model of the influence and feedback of socio-economic factors and the assembly of ecological communities. Self- and facilitated assembly represent the processes generating subsets of species combinations (α) from the regional species pool (γ), with control on assembly governed by the regional geophysical template, or local human perception and behaviour. The magnitude of species turnover (beta diversity; β) should be large among facilitated communities and between self-assembled and facilitated communities, but strong environmental filtering should constrain species composition regardless of location in the self-assembled category. We equate low social investment in community composition with the disregard for ecological integrity that results from development of the built environment. As such, it is a unidirectional constraint on local species composition (left). However, renewed interest in making cities sustainability might have the potential to create a feedback between desirable ecological assemblages and human behaviour (right)

Marzluff 2001; Pouyat *et al*. 2002; Schwartz *et al*. 2006). Locations such as median strips, stormwater ponds, and vacant lots exhibit strong within-habitat similarity and exhibit very low beta diversity (Fig. 3.5.2c, top panel). The strong environmental pressures should, however, reveal a species-sorting signal, with a positive relationship between community and environmental similarity (Fig. 3.5.2c, bottom panel). Until the value of the species composition on the landscape becomes integral to how decisions are made regarding infrastructure at the regional level, as may soon be the case as jurisdictions develop sustainability plans, we suggest no feedback between self-assembled communities and socio-economic factors. Instead, a ripe area for inquiry will be how facilitated communities influence local citizen groups, and if such influence can bring change to bear on the regional policy decisions (Fig. 3.5.1).

Species turnover between self-assembled and facilitated communities is likely to be a combination between distance between locations and environmental filtering (Fig. 3.5.2b). Exchange of species between these two community types might certainly be a function of distance, with larger exchanges occurring in communities closer together (aggregative effects; Fig. 3.5.2b, top panel). This would largely be due to the potential for less dispersive species in facilitated communities escaping into unmanaged habitat but failing to establish far away. However, to exist anywhere in the urban ecosystem, there are certain niche requirements that might accommodate any species, although not completely in all cases, such that there might be a species-sorting signal (Fig. 3.5.2b, bottom panel) but not as strong as in completely self-assembled communities.

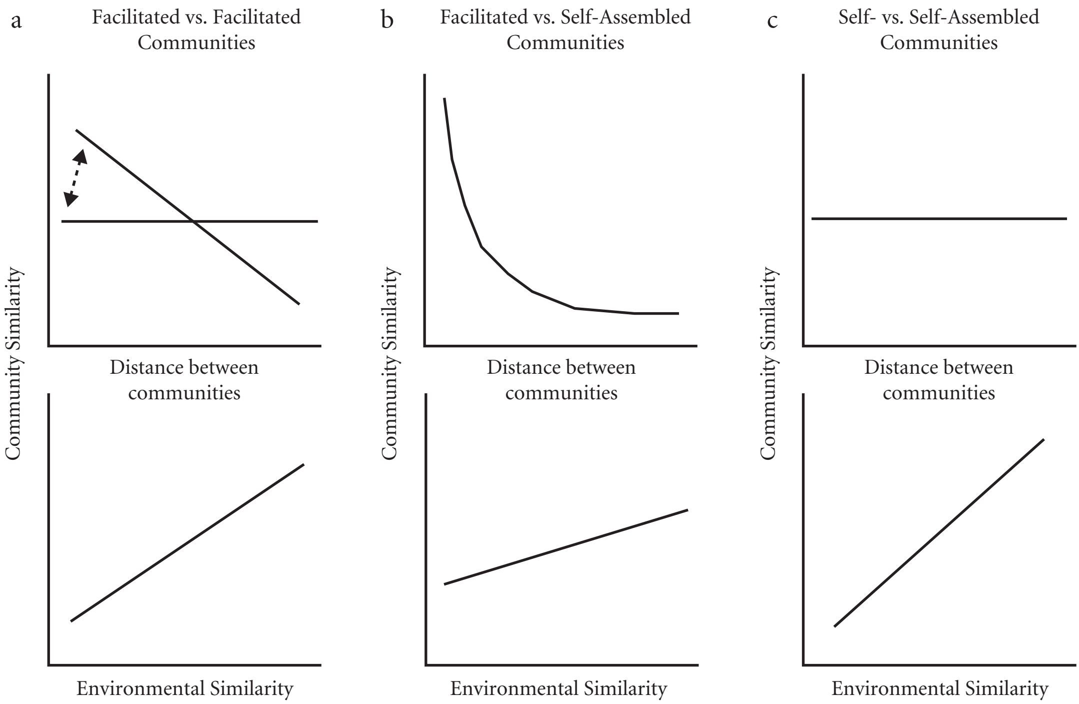

Figure 3.5.2 Predictions of regional (dispersal-driven) effects (top panels) and local (species-sorting) effects (bottom panels) for comparisons within facilitated communities (A), self-assembled communities (C), and between each (B). In facilitated communities (A), human behaviour drives the environmental conditions supporting species composition, resulting in a positive relationship between community similarity and environmental similarity (A, bottom). Given the propensity for humans to move species around, dispersal-limitation in facilitated communities should be low, resulting in no relationship between community similarity and distance (A, top). However, if there is a correlation between environmental similarity and distance, the distance–decay relationship should increase, as indicated by the dashed arrow. For self-assembled communities (C), intense alteration of the geophysical environment imposes strong environmental filtering on colonizing species, revealing again a strong relationship between community similarity and environmental similarity (C, bottom). However, since most disturbance tolerant species are generally good dispersers, and the stress of the urban environment regionally pervasive, no dispersal limitation should be apparent (C, top). The facilitated and self-assembled communities define endpoints of a gradient, requiring predicted be made between the two (B). Community similarity should decline rapidly with distance since invasion of facilitated communities should occur from mainly from geographically close localities (C, top). Furthermore, habitat conditions in facilitated environments need to meet the needs of self-assembled species (and vice versa), suggesting a positive, but perhaps weak, relationship between community and environmental similarity (C, bottom)

3.5.6 Conclusions

The metacommunity concept is a useful tool to begin to explore how local versus regional processes shape local community composition. In the context of urban ecosystems, socio-economic factors can easily be applied to understand why species occur together, how they arrived, and what habitat conditions support their coexistence. The novel, but perhaps often overlooked, observation that biodiversity in urban ecosystems includes more than remnant habitats, parks, damaged streams, and abandoned lots, by including highly managed and manipulated landscapes, suggests a very feasible link between socio-economic patterns and processes and ecological theory. Basic ecology cannot explain completely why certain species occur in a backyard garden. These species did not disperse there naturally, nor are they experiencing natural levels of predation, herbivory, or competition, or even reproducing locally. Yet these species are a subset of urban biodiversity, and theory needs to be revised to explain processes leading to their coexistence. The metacommunity concept, with refinement, can be a springboard to developing specific, testable hypothesis to continue to expand the scope of community ecology to the urban environment.

Summary

Nancy E. McIntyre

The five chapters in Section 3 clearly illustrate that organisms are facing unprecedented challenges (and opportunities) from having to share the planet with an ever-increasing number of us and our constructions. From these chapters, we learn that relative to the indigenous 'background' landscape, cities support an altered biota, with patterns varying within and across taxonomic groups. For example, whereas the diversity of herpetofauna (reptiles and amphibians) may be lower within cities than in the surrounding countryside (chapter by Grant *et al*.), bird species richness may remain relatively constant, albeit dominated by 'urban adapted' species (Clucas and Marzluff). Plant biodiversity may actually be higher within cities than outside them, but this effect is typically due to the presence of accidentally introduced or deliberately planted non-native species (Cilliers and Siebert). And even within a single taxonomic group, different species may show differing tolerances or affinities for urban development, depending on their life-history needs (Kotze *et al*.). Through processes common to all ecosystems, as well as those unique to only anthropogenic ones, these losses and gains create an urban metacommunity that can be used as a model to elucidate general principles regarding the maintenance of biodiversity (Swan *et al*.).

The chapters in this section take a comparative approach, examining organisms from cities around the globe. This approach illuminates the fact that although species may differ from city to city, there are commonalities among urban biotic communities around the world. These commonalities are due to the fact that humans are a common element among all cities, with our actions serving as agents of disturbance, fragmenting native habitats and creating uniquely urban ones, sometimes at the expense of certain taxa. Other organisms, however, are able to live and even thrive in these urban habitats because their resource needs are met. For some species, in fact, cities may serve as oases with dependable supplies of food, water, and shelter. Indeed, for some species, this relationship has been so well-established and long-standing that we now think of them as being archetypically urban, such as pigeons or cockroaches—organisms that are not only found within cities, but in many cases are found *mainly* within cities. But even species of conservation concern may be found within our cities (e.g. falcons nesting on the ersatz cliff-faces that our skyscrapers provide), illustrating that urban ecosystems are far from barren and that 'urban wildlife' is neither a joke nor a denigration. Examining the ecology in cities is thus a delicate balancing act between being optimistic about organisms that persist or prosper within urban settings, while simultaneously being mindful that many organisms simply cannot or will not tolerate urban conditions. The task for future urban ecologists will be to understand and manipulate the factors that generate and govern the abundance and distribution of organisms within cities.

These chapters indirectly reveal that there has been little work at the interface between community and urban ecology. Although there have been studies that examined individual communities in urban settings, they were not studies of urban community ecology per se. The processes that drive community assemblage in urban ecosystems have been assumed to be the same as in non-urban ones, but this assumption remains to be critically examined. Certainly urban communities are influenced by exotic species more so than are non-urban ones, and urban organisms also are subject to altered productivity regimes (often with dampened fluctuations). These and other factors, found in all cities, act as selective forces that cull those species incapable of tolerance or adaptation to urban life, generating the biotic homogenization so characteristic of contemporary cities. The metacommunity approach advocated by Swan *et al*. should help us advance our understanding of how and why certain species thrive whereas others are eliminated from urban ecosystems.

Finally, these chapters demonstrate that cities are worthwhile arenas for ecological research. Although some research in the nineteenth and early-twentieth centuries examined organism–environment relationships within cities, urban ecology only became widely identified as an independent specialization of ecology in the last few decades of the twentieth century. Due to the field's relative youth, as well as to the accelerating pace of urban development, a global synthesis of the state of our knowledge is necessary if we are to influence future urbanization in such a way as to conserve biodiversity—hence this book. Although the chapters in this section summarize what is known about organisms in urban environments, much remains to be understood. Some fruitful areas for future research include examination of less well-studied taxa (particularly urban mammals, fish, and non-terrestrial arthropods), inclusion of cities from more parts of the world (particularly South America and Asia), and design of experiments that will elucidate mechanisms behind the urban ecological patterns we are still in the process of discovering. And future research in urban ecology should examine influences from the sociological environment as well as from 'typical' environmental variables.

Our world is becoming an increasingly urban one. Both the human and non-human occupants of cities are facing an evolutionary test unlike anything previously experienced. Moreover, just as organisms are affected by urbanization, so are the ecosystem functions that they perform. As a consequence, foodweb dynamics, decomposition rates, nutrient cycling, pollination/seed set, and carbon mitigation are altered relative to non-urban regions, to name but a few. The implications of these patterns on extinction risk, genetic and species diversity, human perceptions of and experiences with wildlife, and other facets are currently unknown; future urbanites will come to know the consequences of current urban design. We must use what we learn to guide land-use planning and development, combining science with design and policy. The sustainability of our urbanizing world literally hangs in the balance.

SECTION 4

Ecosystems, Ecosystem Services, and Social Systems in Urban Landscapes

SECTION EDITOR: **Thomas Elmqvist**

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Introduction

Thomas Elmqvist

Ecosystem services are the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfil human life and are often defined as 'the benefits human populations derive, directly or indirectly, from ecosystems' (TEEB 2009). The concept of ecosystem services has proved to be useful in describing human benefits from ecosystems. The Millennium Ecosystem Assessment—MA (2005)—prepared by a group of over 1,300 international experts, found that 60 per cent of ecosystem services assessed globally are either degraded or being used unsustainably. Seventy per cent of the regulating and cultural services evaluated in the assessment are in decline. However, there are considerable knowledge gaps about urban ecosystem services. MA covered almost every ecosystem in the world, but barely mentioned urban systems. On the other hand, the World Development Report (World Bank 2009), the world's largest assessment of urbanization, left out ecosystems. The goal of this section is, therefore, to contribute to fill the knowledge gap and review and discuss the scientific literature focusing on understanding the complexity of urban social-ecological systems and the interplay between humans and ecosystems through the lens of ecosystem services generated both locally and on a regional scale.

In Chapter 4.1 McDonald and Marcotullio start with addressing ecosystem services from a global perspective and ask a question that at first looks very simple but in fact is extremely complex: Is urbanization good or bad for the environment? The question is used as a starting point for a broad overview of how cities affect the environment and whether there is any alternative to global urbanization. The chapter discusses this based on analyses of supply and demand of some specific ecosystem services, including freshwater and climate regulation. The authors conclude that one answer to the question is that at a local level, cities may negatively impact some services such as freshwater, but at a regional or national level, concentration of human population in dense settlements may often provide benefits.

Redman in Chapter 4.2 asks the question: To what extent it is relevant to analyse episodes from the past for dealing with problems of the present and future urbanization? While the past does not offer simple, detailed, directly applicable solutions to the challenges of the present, the past does offer essential insights into the nature of socio-ecological interactions under a huge number of diverse conditions and the implications of the decisions that were made. After an historical overview of the rise of urban complexity, including the diverse set of drivers of urbanization through time, Redman addresses resilience in the urban context. Enhancing resilience of a system often means encouraging flexibility and adaptive capacity in the forms of redundancy, inclusiveness, monitoring, and preparedness for multiple futures. All of these efforts have associated short-term costs in a world striving for efficiency and 'lean' organizations, and lessons from history tell us the desired outcome may not be very easily achieved.

In Chapter 4.3 Alfsen *et al*. point out that cities are connected social, cultural, and ecological systems. The chapter gives an overview of ecosystem services in the urban landscape and emphasizes that there is a need for a holistic approach to governance and management. The ecosystem approach is a strategy for the integrated management of land, water, and living resources that promotes conservation and sustainable use in an equitable way but does not, however, make the link between ecosystem functions and human security explicit. Today, with increased incidence of natural disasters and the looming threats associated with climate change, there is a deeper understanding of immediate and inextricable connections between human security, well-being, and functioning ecosystems. Our current systems of governance, resource allocations, and public information have not, however, adapted in response to these threats. Several global initiatives aimed at improving governance of urban ecosystems, such as ICLEI/LAB and URBIS, are presented.

In Chapter 4.4 Bridgewater points out that there is a close cultural connection between urban settlements and wetland systems since many urban areas have been founded along the banks of great rivers, areas of freshwater seepage, or where groundwater is (or increasingly was) close to the surface. Bridgewater argues that the ecosystem approach could also be used as an organizing frame for improved management and governance of constructed wetlands. The role of these so-called artificial systems in ensuring continued functioning of ecological systems, and maximizing the expression of wetland biodiversity in urban and peri-urban systems, is highlighted.

Finally, in Chapter 4.5 Colding addresses ecosystem services in the context of urban spatial planning. Urban planning has traditionally meant planning 'for development' and recently shifted more towards 'sustainable development'. However, even such planning often ignores ecosystems and the natural configuration of the land. As a consequence, ecosystems are rapidly becoming fragmented, transformed, or entirely lost, causing loss of ecosystem services, such as impoverishing water and air quality. This may erode ecosystem resilience, or the capacity of natural systems to buffer and reduce disturbances like heat waves, flooding, pollution, and human-induced management mistakes. Colding reviews the current development of smart growth and green infrastructure planning strategies and concludes that there exists a whole arsenal of green designs that can be adopted in cities to provide a range of ecosystem services, promote sense of place, and revitalize degraded neighbourhoods.

As evidenced from this section there is a rapidly growing literature and awareness of the values of urban ecosystem services and strategies for improving management and governance linking the local to the global. However, urban areas are in continuous and rapid change and are facing enormous challenges, such as climate change and transformation to a future beyond oil. Ecosystems may have a large role in facilitating this transformation. Ecosystems provide flexibility in urban landscapes and help build adaptive capacity to cope with increased temperature and changing precipitation, for example, and promote human well-being through ecosystem services. However, knowledge is lacking and there is a need for creating learning arenas. Everyone involved in urban development, policymakers, urban designers, scientists, and planners need to facilitate, to a larger extent, more experimental designs and learning about the interactions between humans and ecosystems. Such strategies may gain from insights on adaptive comanagement of ecosystems and lessons provided from sustainable natural resource management, to increase the potential for adaptive learning and the avoidance of vulnerability traps during the process of urban transformation.

CHAPTER 4.1

Global Effects of Urbanization on Ecosystem Services

Robert McDonald and Peter Marcotullio

4.1.1 Introduction

Many chapters in this book mention the demographic fact that more than 50 per cent of *Homo sapiens* live in urban areas (UNPD 2007). This statistic is perhaps a bit stale, but many scientists still rehash it (e.g. McDonald 2008) to emphasize how truly dramatic global urbanization is. In tens of thousands of cities around the globe (Fig. 4.1.1), concrete is being poured, bricks are being laid, and the great cities of the twenty-first century are being built. Humanity, in a metaphorical and sometimes literal sense, is on the move. It seems appropriate, then, to step back and ask what urbanization might mean for the environment.

This chapter attempts to answer a simple question often posed to the authors: Is urbanization good or bad for the environment? This question is so simplistic, but concerns such a complex topic, that perhaps it is unanswerable. Here we use it as a starting point for a broad overview of how cities affect the environment. Our discussion is focused on past several decades and the predictions of what the next decades may bring, in contrast to the long-term historical perspective of the following chapter by Redman. We first examine whether there is any alternative to global urbanization, which is implicit in our overarching question. Next, we provide some background information on the urban environmental transition, a theory that takes a historical perspective on urban-environmental relations. Similarly, we describe the concept of ecosystem services, which play an important role in our analysis. The bulk of the chapter is taken up with specific examples of urbanization's effect on provisioning, regulatory, and supporting services. We end by returning to our big question to analyse how close we can get to answering it.

4.1.2 What is the alternative to urbanization?

Ironically, in an age of advancing global urbanization, cities are taking a lashing from environmentalists. Lester Brown (2001, p. 188), for example, emphatically states that 'Cities are unnatural' and costly too. He argues that 'People living in cities impose a disproportionately heavy burden on the earth's ecosystems simply because so many resources must be concentrated in urban areas to satisfy residents, daily needs' (Brown 2001, p. 190). This viewpoint has been supported by the notion of the vast ecological footprints of cities (Wackernagel & Rees 1996). The result, in the eyes of ecologist Eugene Odum (1991), is that a 'realistic' portrayal of the cities-to-Earth relationship is of that of parasite-to-host.

We question these categorical conclusions, by attempting to separate urbanization from other processes. Moreover, we ask, is there a plausible alternative to urbanization? Are there societies that are wealthy and not urban? Are cities being fled from as outdated technologies and modes of organization? The answer to all these questions is categorically, no. We define urbanization as a spatial phenomenon: the concentration of population. Many other changes occur with concentrating population during the development process. Urbanization contributes to these changes as both an outcome and significant driving force. Given the

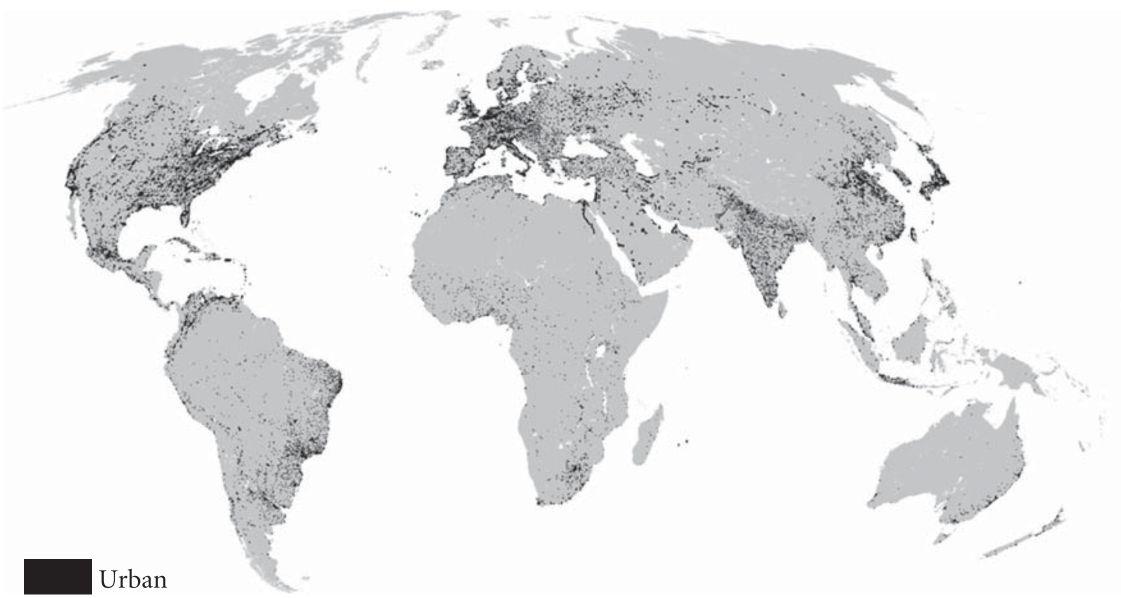

Figure 4.1.1 Urban land cover, as defined by the Global Rural/Urban Mapping Program (GRUMP alpha 2004). To make urban area visible at this scale, the width of each urban area is expanded slightly

powerful intertwining of forces that associate development with urbanization it is futile, if not dangerous, to promote dramatic alternatives.

Demographers, urban geographers, economists, and planners point to a number of factors that promote urbanization (for a review see Montgomery *et al*. 2003). Urban economic growth is related to: communications and transport; economies of scale and agglomeration economies; personal contact among workers and entrepreneurs; and efficiency gains from the high population density in cities. Urbanization is also strongly correlated with changes in population structure and decreases in fertility. These dynamics bring substantial benefits for and changes to industries and society (Montgomery *et al*. 2003).

Economists have noted that firms often gain benefits from being clustered near other similar firms, such as lower costs of production and increased specialization of the workforce. These benefits are examples of economies of scale and they encourage firms to agglomerate together in cities. Agglomeration economies are an important contribution to urban economic growth and wealth creation.

If transport and communication costs are high it becomes costly to move people, goods, and ideas. High transport costs enhanced urban growth during the early industrial revolution and in many cases cities developed around trans-shipment points, where reduced transport costs mixed with the economic benefits of goods handling (Anas, Arnott, & Small 1998). However, over the past few decades, transport costs have been reduced and communication has become easier, faster, and more convenient. Within urban regions, workers can commute long distances, leading to concern about urban sprawl and an expansion of urban area in many rich countries. Companies have dispersed across countries, regions, and even the globe. Moreover, in the developed world, economic structures have shifted from largely manufacturing to services. This decentralization of industry has resulted in the re-constitution of urban employment around service production, not manufacturing.

While technologies have reduced and simplified long distance connections and production has decentralized, the dispersion of population out of urban areas predicted by many (Toffler 1980; Webber 1964) has not emerged. Cities continue to exist and thrive. It is not just cheap transport and communication and the agglomeration of production that advances urbanization. Personal contact enhances levels of trust and ensures confidentiality, among other important business fundamentals. This appears especially important for the production of advanced business and producer services, which is now the source of agglomeration economies in particular cities (Sassen 2001). The importance of personal contact has not diminished with technological advances (Montgomery *et al*. 2003). Indeed, technology and personal exchanges may often be complementary, forcing activities to agglomerate rather than disperse (Glaeser 1998).

Other factors responsible for continued population concentration include the fact that cities lower the costs per household and per enterprise for the provision of infrastructure (roads, pedestrian paths, piped water, sewers, drains, electricity) and services (day care, all forms of schools, healthcare and emergency services), and make it easier to govern large populations (Satterthwaite 2007; Bairoch 1988). Rising incomes within cities also support the development of specialized private markets (Montgomery *et al*. 2003). Given these dynamics, the long suggested correlation between urbanization and income (Williamson 1965) remains important today (Satterthwaite 2007).

Demographers also note that changes in employment, population structure, and household fertility accompany urbanization. As people move to cities they leave the agricultural sector for employment in industry and services. Fertility is typically lower in cities than in rural areas (UNFPA 2007) as children become financially more burdensome. Hence, part in parcel with the demographic transition, societies urbanize.

Because urbanization is tightly bound up with a combination of processes often called development, there is simply no counterfactual to the emergence and dominance of societies organized around dense settlements. All economies urbanize and there is not a country with high economic development that has not also experienced urbanization. In some regions, such as Latin America, urbanization was experienced at low levels of income, but by and large, low-income societies are less urbanized and high-income societies are urban.

That is not to say that national governments have not tried to control urbanization. As pointed out by the UNFPA (2007), there are a growing number of economies that have implemented policies to lower migration to urban agglomerations; from 51 per cent in 1996 to 73 per cent in 2005. The UNFPA suggests this is a sign of growing 'anti-urban' policy bias. More importantly, these policies have had little long-term effect on urbanization and an arguably negative impact on economic growth. Indeed, some have argued that to reduce poverty, societies should accelerate urbanization.

Given the imperative that societies continue to organize spatially around and through cities, our task is to tease out the impacts of spatial concentration from other aspects of development and to uncover the complex ways in which activities in urban areas can be changed so as to promote sustainable development.

4.1.3 The urban-environmental transition

One powerful tool with which to explore the relationship between the environment and urbanization is urban-environmental transition (UET) theory (Fig.4.1.2, adapted from McGranahan *et al*. 2001). This notion argues that dominant conditions in urban environments shift along three dimensions with increasing wealth. The UET argues that as poor cities grow in wealth and begin the rapid industrialization process,

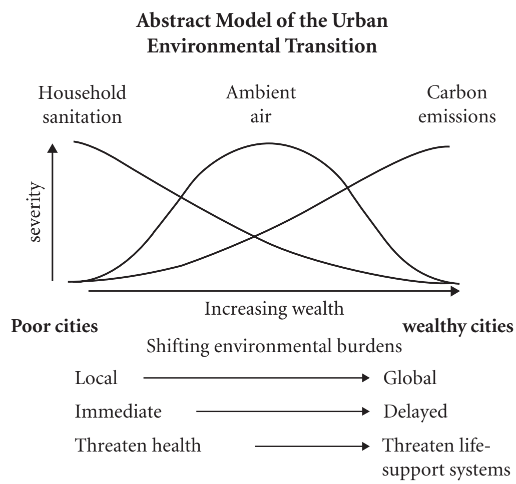

Figure 4.1.2 The urban-environmental transition. As cities get wealthier, the type of environmental burdens they face changes systematically (McGranahan, 2001). Reproduced with permission of Earthscan Ltd www.earthscan.co.uk

environmental priorities shift from the 'brown' agenda, which prioritizes concerns such as inadequate water supply and sanitation, indoor air quality, drainage and solid waste disposal, to those at the metropolitan–regional scale. At the metropolitan scale the dominant issues include water and air pollution, sometimes called the 'gray agenda' (Marcotullio & Lee 2003). As cities continue to grow in wealth, urban environmental burdens shift again from metropolitan scale challenges to those most easily observed at the regional and global scale. Global scale challenges are associated with high consumption and emissions levels and the 'green' agenda includes such concerns as acid rain, water scarcity, greenhouse gas, and ozone depleting substance emissions.

The UET is not only a powerful tool for understanding shifting environmental conditions within cities, it also complements theories of development. Historically, sometime during the development process of Northern economies, a series of transitions occurred making urban life more hospitable than rural life. Residents of cities began to live longer and healthier lives (due to better diets and healthcare), increase household income faster, and generally enjoy a better quality of life than their rural counterparts (see for example Haines 2001). With these shifts there was also a shift in the intensity of selected per capita environmental impacts of rural versus urban residents, such that urbanites became more efficient and had lower impact than rural peoples. For example, urban residents now use less energy per capita and therefore have lower greenhouse gas emissions than their rural counterparts (Brown *et al*. 2008). This may not be true in early stages of development (IEA 2008). Hence, the UET complements descriptions of social, economic, and environmental changes with development in the now developed world.

As with development, however, shifts in environmental burdens and agendas are not only due to increasing population concentration; increasing wealth, technology, political, social and environmental crises, and institutions also play a role. Technological advances have been considered vital to understanding changes during development (Pacey 1991; Diamond 1999) and have facilitated environmental transitions. For example, motorization was a technological solution to the health crises in cities due to urban horse urine and faeces (McShane 1994). A number of studies have also emphasized the importance of crisis in promoting dramatic changes in social organization and environmental conditions (Holling 2001). Changes in institutional context are also vital in explaining the history of development (North 1990). In the United States, a series of institutional shifts at the local, state, and federal level helped to facilitate the consolidation of large metropolitan areas at the turn of the nineteenth century, promoting development of large infrastructure projects during the middle of the twentieth century, generating environmental policies that attempted to reduce pollution during the 1970s and 1980s, and providing the global context from which cities, nations, and regions are attacking the greenhouse gas problem.

Moreover, as mentioned above, the drivers of urbanization have changed over the past decades, with globalization, advances in technologies, new crises (financial, social, and environmental) and the emergence of new institutions at all levels of governance, and so too has the shape of the urban environmental transition changed. The linear, staged sequential pathway of North American urban environmental transitions no longer exists for currently rapidly developing economies. Specifically, the transitions now occur faster, sooner, and more simultaneously (Marcotullio 2007). For example, consider a poor or lower income city in South-east Asia. Rather than simply experiencing challenges at the local level, urban residents are also faced with motorization and industrialization related pollution, urban sprawl, rising consumption, and even green agenda issues, such as climate variability and threats of sea level rise.

As most of the urban population growth is now in the developing world, the majority of urban residents are experiencing a number of environmental and ecosystem burdens simultaneously. These burdens are increasing at unprecedented speed. While these changes are frightful, these dynamics also provide opportunities. Some urban centres have taken advantage of these opportunities and have implemented integrated policies. For example, Singapore was one of the first cities to implement congestion pricing and Curitiba, Brazil, was one of the first urban centres to implement integrated land-use transport-planning and Bus Rapid Transit. While they exist, however, the number of cities that have benefited by integrated environmental planning remains low.

4.1.4 Ecosystem services

As emphasized in the introduction to this section, ecosystem services has emerged as one of the most popular concepts in ecology of the last decade, and one that has helped unify economics and conservation biology. In many cases, ecosystem services are the opposite of the environmental burdens often studied as part of the UET. For instance, water pollution (an environmental burden) becomes troublesome when clean drinking water (an ecosystem service) is no longer available because the filtering capacity of natural ecosystems is overwhelmed.

An ecosystem service only has value, in a utilitarian or economic sense, when there is sufficient natural provision of some quantity and there is someone who wants that quantity. Both the supply of the natural resource and the demand are equally important in determining the value of an ecosystem service. Within ecology, ecosystem service supply mapping has become fairly common (Chan *et al*. 2006), while techniques for understanding the spatial variability in demand for ecosystem services have been less explored, with the exception of the concept of area-dependent ecosystem services like pollination (Kremen *et al*. 2007). Cities, as the home of the majority of humanity, become centralized places of demand for and consumption of ecosystem services (McDonald 2009).

Ecosystem services vary in how close the place of generation must be to the place of demand. For instance, shade trees can significantly reduce air conditioning costs when within 10–20 m of a house. For parks to be used frequently for day-to-day recreation, they must be within a few kilometres of people's homes. Clean water for drinking must generally be taken from the same regional area in which a city exists. Some ecosystem services are in effect global, such as the ability of ecosystems to sequester carbon dioxide and thus mitigate climate change. Thus, the spatial scale at which cities consume ecosystem services varies greatly among services.

4.1.4.1 Clean water

Perhaps the most direct connection between human well-being and ecosystem services are provisioning services, resources humanity obtains from the natural world which are essential to our livelihood. Food, from cropland or rangeland, is an obvious example, for although agricultural systems are simplified versions of natural ecosystems, they would not be possible without supporting natural processes like soil nutrient and water retention. All the uses we make of plant cellulose for paper, cotton, and saw timber are another example, sometimes collectively called fibre provision. Provisioning services extend beyond these two examples to include most of the subject matter of natural resource management, such as fisheries. Provisioning services are also relatively easy to value monetarily, because the commodities they generate are often already traded in markets.

Here we consider one crucial provisioning service: access to clean water for drinking and other personal use. Natural ecosystems help humans have access to sufficient clean water. Forests and wetlands, for example, help filter out pollutants and sediments before they reach reservoirs, safeguarding water quality (Dudgeon *et al*. 2006). Similarly, forests can help generate sufficient water quantity in some climates, through water condensation on tree foliage, and wetlands can mitigate floods, through reducing the height of peak flow and slowing the movement of water through the hydrologic system. Water for urban residents is generally drawn from the same regional watershed in which it lies, giving this particular ecosystem service a unique spatial scale and directionality (upstream/downstream). It is worth noting that, over time, cities build aqueducts to increase the spatial scale from which they can draw water, often going far away to find an adequate supply (McDonald 2009).

The amount of water available varies globally over several orders of magnitude (Fig. 4.1.3, top). Desert and grasslands biomes generally have less available water than forests, so cities in these biomes must also generally deal with having a smaller water supply. However, there are many different ways cities can get water, which complicates this simple picture. Some cities draw their water from rivers, making them relatively dependent on natural flows of water, although dams can create reservoirs that moderate seasonal or annual fluctuations. Some cities draw from groundwater, reducing their

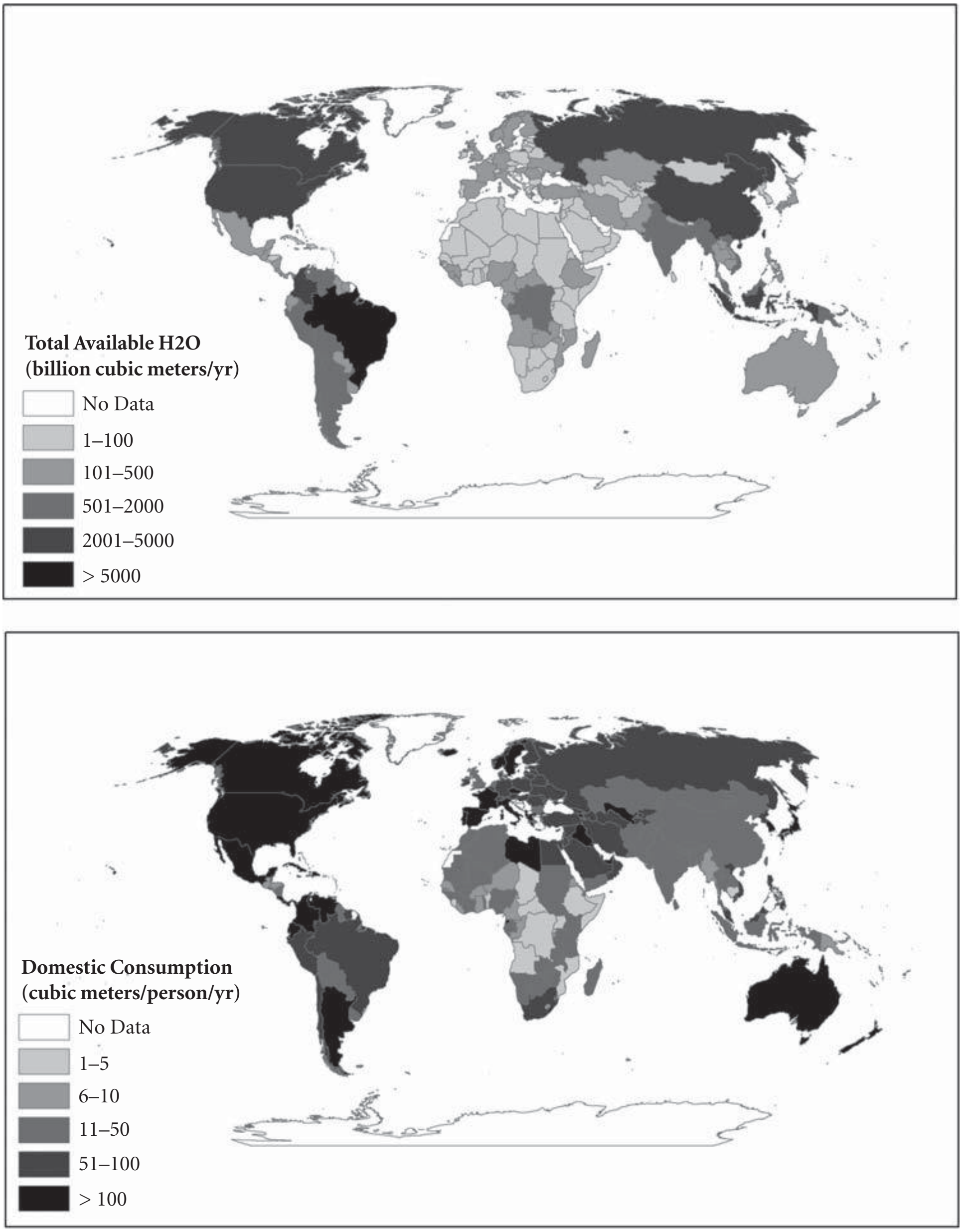

Figure 4.1.3 Global patterns of water availability (top) and domestic consumption per capita (bottom). Data from FAO AquaStat (2009)

dependence on annual rainfall inputs to their watershed. However, if groundwater withdrawals exceed aquifer recharge, as is the case in Mexico City (Birkle *et al*. 1998), the practice is unsustainable over many decades, and the city is essentially 'mining water'. Many cities are also dependent on snow and ice melt from higher elevations far upstream. Famously, most of the water supply of Los Angeles is snow melt from mountains more than 300 km away.

The amount of water used per capita varies among cities. One central challenge in analysing such statistics is how to define the conceptual boundary of a city's water use. At a national level, agriculture is always the largest consumptive use of water, often followed closely by industry. However, much of the output of these two sectors goes to urban dwellers, so in some sense a portion of these sectors' water use is also attributable to cities. Some insight into urban trends can be seen by looking at national level data on domestic withdrawal of water and dividing it by the total population in that country (Fig. 4.1.3, bottom). This is not an ideal metric of water use by urban residents, for domestic withdrawals are measured for water districts, which include some industry, and conflicting definitions of urban population make it impossible to use urban population rather than total population in the denominator (FAO 2009). However, it suffices to show a general pattern: countries with greater water resources tend to use more water per capita for domestic use, although this correlation is not particularly strong (R=0.11). Within the United States, more detailed city-level data reveals significant variation among US cities. For example, residents in San Diego use 700 litres/person/day, while residents in Reno, NV use 1,166 litres/person/day.

Per capita domestic water use tends to increase as the average income increases (FAO 2009). For example, the average resident of Indonesia ($3,900 GDP/capita, in purchasing power parity) uses 28.9 m^3/person/year, while the average resident of Canada ($40,200 GDP/capita) uses 276.0 m^3/person/year. The overall correlation between per capita domestic water use and per capita GDP is fairly high (R=0.59). There are at least two reasons for the increase in water consumption with income. First, richer cities have fewer (or no) citizens without access to clean drinking water, an enormous gain for human health but one that does increase aggregate demand for water. For instance, 27.6 per cent of Sub-Saharan urban residents lack access to clean drinking water, 12.3 per cent of Latin American and Caribbean urban residents, and essentially 0 per cent of urban residents in the United States (UN-HABITAT 2006). Second, richer urban residents have access to technology that requires significant water to run, such as dishwashers and washing machines. Moreover, there is little economic incentive for urban residents to conserve water, because water tends to be underpriced for social and political reasons (e.g. there are significant government subsidies to create most urban water districts). In addition, urban residents are often wealthy enough that, if charged for their water, they can afford to pay more for water than the agricultural sector. For example, in California urban water districts often buy water rights from upstream farmers.

In theory, rural to urban migration might lessen the population pressure on the environment in rural areas. However, as mentioned above, the agriculture sector is usually the main consumer of water in a country, so the fate of agriculture during urbanization is key to overall water use. Usually the agricultural sector remains on the same land area, after urbanization, albeit with fewer labourers per hectare of agricultural land. Moreover, migration to cities increases the purchasing power of newly urban residents, who in turn consume more calories and a different mix of food products. The net effect is generally more agricultural production, which takes more water.

On balance, then, urbanization in developing countries appears to increase water use. Urban resident use of water is a small part of the total water use. However, the rise in incomes associated with urbanization causes a rise in consumption, increasing the total water footprint of a city. There is potential for most of this increase in water use to be met through improvements in efficiency in agricultural systems. For example, significant amounts of water used in irrigation is lost to evaporation, a quantity that could be significantly reduced through other methods like drip irrigation.

4.1.4.2 Regulating services

Regulating services include a wide variety of functions that benefit humans through the control and maintenance of ecosystem processes. Examples of regulating ecosystem services include air quality maintenance, climate regulation, water regulation, erosion control water purification and waste treatment, regulation of human diseases, biological control, pollination, and storm protection. Regulating services may not be as well recognized as provisioning services, but attempts to put monetary values on all services suggest that regulatory services are the most important of ecosystem services for humankind (Costanza *et al*. 1997).

One important regulatory service is the ability of natural ecosystems to regulate climate, particularly in regards to humanity's greenhouse gas emissions causing global warming. Ecosystems worldwide are currently net sinks for CO_2. That is, they are absorbing and sequestering more carbon than they are losing or giving up. This was not always true. Terrestrial ecosystems were on average a net source of CO_2 during the nineteenth and early twentieth centuries, but became net sinks around the middle of the last century due to afforestation, reforestation, and forest management in North American, Europe, and China, among other regions, as well as due to the fertilizing effect of nitrogen deposition and increasing atmospheric CO_2 (MEA 2005).

Global atmospheric CO_2 concentrations have been dramatically impacted by anthropogenic activities, largely from the burning of fossil fuels and land-use change (Alley *et al*. 2007). In both cases urbanization and cities are implicated in generating more CO_2 than the Earth's ecosystems can absorb. Urbanization decreases natural vegetation cover, decreasing potential carbon sequestration locally, while cities house the majority of human activities that involve fossil fuel consumption (industrial and commercial activities, which are heavy uses of energy). Globally, while only 50 per cent of the world's population resides in cities, urban activities contribute more than 70 per cent of anthropogenic greenhouse gas emissions (IEA 2008).

Activities in all urban centres, however, are not contributing to these emissions equally. Cities of the developed world are considered to be the largest generators of CO_2 emissions. Cities in the European Union tend to use less energy per capita than US and Australasian cities, and therefore have lower CO_2 emissions (IEA 2008) and developed economies tend to have higher emissions than developing economies (Fig. 4.1.4). At the same time, some

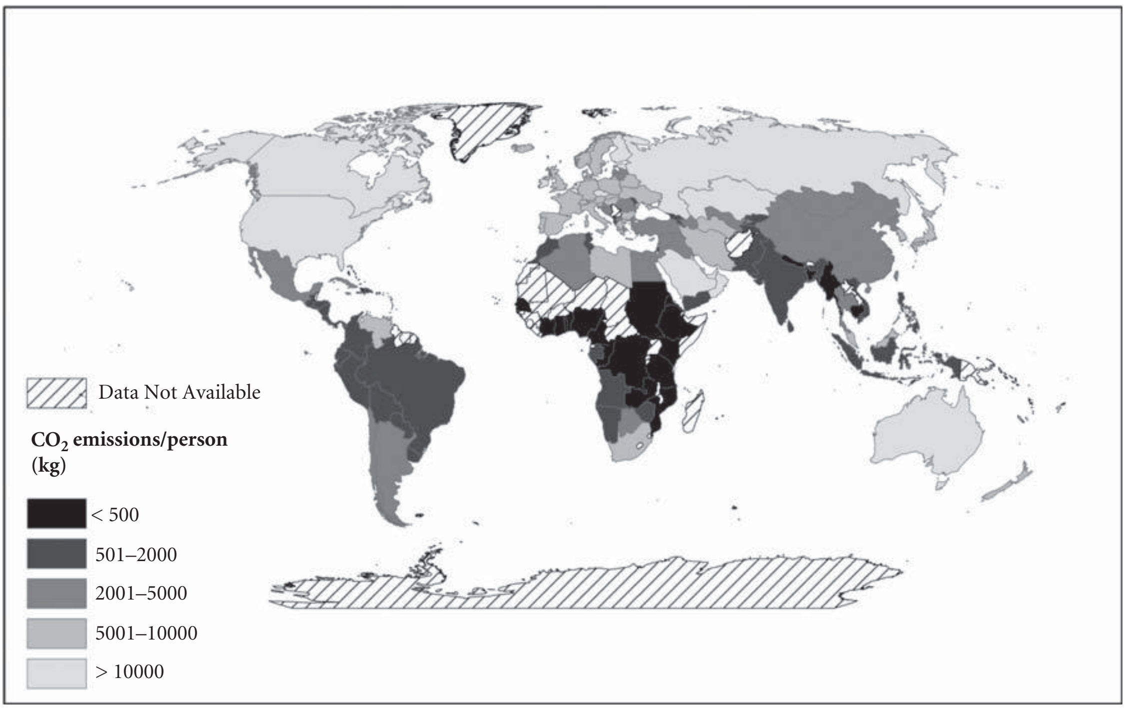

Figure 4.1.4 Global patterns of carbon emissions per capita. Data from IEA, 2008

studies suggest rapidly industrializing cities are also high contributors. For example, Dhakal and Imura (2004) report that in 1998, while Tokyo's emission levels were 4.84 tonnes of CO_2 per capita, Beijing's emissions were 6.9 tonnes per capita, and Shanghai's reached 8.12 tonnes per capita.

While cities are typically larger total contributors of CO_2 emissions than rural areas, residents in cities are sometimes more efficient and have lower per capita emissions. This is particularly true in cities in the developed world, where higher population densities but similar per capita income relative to rural areas allows for less use of cars, more extensive urban public transport systems, and more district heating. A study of the largest 100 metropolitan areas in the USA in 2005 estimates that despite housing 66 per cent of the nation's population and 75 per cent of its economic activity, these locations emitted only 56 per cent of the national carbon emissions from highway transportation and residential buildings (Brown, Southworth, & Sarzynski 2008). The authors conclude that metropolitan carbon per capita footprints are lower than the national average.

This relationship, however, is reversed in developing countries, where residents in cities use significantly more energy as a result of higher incomes and better availability of energy services compared with rural areas. For example, the average resident of cities in China demands over 80 per cent more energy than the per capita national average (IEA 2008).

An important question is whether population concentration contributes more CO_2 emissions than the alternative. This will necessarily remain an open question, as we currently do not have an alternative with which to compare. At the same time, we can focus on what current trends portend. On the one hand, dense settlements provide many services that reduce the carbon footprint of citizens, but this can only be accomplished if the necessary infrastructure is in place. In North America, Europe, and Japan, living in cities provides benefits to regulating services by reducing per capita carbon footprints. On the other hand, in the developing world where most of the world's urban population growth will occur, residents consume more energy than those in rural areas and dense urban settlement patterns are therefore associated with higher carbon emissions. As the world urbanizes, we can expect that energy consumption, and therefore CO_2 emissions will increase dramatically in developing regions. Creating low-emission, green cities on a budget that developing countries can afford will be a major design and engineering challenge in the future.

4.1.4.3 Biodiversity

Important, but often overlooked, are the ecosystem structures and processes that enable other services to exist but which are arguably not of use themselves. For instance, soil forms very slowly, from the breakdown of minerals by biotic and abiotic processes, yet without soil most crops could not be grown. Similarly, over time soil profiles tend to become stabilized by roots and other organic matter, reducing the likelihood of mass erosion. Despite their importance, these processes are often ignored because they do not directly affect most people's well-being. Moreover, they are extremely difficult to value in economic terms because no viable markets for them exist, and because most people find it difficult to contingently value them.

Biodiversity is often an important component for ecosystem service provision. For example, nitrogen fixation is very important for biological productivity, and only a few plants, such as legumes, can perform this service. Often, scientists do not know which components of biodiversity are necessary for ecosystem service provision, but they are certain there is some link. Paul Ehrlich illustrated this with his rivet metaphor: if one were flying in a plane and looked out a window and saw rivets begin to fall one-by-one off the wing, one could be certain that at some point the wing would detach from the plane, although it would be difficult for an engineer to say which particular rivet would be crucial (Ehrlich & Walker 1998).

Here we focus on the species-level component of biodiversity, discussing species richness and endemism. Both quantities are very unevenly distrib-

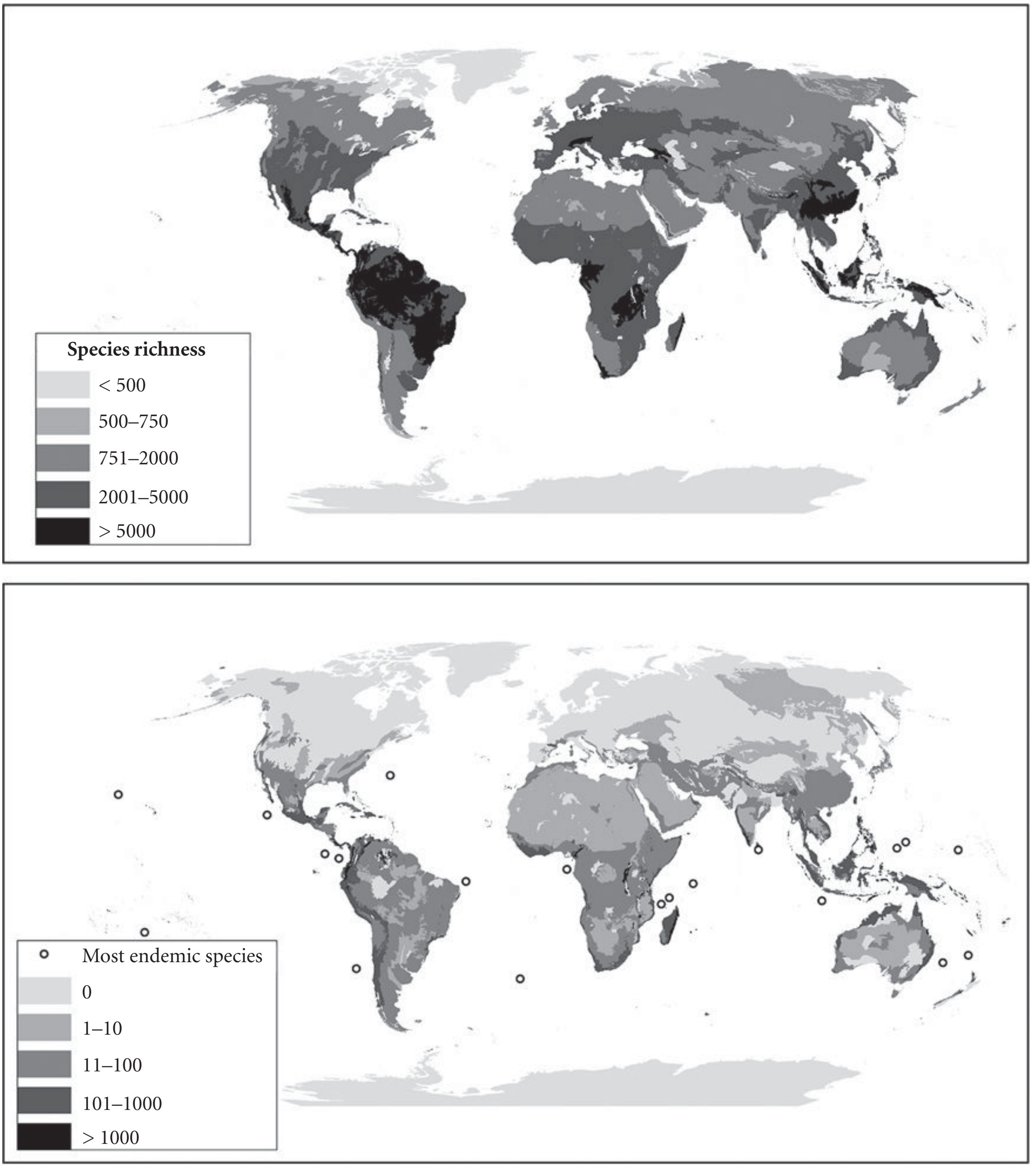

Figure 4.1.5 Global patterns of species richness (top) and endemism (bottom). Species richness is the total number of plant and vertebrate species in WWF ecoregions. Endemism is expressed as the number of endemic species per million km^2 of area in the ecoregion. For reference, the 20 ecoregions with the highest endemism are marked with circles. Data from WWF Wildfinder (2006)

uted over the Earth's surface (Fig. 4.1.5), and a whole academic discipline, biogeography, is dedicated to mapping them. Species richness is generally higher in higher productivity sites like tropical rain forests and lower in lower productivity sites like arctic tundra. The causes of this trend have been much discussed and disputed (Willig, Kaufman, & Stevens 2003). The number of species unique to one location, its endemism, is of more conservation concern, and is distributed very differently. Islands tend to have an unusually high degree of endemism because the geographical isolation encourages speciation. For both terrestrial and marine organisms, coastal areas are also places with a high degree of endemism because of the high habitat diversity (Dirzo & Raven 2003).

Cities tend to be similarly concentrated along coastlines and some islands (Fig. 4.1.1), as well as major river systems. The ecology literature has explained this pattern by examining the correlation between human population density and productivity (Luck 2007), while the urban planning literature has focused on the importance of freshwater and marine trade routes for city formation. Regardless of its cause, the spatial correlation between urban growth and endemism means urban growth poses challenges for biodiversity disproportionate to its area. Within land converted to urban land-uses, most endemic species and much overall biodiversity is lost. However, a few generalist species will remain, joined by a suite of invasive species that tend to follow human settlement (e.g. the Norway rat). The net result of the expansion of urban land cover is a reduction in total, global biodiversity, with the depauperate flora and fauna of cities sharing a common, 'cosmopolitan', set of species (McKinney 2006).

One useful measure of land-use is the built-up area created for each new inhabitant of a city. A large study by Angel *et al.* (2005) calculated this measure for a globally stratified sample of 90 cities. Per capita land-use varies by income, with low-income countries (< \$3,000 GDP Per Capita PPP) taking 85 m^2/person, moderate-income countries (\$3000–\$5,200 GDP Per Capita PPP) taking 115 m^2/person, high-income countries (\$5,200–\$17,000 GDP Per Capita PPP) taking 170 m^2/person, and very high-income countries (> \$17,000 GDP Per Capita PPP) taking 350 m^2/person. The number of cars per 1,000 people is also strongly correlated with income, and the presence of cars presumably plays some role in encouraging more sprawly patterns of development in rich cities (Kenworthy & Laube 1999). Over time, as urban areas in the developing world grow richer and cars grow more common, new settlement density in the developing world will continue to fall. However, it is worth noting that the density of settlement in the developed world varies widely, from 185 m^2/person in London to 621 m^2/person in Minneapolis, and so it is not clear exactly how sprawly the new cities in developing countries will be.

Rural land-use is generally less intense than urban land-use, and a greater proportion of species remain after human alteration of the natural ecosystem. There is, of course, substantial variation in the biodiversity impacts of different types of rural landuse, with agricultural systems losing more of their native biodiversity than pastureland or timberland. It has been occasionally argued that global urbanization may decrease loss of habitat in rural areas (Wright & Muller-Landau 2006). However, two factors limit the spatial scope of this 'depopulation'. First, many countries with fast rates of urban population growth also will continue to have rural population growth for the foreseeable future, albeit less than might have happened if the urban migration was hypothetically not present. Presumably habitat loss will continue in rural areas under these conditions. Second, rural areas are still needed to feed everyone, even if a greater proportion of humanity is in cities, so agricultural expansion may continue even if rural populations decline. Nevertheless, it is possible that certain rural locations that are marginal for farming and have declining populations may return to a semi-natural land cover. In this case, some native biodiversity may return, although experience with ecological restoration suggests secondary natural habitat usually never regains a full complement of native biodiversity (Ehrenfeld 2000b).

On balance, urbanization and the associated economic process of development appears to increase the rate of biodiversity loss. There is a relatively small impact on global biodiversity from urban land area expansion itself (McDonald, Kareiva, & Forman 2008). However, in developing countries migrants to urban centres typically increase their income and their consumption, increasing the ecological footprint of cities. Cities, directly or indirectly, control the land-use of most rural areas, so an expansion of cities' ecological footprint will potentially affect a much larger amount of biodiversity.

4.1.5 Conclusions

Do cities and the urbanization process provide benefits or costs to environmental quality? We have argued that it is difficult to provide a straightforward answer to this question because urbanization is so completely bound up with other development processes that it is difficult to separate out the impacts of this process from other trends such as increasing wealth and consumption. Often critiques point to activities in cities, suggesting that they are disproportionately burdensome without comparing the impacts of the same type and level of activity in non-urban areas. We have yet to be able to calculate the impacts of this type of spatial organization, as there are no non-urban developed societies to which it can be compared.

So, our answer to the question of the impacts of urbanization is far from clear. There is evidence, however, that the influence of dense settlement patterns depends upon the ecosystem service, scale of impact, and the development level of a city. For example, urban settlement is typically associated with the degradation of local water resources and the simultaneous extraction of this provision service from larger regions or from underground sources. Moreover, as water is used for agriculture that increasingly feeds urban populations, there are both direct and indirect connections of urban demand and over-use. While there are opportunities for water conservation for cities (as compared to rural users), these advantages often go unclaimed. Yet, if we consider an alternative provisioning and distribution system at the regional level, the benefits of urbanization for water service conservation become clearer. That is, if people were spread over the landscape there is potential impact on water quality as well as loss and inefficiencies through distribution systems to provide the service to those demanding it. Hence, we would suggest that at a local level, cities may negatively impact water services, but at a regional or national level, they may provide benefits.

The same pattern may be true for regulating services and biodiversity, with caveats. Certainly, fossil fuel consumption in and around dense settlements exacerbates climate change. At the national and perhaps global level, however, organization in cities versus sprawled settlements provides benefits through energy efficiency and also may facilitate afforestation and reforestation. Together these processes can lower emissions levels and increase sequestration potential. On the other hand, however, these trends are complicated, as the lower energy efficiency levels of rural areas compared to those of urban areas only emerge with higher levels of development and advanced urbanization.

Finally, it seems that the location of cities has negatively impacted biodiversity at both the local level (where urbanization accompanies homogenization and general loss of species richness) and the regional level. At the regional level, increasing consumption within cities also increases pressure on hinterland biodiversity. Yet at the global level there is relatively low impact of urban land-use on biodiversity. There is also an argument that concentrating population in cities provides more ecological space and multiple use conservation/recreational opportunities for delicate ecosystems.

It would seem then that our task for the future is complex and multi-fold. We suggest that there needs to be a concerted effort to indentify and analyse the specific costs and benefits of dense settlement organization. The goals are to seek out opportunities for both ecological and social solutions. These solutions may emerge through design, such as providing for natural biodiversity habitats in cities, increasing the diversity of types of open spaces in and around cities, recycling water and further increasing energy efficiency of buildings, improving transit infrastructure, and the like. There are examples where some of these programmes are already in place (Malmo, Sweden; Curitiba, Brazil; Singapore; Freiberg, Germany). In pursuing these goals society must also reduce poverty and address unequal access to ecosystem services. We recognize that in the quest to create more benign urban centres, there will be trade-offs to be made, but opportunity costs should not be at the expense of livability for all peoples.

CHAPTER 4.2

Social-Ecological Transformations in Urban Landscapes—A Historical Perspective

Charles L. Redman

4.2.1 Introduction

The growth of urban centres in virtually every part of the globe has engendered a profound alteration in all forms of social-ecological relationships. Although the amount of land covered by cities remains small, and the proportion of the global population living in cities historically has been low, the direct demands of urban dwellers and the reorganization of productive systems and impacts of urban decision-makers have transformed social-ecological systems throughout history (Boone & Modarres 2006). It is possible to trace the development of these patterns in historic cities in various parts of the world. Processes such as population growth, enhanced food production systems, and specialization of labour and trade underwrote the emergence of cities (Redman 1999). At the same time the increasing size and ubiquity of cities set in motion important feedback processes whereby social-ecological systems were transformed in ways to further support and facilitate the continued growth of urbanism and the associated social hierarchies. Along with this increasing complexity came a restructuring of social relationships and institutions related to governance. Hierarchical governance, economic interdependence, and class-structured society became, and continue to be, hallmarks of urban society. These very processes and the feedbacks that promoted the growth of cities have at the same time introduced into these systems vulnerabilities that threaten their sustainability.

All of these processes emerged as a result of decisions made by individuals and groups of individuals organized into informal and formal institutions, including governments at all scales from local to international and of various political persuasions. If all decisions could be characterized as a series of discrete trade-offs with predictable outcomes, then the historical trajectory and contemporary condition of social-ecological systems would be straightforward to explain. However, many if not most, of the important decisions that are made in society are made under conditions where the optimal trade-off is not readily identifiable due to differing valuations among potential participants, changing contexts in the future, and an inherent uncertainty of outcomes even if conditions could be predicted. Hence, examining the origin and history of cities and networks of cities has the potential to reveal how decisions actually have been made in many varied contexts and the resulting short- and long-term implications of those decisions. The past can be a rich source of insight in evaluating the options we have for the future.

4.2.2 Why look to the past?

Some question the relevance of episodes from the past for dealing with problems of the present and future. While it is not being claimed here that the past offers simple, detailed, directly applicable solutions to the challenges of the present, it is strongly argued that the past does offer essential insights into the nature of socio-ecological interactions under a huge number of diverse conditions and the implications of those decisions made (Costanza *et al*. 2007). Two additional points strengthen this case. First, for at least the last 20,000 to 40,000 years, people have been biologically and mentally indistinguishable from those of us alive today; and second, people in the past faced real environmental crises. There is little question that during the past 10,000 years we have modified, with increasing speed, our cultural trappings and social organizations, creating in the ensuing years environmental problems of increasing severity that are more global now than they were in the past. We must not, however, underestimate the impact environmental crises had on people in antiquity (Butzer 1982; Crumley 1994; Deneven 1992; Redman 1999; Diamond 2005). Having environmental degradation force the abandonment of a homeland in which generations of one's ancestors lived and died is a disaster that few of us face today.

There are many scientists that argue that we have entered an era that is unique due to the unprecedented number of people on the Earth, the intensity and extent of human impacts on the Earth, and the problems introduced by modern technologies (see McDonald and Marcotullio, Chapter 4.1). Pulitzer Prize winning ecologist and geographer, Jared Diamond, acknowledging this position, has recently identified what he considers to be the twelve fundamental social-environmental threats facing the modern world (2005). Eight of those twelve he believes were fully operational in the past and historic and prehistoric case studies may reveal insights that would contribute to solutions for the present and future. In terms of human decision-making and the operation of socio-natural ecosystems, the past has a great deal to tell us about how people confront threats to the sustainability of relationships between and among social and ecological systems.

In addition to being a general source of insight, the past does offer some unique insights unavailable from contemporary studies. First, contemporary studies usually have to content themselves with investigating a truncated historical cycle, that is, not having completed one cycle, while archaeological and other historical case studies can provide not only completed cycles, but often multiple completed cycles. This allows greater understanding of the dynamics of phases of a single cycle, of linked cycles, and of how they might change as systems reorganize. It also permits more in-depth monitoring of the slow processes and low frequency events that appear to be the key to ultimate system resilience (Scheffer *et al*. 2001; Gunderson & Folke 2003). Although ecologists know that ecosystem structure and function may take decades or centuries to fully respond to disturbance, ecological studies almost exclusively examine ecosystem dynamics over intervals of a few days to a few years. Rare decades-to-century scale studies suggest that some human impacts are enduring, yet only a few integrative ecological studies of human land-use cover timescales even as long as a century (Roberts 1998; Foster 2000; Mann 2002; Heckenberger *et al*. 2003; Vitousek *et al*. 2004). Based on the growing availability of long-term databases, an increasing number of ecologists strongly assert that the current condition of many landscapes and the dynamics that govern them could not be understood without close attention to the effects of historic land-use (Foster *et al*. 2003; Redman & Foster 2008).

Second, the 'deep time' perspective allows us to understand the ultimate, as well as the proximate, causes of the collapse of social and ecological systems. Ecologists have historically viewed land-use as a key human impact without addressing the social dynamics that lead humans to alter the landscape in diverse ways. Collaborations with social scientists who seek to address these dynamics will allow ecologists to understand the ultimate, in addition to the proximate, drivers of human–environment interactions (McGlade 1999; Collins *et al*. 2000; Diamond 2005). In particular, the long-term history of human–environment interactions contained in the archaeological record reveals that many human responses and strategies, while apparently beneficial in increasing production in the short term (even over a few generations), nonetheless led to a serious erosion of resilience in the long term, resulting in the collapse of both environmental and social systems (McGovern *et al*. 1988; van Andel *et al*. 1990; Kirch *et al*. 1992; Redman 1999; Diamond 2005). It is only with the long time perspective that we can identify which of many seemingly beneficial near-term actions truly contribute to long-term resilience, and identify the ways in which some seemingly rational choices lead, in the end, to undesirable outcomes. The converse of this is that some social adaptations or cultural traditions may appear inefficient or 'illogical' when viewed in the short term, end up reducing risk and increasing resilience in the long term (Butzer 1996).

Third, the fields of archaeology and history, when supplemented with anthropological and sociological perspectives, allow a rich understanding of the linked dynamics of human behaviour, social dynamics, and ecological systems across broad scales of organization—from individual households to hamlets, villages, cities, and civilizations. Few other social sciences encompass such a broad organizational spectrum, preferring instead to concentrate more narrowly on 'bottom up' (e.g. household, village) or 'top down' (e.g. nation, state) levels of organization. If the operation of a system is predicated on linked dynamics across scales—particularly the interaction of 'fast' and 'slow' variables or the 'mismatch' of scales at which social and ecological variables interact—then examination of these linkages from both social and ecological perspectives will be crucial (Folke *et al*. 2002). Archaeological perspectives can provide critical bridges to fill some of the gaps left by present-day or near-historic studies focused on a narrower spectrum of human organizational scales.

Finally, the archaeological record allows us to identify those emergent features that appear to be inevitable—or at least highly probable—in societies increasing in complexity, including social stratification, compartmentalization of information, and, at certain scales, ecological simplification. One of the first challenges of conceptualizing sustainable futures will be to distinguish those features of social systems, and human interactions with the environment, that can be altered to achieve more desirable social and ecological outcomes, and those that are so much a product of history, human development, and biological, social, and cultural evolution, that we must accept their undesirable constraints in fashioning our visions of the future (National Research Council 1999). An archaeological perspective can contribute to meeting this challenge by documenting the full array of interactions that have and have not existed, at least up to this point in history.

4.2.3 Drivers and consequences of urbanization through history

One of the most significant milestones in the development of human societies was the growth of the first cities of Sumer on the Mesopotamian plain of southern Iraq. This process involved much more than just an increase in the size of settlements; it included fundamental changes in the way people interacted, in their relationship to the environment, and in the way they designed their communities and the world around them. Writing, legal codes, the wheel, the plow, metallurgy, mathematics, and many engineering principles—all commonplace in our modern world—were first developed in the cities of Sumer (the name they called themselves). Despite the vast scope of these technical developments, the most significant changes were those of social organizations. As some groups of people acquired access to resources that allowed increased production—for example, better farmland, more irrigation water, or rare goods traded from other regions—social class became one of the main principles structuring interactions within communities. These communities became organized according to the emerging hierarchical political and administrative systems, which often used written legal codes and centralized wealth and relied on coercive force. These processes occurred at different times in each part of the world, but there is good archaeological evidence for what we are willing to call cities in at least Mesopotamia by 3000 BC, soon thereafter in many other parts of the Old World, and by the early Christian era in specific regions of North and South America.

The emergence of urban society introduced a whole new set of socio-ecological interactions. One set of interactions derives from the fact that there were just more people in the world, requiring greater food production and extraction of more materials to build their cities, processes that in themselves allowed further growth of population. Advancing agrarian techniques and urban life allowed people to invest their labour in permanent facilities and to accumulate more goods. The creation and concentration of goods and the productive capacity to create more became the hallmark of urban society. The increased demands put on local environments by growing urban populations were partly mitigated by the greater labour invested to transform their landscapes to sustain a high level of production and extraction. Many efforts employed to increase productivity took the form of management and transformation of the ecosystem through redirecting the hydraulic system through irrigation or modifying the landscape structure through terracing. Other efforts involved technological innovations such as breeding for crop/animal improvements, the plow and eventually tractor to expand the scale of positive soil manipulation, and increasingly complex tools from sickles and silos eventually to combines and food processing facilities that improved the harvesting and storing of an ever increasing quantity of food and other produce.

At least at the outset, each of these basic strategies did act to increase the amount of food and other goods available to a society, facilitating the growth of its population and an expansion in the energy it could devote to civilizational institutions. However, these same strategies demanded increases in labour input and often an over-utilization of the landscape, diminishing its long-term productivity and at the same time potentially stimulating social unrest. Around the world, urban societies that are based on food production systems that rely on increasing labour input and potential over-utilization of the landscape, have developed institutions and supporting ideologies that act to keep the society integrated and operating in a reasonable manner. However, successful these strategies are at most times and in most places, they do introduce several weaknesses into the system that make it more vulnerable to crises. Slowly diminishing soil fertility, lack of alternative food sources, decision-makers who are removed from the scene of production, and reliance on outside groups for essential goods are all conditions that weaken the resilience of a society when it faces trouble, and these continue to threaten our contemporary world (Redman 1999; Diamond 2005; Fisher *et al*. 2009).

A second major transformation accompanying urbanization is the change in the mobility patterns of individuals and community groups. In pre-modern times individuals were very tightly tied to their social group and community, but many of these communities were not expected to remain in the same location over time, and in fact many societies built in the expectation of movement whether it be generational or seasonal. These movements were responses to the productive potential of the landscape and it was easily recognized that under many circumstances it would be easier to move the people to the resources than the resources to the people. Mobility could also be a response to a group size getting too large or a regional population too dense. This mobility characterized the early millennia of the human career. However, with the advent of agricultural village life, substantial labour was invested in ecosystem management strategies in order to maintain and enhance the yield of a particular landscape, providing a strong incentive not to abandon the location. The permanence of communities themselves was further encouraged by the investment in stationary facilities, from housing to manufacturing and food storage facilities. As some villages grew into towns and some of those into cities, they were characterized by increasing investment in immoveable facilities, ultimately city walls and public buildings, that would further discourage mobility.

This transformation in the built environment was accompanied by social and political changes that also appeared to encourage sedentarization. With sedentarization, people have developed a strong psychological and cultural attachment to where they live—a sense of place. This attachment to a particular landscape is often reinforced by cultural traditions and myths of origin. However, it is likely that these ideologies of attachment to a territory grew in concert with changes in the broader political organization of society. Early in the development of urban society, kinship, that had been the primary means of organizing human relationships, was superseded first by religiously defined authority and then by a secular, often territorially defined, organizational structure, the state. Political authority quickly became paramount in defining the social order, the duties of individuals, and the economic system. Kinship has remained important to this day in many relationships, and religious authority in some societies has retained, or is currently attempting to reassert, its primacy, but in general, kin, religion, and even ethnicity have become subdivisions within the more encompassing secular/territorial state and retain their hegemony over only restricted sets of activities. This has led political leaders, in an attempt to increase membership in their state (or city state), to enact measures to attract and retain a large population of potential labour (and possible military conscripts) within their territorial boundaries. Identity has become associated with the state, as have rights, privileges, and obligations. At different points in history, such as with the Assyrians, Romans, and ultimately European colonial expansion, the states themselves have moved major segments of their population around to ensure political control or enhance economic success. However, the more pervasive pattern has been to encourage and enforce stability of residence. The mobility of particular social groups between urban and rural areas, and even among different cities, is as old as urbanism and has served a variety of purposes. However, with the increasing necessity of states identifying their territory with precision, it was inconvenient to have ambiguous borders and large groups of people moving up and back between what the rulers considered different states. In mediaeval times, one was closely tied to the feudal lands associated with your city and ruler, while in China even today one belongs to a particular territory or city and loses significant rights by leaving that location. During the nineteenth and twentieth centuries in particular, many countries have tried to settle the still nomadic elements of their societies and to fix national borders.

Toward the end of the twentieth century a new process challenged the primacy of impermeable, fixed state borders—globalization. The movement of raw materials, manufactured goods, food, and financial instruments has increased in scope to the point where many consider the world a single productive unit and market. Goods are being moved in unprecedented quantities from where they can be produced cheaply to where people have the financial resources to purchase them. At first glance this appears to contradict the initial observation cited above, that it is easier to move people to resources than resources to people. Although this appears to be true at one level, a countervailing pattern of people moving to the productive resources is picking up momentum with globalization and may reverse several centuries of sedentarization. In a totally globalized world the formerly industrialized world (North American and Western Europe) would produce few goods, purchasing them from countries with the least expensive labour, energy, and raw materials. To a great extent this is true, but there are other forces at work. The industrialized nations of the world want to hold on to the benefits of being producers for both economic and political reasons, and that means they need to add inexpensive labour to their countries. This has meant a great flow of labour, and often families as well, from Africa, Latin America, and parts of Asia to North America, Western Europe, and the Gulf States. This process may allow former industrialized countries to remain somewhat competitive in the global markets, but given the 'territorial' basis of citizenship it creates a conundrum for how to treat the labour immigrants. We are in the middle of this complex process and it is difficult to predict its future trajectory, but it is profoundly affecting the social-ecological interactions in both donor and recipient states and, at a grander scale, the political-demographic configuration of the world.

The third major transformation that was accelerated by the growth of cities is for problem-solving to be achieved through the formation of institutions that often involved more complex forms of information flow and hierarchical ordering of human interactions. A key attribute of cities from the earliest times is that they emerged as foci of diverse activities. They allowed people to specialize in their productive activities and then to exchange goods and services as needed. As the numbers of people involved and the diversity of tasks increased it became obvious that production and exchange would be best handled by specialized institutions, whether they relied on a formal administrative apparatus, as characterized the Sumerian redistributive economy, or a market exchange system that depended on an institutionalized order but was more bottom-up, as in late pre-Hispanic Mexico. With the emergence of these institutions there also arose an 'elite' that benefitted disproportionately from the productivity of the more numerous 'commoners'. Other elements of the administrative hierarchy could be involved with maintaining order and ensuring security through a monopoly on the use of coercive force and the creation and promulgation of various ideologies that help explain the way people were to behave and the importance of the hierarchical order. Overall, there was a flow of goods and information from the numerous citizens through a series of administrative levels to the top administrators of the city or state. They in turn, made decisions, and allocated resources and positions back down through the hierarchy. Although this has relatively effectively served the needs of large social groups around the world for millennia, it has also introduced potential vulnerabilities into the urban system that periodically bring the system down. The first is the very cost of the administrative hierarchy and particularly the elite at the top. Introducing complexity into a system may have some functional advantages, but it also must be recognized that there are serious costs as well. The second weakness is that as information and goods flow up and down the administrative hierarchy there is a 'loss' in the precision of the information and a drain of the goods traversing the hierarchy. The third weakness that is introduced is that people at different positions in the hierarchy may in fact have different views on the world and its proper operation. This contextual aspect to decision-making is related to both perception and values and is the most insidious of the weaknesses of increasing social complexity. When societies solve problems confronting them or take advantage of opportunities afforded them, again and again there are winners and there are losers. Who is in each camp seems to be patterned by which party already holds the wealth, access to information, and power within the society.

4.2.4 Towards an understanding of urban resilience

The continued resilience of cities and the broader urban society relies on an improved understanding and favourable resolution of several important issues in socio-ecological relationships. By bringing together such a large number and diversity of people into cities it has become clear that most problems do not have simple solutions. Most solutions will involve trade-offs and ideally the concerned parties can agree on what an optimal set of trade-offs would be. Building a freeway reduces congestion for drivers who pass through a neighbourhood, but how does one measure that against the cost to the people whose home must be destroyed to build the highway, let alone the more general shared impacts like increased air pollution or greenhouse gases? Hence, the resolution of most sustainability challenges requires explicit attention to the normative values of the parties involved. For those with a natural or physical sciences background, problems are solved by 'objectively' applying the scientific method to empirical evidence allowing one to reach the 'correct' solution and, hence, they are not accustomed to problems having multiple 'correct' solutions depending on how one figures in context and competing value systems. Nevertheless, this is the reality of socio-ecological relationships and sustainability problems. The issue should not be whether to acknowledge the importance of incorporating normative values—they must be—but how to compare or 'rank' alternate values. This is a fundamental problem of sustainability. It is possible to look at the past and evaluate which values were adhered to or abandoned by societies with long-term success, versus decisions made by societies that collapsed (Diamond 2005). The difficulty comes when trying to apply this framework to the future where the context and the values to be held at a future time are not known with clarity.

If one acknowledges the necessity for taking account of the values of participants then it becomes extremely important to have a methodology for who is to be included in decision-making. In an ideal situation it might be best to have 'all' parties represented, but the definition of 'all' and the power relationships among them would still mean that the ideal is not likely to be attained. The other problem with relying on a totally inclusive set of participants is that the group could become gridlocked by too many viewpoints and too many competing values to resolve. Hence, to move forward in an analysis to support a decision, one must develop an explicit framework for evaluating the range and weighting of participants in the activity. This opens the possibility for biases being introduced by selection of participants, but this is virtually always the case though done implicitly and hence without recognition of the bias.

A related issue in reaching decisions about managing socio-ecological interactions is, once a trade-off is agreed upon, determining which individuals and groups are to bear the cost of the solution reached (i.e. the losers). This is often only dealt with in a superficial manner (e.g. the taxpayer) without in-depth analysis of less obvious costs and cascading impacts of the proposed solution. In general it is the politically weak, the economically disadvantaged, and future generations that appear to most often bear the cost of sustainability solutions. How to reverse this process does not so much require scientific insight as it requires political agility and willpower.

One further issue to be addressed at this point involves how one deals with the ubiquitous trade-off between having socio-ecological processes operate efficiently versus building resilience into those relationships. Enhancing resilience of a system often means encouraging flexibility and adaptive capacity in the forms of redundancy, inclusiveness, monitoring, and preparedness for multiple futures. All of these efforts have associated short-term costs, yet we live in a world that is striving for efficiency and 'lean' organizations. Just-in-time production/distribution systems, small government, and even cost-effective non-profits are driving forces of our competitive, globalizing world. A strong argument for 'investing' additional resources in building resilience is that in the 'long-term' these costs are well justified. However, the issue remains that to succeed one must get through the short term before the long term is relevant. We need to build approaches and systems that instil long-term logic and investment into short-term decisions.

CHAPTER 4.3

The Urban Landscape as a Social-Ecological System for Governance of Ecosystem Services

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4.3.1 Introduction

The constantly evolving urban landscape is a complex mosaic of human modifications, metabolic flows, networks, and built structures (e.g. Alberti *et al*. 2003). As emphasized by, for example, McDonald and Marcotullio (Chapter 4.1) and in many other chapters, the understanding of how urban ecosystems work, how they change, and what limits their performance, can add to the understanding of ecosystem change and governance in general in an ever more human-dominated world (Pickett *et al*. 1997; Elmqvist *et al*. 2008). Today, cities are facing enormous challenges, such as climate change and transformation to a future beyond fossil fuels. Ecosystems may play an important role in facilitating this transformation. Ecosystems provide flexibility in urban landscapes and help build adaptive capacity to cope with problems such as increased risk of heat waves and flooding. Urban ecosystems generate ecosystem services, many of which are crucial for the well-being of urban citizens (e.g. Elmqvist *et al*. 2008). In this context we argue for the necessity to view urban landscapes as truly interconnected social, cultural, and ecological systems and to recognize, map, and sustain ecosystem services within urban regions.

There have been various approaches during the last two decades at the global scale, recognizing that ecosystem management and conservation strategies must be connected to human well-being. These include the Ecosystem Approach of the Convention on Biological Diversity (CBD 2000–08), a strategy for the integrated management of land, water, and living resources that promotes conservation and sustainable use in an equitable way. Further, the UNESCO Seville Strategy (UNESCO 1996) explicitly recognizes connections between cultural and biological diversity and their role in sustainable development. The ecosystem approach, although acknowledging that the cultural diversity of humans comprises an integral component of many ecosystems, was originally meant to enhance communities' ownership of conservation strategies in areas defined as 'natural'. Hence, the ecosystem approach, while integrating the sectors/disciplines needed to manage the landscapes, was not intended to be applied to complex and fast transforming urban systems. Conceived as it was before the threats associated with climate change had reached public consciousness; it did not make the link explicit between ecosystem functions and human security and well-being. Today, with increased incidence of natural disasters and the looming threats associated with climate change, we have a much deeper sense of the immediate and inextricable connection between human security, well-being, and functioning ecosystems (MA 2005a). However, our current approaches to governance, resource allocation, and public information have yet to adapt in response to these threats.

This chapter focuses on the challenges of combining our theoretical understanding of ecosystems with perspectives on alternative governance systems based on the ecosystem approach in human-dominated landscapes. On a human-dominated planet, environmental solutions increasingly have to rely on innovative applications of science and technology rather than on strict nature preservation methods. We argue that the urban landscape is the right place to begin developing new forms of governance based on integrated and cross-sectoral responses to global challenges. Urban regions have the advantage of having concentrated scientific, technological, and financial resources needed for innovations (Redman, Chapter 4.2) and a potential capacity to mobilize resources, people, and governments at all levels to build social-ecological resilience through adaptive and strategic planning.

4.3.2 Urban ecosystems and ecosystem services

A city is, among other things, a physical and social mechanism for acquiring and delivering ecosystem services to a dense human population (Lee 2006). The Millennium Ecosystem Assessment (MA) (2005a) found that 60 per cent of ecosystem services assessed globally are either degraded or being used unsustainably. Seventy per cent of the regulating and cultural services evaluated in the assessment are in decline. However, in MA, urban areas were largely left out even though urban ecosystems contribute to several environmental regulation services and cultural services of importance for human well-being (Bolund & Hunhammar 1999).

In general, the urban landscapes present novel ecological conditions, such as rapid rate of change, chronic disturbances, and complex interactions between patterns and processes. Firstly, compared with ecosystems in rural areas, urban systems are highly patchy and the spatial patch structure is characterized by a high degree of isolation. Second, disturbances such as fire and flooding are suppressed in urban areas, and human-induced disturbances are more prevalent as well as intense human management of urban habitats. Third, because of the 'heat-island' effect, that is, higher mean temperatures in cities than in the surroundings, cities in temperate climates have significantly longer vegetation growth periods. Fourth, ecological successions are altered, suppressed, or truncated in urban green areas, and the diversity and structure of communities of plants and animals may show fundamental differences from those of non-urban areas. Furthermore, cities are the most important points of introduction of exotic species. Although cities may sometimes be species rich, having higher species diversity than surrounding natural habitats, this is often due to a high influx of non-native species and formation of new communities of plants and animals. To what extent exotic species contribute to reduced or enhanced flow of ecosystem services is virtually unknown for any urban area, but since introduced species make up a large proportion of the urban biota, it is important to know not only to what extent introduced species are detrimental, but also to what degree some of the introduced species may enhance local diversity and maintain important functional roles.

The potential for generation of ecosystem services in urban areas is often substantial (e.g. Pickett *et al*. 2008), albeit not very often realized. For example, urban parks and vegetation reduce the urban heat island effect and there is an important potential for lowering urban temperatures when the building envelope is covered with vegetation, such as green roofs and green walls, with the largest effect in a hot and dry climate (Alexandri & Jones 2008). In relation to overall climate change mitigation, urban ecosystems may assimilate large quantities of carbon, for example, in Stockholm County, ecosystems assimilate about 41 per cent of the CO_2 generated by traffic and about 17 per cent of total anthropogenic CO_2 (Jansson & Nohrstedt 2001), and residential trees in the continental United States could sequester 20 to 40 teragrams of carbon per year (Jenkins & Riemann 2003). Vegetation may reduce noise levels, and dense shrubs (at least 5 m) wide can reduce noise levels by 2 dB(A), while a 50-metre wide plantation can lower noise levels by 3–6 dB(A) (Bolund & Hunhammar 1999). Evergreen trees are preferred because they contribute to noise reduction year round (Ozer *et al*. 2008).

There are also direct health benefits of green areas, vegetation and trees, for example, in a study from New York, presence of street trees was associated with a significantly lower prevalence of early childhood asthma (Lovasi *et al*. 2008). Green area accessibility has also been strongly linked to reduced obesity (e.g. Ellaway *et al*. 2007). In a review by Bird (2007), links were noted between access to greenspaces and a large number of health indicators, such as coping with anxiety and stress, treatment for children with poor self-discipline, hyperactivity and Attention Deficit Hyperactivity Disorder (ADHD), benefitting elderly care and treatment for dementia, concentration ability in children and office workers, healthy cognitive development of children, strategies to reduce crime and aggression, strengthened communities, and increased sense of well-being and mental health (see also Tzoulas and Greening, Chapter 5.2). The distribution and accessibility of greenspace for different socio-economic groups, however, often reveals large inequities in cities (e.g. Pickett *et al*. 2008), contributing to inequity in health among socio-economic groups, although confounding effects are not always possible to separate (Bird 2007).

To what extent biodiversity and variation in species composition plays a role in the generation of environmental quality services is still poorly investigated (Elmqvist *et al*. 2008). For air quality, filtering capacity increases with leaf area, and is thus higher for trees than for bushes or grassland (Givoni 1991). Coniferous trees have a larger filtering capacity than trees with deciduous leaves (Givoni 1991). In the urban core there are probably fewer species and often very different species involved in the generation of ecosystem services than in more rural areas. Interestingly, the number of plant species in urban areas often correlates with human population size, and diversity may correlate with measures of economic wealth as shown, for example, in Phoenix, USA (Kinzig *et al*. 2005).

Cultural ecosystem services refer to the aesthetic, spiritual, psychological, and other benefits that humans obtain from contact with ecosystems. Such contact need not be direct, as illustrated by the popularity of the virtual experience of distant ecosystems. Nor need such contact be with wild or exotic nature, as shown by the ubiquity of urban gardens, for example (Butler & Oluoch-Kosura 2006). Many cultural services are associated with urban areas and there is good evidence that biodiversity in urban areas plays a positive role in enhancing human well-being. For example, Fuller *et al.* (2007) have shown that the psychological benefits of greenspace increase with biodiversity, whereas a green view from a window increases job satisfaction and reduces job stress (Lee *et al*. 2009). This may have a strongly positive effect on economic productivity and hence regional prosperity. Several studies have shown an increased value of properties (as measured by hedonic pricing) with proximity to green areas (Tyrväinen 1997; Cho *et al*. 2008).

As the urban landscape grows in both area and population, it is increasingly becoming the landscape of greatest familiarity to current and future generations. It will likely become a landscape of increasing interest and investigation amongst scientists, academics, citizens, and governments. Urban nature is often the most visible and easily accessible component of this landscape and may reflect city planning, as in the case of metropolitan parks, greenways, wildlife corridors, and botanical gardens. However, just as often, urban nature represents the practical and spiritual needs of urban residents wherever it occurs in the cramped home and rooftop gardens, community gardens, and unchecked regeneration in abandoned lots. In both a historic and contemporary sense, the urban landscape exists as an open book, subject to the contributions and modifications imposed by all generations and demographic groups of different ethnic, cultural, and socio-economic origins. As such, manifestations of the natural landscape within cities are subject to constant change, as are the narratives and values affixed to them.

4.3.3 Governance of urban ecosystems

While different cities in different regions of the world face many of the same environmental challenges in light of growing populations (McDonald and Marcotullio, Chapter 4.1), the capacity to govern and respond to challenges attached to such fast transformations varies drastically. In trying to understand these differences, some classifications have been attempted, such as the one conceived by the United Nations University which conceptualizes world cities as belonging to three different groups; least developed, rapidly developing, and developed (UNU-IAS 2003).

Least developed cities have been defined as those which have experienced few benefits from globalization flows, although they are rapidly urbanizing. The ability to expand and modernize their infrastructure is impaired by socio-economic inequalities, often accompanied by severe environmental degradation (UNU-IAS 2003). Rapid urbanization has proceeded in the absence of accompanying economic growth, resulting in widespread poverty. Lagos, Nigeria is one of the emerging global mega-cities of the South, with a population expecting to reach 17 million by the year 2015 (UN Department of Economic and Social Affairs 2004).

Rapidly developing cities have often had growth facilitated by investment from public and private sources as well as far-sighted policies from their national government. Often, successes in these cities have come with heavy environmental repercussions. Technical and socio-economic changes occur so rapidly that these cities are ill-equipped to address the simultaneous issues of public health, equity, and environmental burdens that arise. This can be illustrated with the example of the increasing health and environmental costs of transportation in Asia, where now substantially more new cars are sold than in North America and Western Europe combined (UNU/IAS 2003).

Developed cities in rich countries deal with a specific set of environmental issues, often related to urban encroachment upon rural areas of agricultural and forest land. The sprawling growth at the margins of urban centres is a major contributor to greenhouse gas emissions as household numbers increase despite decreasing household size, and cars are increasingly relied upon to connect people to the city centres where they often work.

While such typologies have helped in differentiating between cities, and have helped caution against generalization of management models, they still do not reflect the degree of complexity required to manage linked social cultural and ecological systems at a variety of scales. It could be argued that some cities in rich countries, such as New Orleans and even New York City, still exhibit in some neighbourhoods, deep socio-economic inequities which are an obstacle to sustainable urban planning. Some rapidly developing cities, such as Shanghai, while being major contributors to greenhouse gas emissions, also are pioneering urban designs for carbon neutral development and energy saving measures at the frontier of science and technology. Some rich countries also have to contend with failed urban settlements and the problems associated with depopulation, as witnessed in some parts of the United States in the wake of the financial crisis.

These cases illuminate the necessity to address the needs of the population within urban areas through environmental services. Cities are at different stages in this process depending on their conceptual foundations and development status. In Chicago, for example, the origin of such an approach can be traced to far-sighted planners who at, the turn of the twentieth century, established a system of forest preserves in and around the Chicago Metropolitan regions. This greatly facilitated the establishment of Chicago Wilderness, which now includes Chicago metropolitan region and captures an array of rare ecosystems including tall grasses prairies, oak woodlands, oak savannahs, sedge meadows, marshes, bogs, fens, and other prairie wetlands. Through the dedication of an informed critical mass of people, the use of efficient science in policy bridging tools such as the Chicago Atlas of Biodiversity, and massive communication efforts, Chicago Wilderness succeeded in having citizens, policy-makers, and private land owners realize the existence and value of such ecosystems and having the Chicago Wilderness Biodiversity Recovery Plan integrated into regional development plans (Moskovits *et al*. 2004).

Urban nature has different meanings for different people, a central consideration in the implementation of an urban planning approach based on connected social and ecological systems. In cities of the developing world, urban nature is particularly valued as a source of food, livelihood, fuel, medicine, and spiritual fulfilment. Thus, in dealing with the urban landscape, there may be important factors that are not traditionally regarded as scientific in nature and yet need to be recognized, valued, and incorporated in urban planning. For example, in the city of Salvador, in North-eastern Brazil, contemporary cultural practices with strong roots in African tradition have led to the development of a pharmacoepia of herbs that have been documented to treat a total of 156 diseases (Voeks 1997), and are not only accessible remedies for the urban poor, but are also now being sold as teas in upscale Brazilian supermarkets (Kanth 1999).

If we are serious about adopting an approach that combines urban planning with healthy ecosystems, it is therefore necessary to start a conversation about urban ecosystem governance with all this diversity among cities in mind.

4.3.3.1 Global initiatives for improving urban ecosystem governance

Spurred into action by local movements, a number of global and international initiatives are starting to merge, linking municipalities around nature conservation goals. Reports increasingly illustrate efforts, investments, and challenges of local authorities in assuming a more proactive role to ensure a safe, prosperous, and healthy future for their citizens and natural resources. The ICLEI LAB initiative, begun in 2006, gathers 21 cities that have, at different scales and in different ways, operationalized nature conservation into municipal programmes. Some have taken steps to conserve biodiversity at the municipal level, such as Curitiba, and have in the process acquired the status of 'model cities'. Others have acknowledged that biological diversity conservation must take place at the regional level (e.g. Ile de France) and have strived to bring together the many jurisdictions and stakeholders concerned in regional development.

Another global initiative, the UNESCO URBIS Initiative promotes science and knowledge for sustainability and resilience, education for empowerment and change, and land-use planning for mediation and conflict resolution in the urban landscape. The UNESCO URBIS Initiative draws upon the pioneering 'Biosphere Reserve concept' defined in the 1995 Seville Strategy, and the ecosystem approach defined by the CBD, to promote the interdependent goals of biodiversity conservation, sustainable use, and equitable access.

The UNESCO URBIS seeks to contribute to an increased understanding, valuing, and reconnection of people and ecosystems in the urban landscape to:

1) Increase the resilience of urban social-ecological systems. Ecosystems such as wetlands and coastal areas may provide a buffer against natural disasters such as floods and hurricanes. But for this to be effective, the population must be made aware of these natural functions and learn to incorporate nature in their daily life, even in heavily urbanized areas. In New Orleans, it is now quite widely recognized that the human tragedies caused by Hurricane Katrina constituted a manmade rather than natural disaster. Social and economic fragmentation and lack of preparedness played a key role in the lack of initial reaction by some of the inhabitants, which led to the human debacle witnessed by all. To this day, less than half the population has returned and the public services, public health, and educational systems have barely been re-established, thus threatening the long-term viability of the city. Solutions thus lie not only in rebuilding the levees but more importantly in rebuilding the social fabric and cultural heritage so badly battered by the storm. The approach selected by Tulane University in a project conceived and implemented in collaboration with UNESCO and the Stockholm Resilience Center aims at creating a democratic and informed public forum to foster the dialogue on recovery through research, public debate, grassroots exchange, and policy formulation concerning social, urban, and ecological resilience. The River Sphere, a riverside campus of Tulane University dedicated to researching and understanding the Mississippi and the other great rivers of the world, provides such a platform. There it is proposed that teams of scientists, engineers, educators, industrialists, and residents will convene and conduct research to increase understanding of coastal ecosystems and communities. (Meffert *et al*. 2004).
2) Improve equitable terms of access to resources. In Cape Town, South Africa, the Cape Flats Nature project implemented by the South Africa National Botanical Institute builds good practice in sustainable management of nature sites in the City's Biodiversity Network. This is done in a people-centred way that develops local leadership for conservation action and benefits the surrounding communities, particularly townships

where incomes are low and living conditions poor (Stanvliet *et al*. 2004).

3) Promote adaptive governance of urban landscapes at the relevant spatial and temporal scales by bringing all stakeholders together around a neutral space for learning and creation of public wealth. By applying the URBIS methodology, the public and private sectors will be able to share findings and make joint decisions about urban planning and development. Intercity agencies will be able to collaborate and contribute to an improved social, cultural, and ecological management platform. Furthermore, cities and urban regions around the world will be able to learn from each other and instigate better resilient design and planning for urban sustainability.

There is hope that the ICLEI LAB and UNESCO URBIS initiatives will help to bridge science, planning, policy, and practice and reinvent a more holistic and symbiotic mode of urban development and resource use. This will not just require integration of disciplines and sectors. The emerging paradigm should be built on the principles that saliency, legitimacy, and accountability are integral part of everything we do as public and private actors in the landscape. That applies to science, education, and governments, but also to those in the corporate sector that shape our cities through infrastructure and networks.

In summary, in the urban landscape we propose to translate the general principles of the ecosystem approach and adaptive governance (Folke *et al*. 2005) into a set of methodologies, discourses, and planning tools ranging from global to local scales, recognizing that: (1) in an urbanizing planet cultural and biological diversity is key to resilience of social, economic, and ecological systems, (2) knowledge, either scientific or local, is key to management, (3) education is the main conduit for mainstreaming and empowering communities, and (4) adaptive management is not only a matter of building flexible institutions and governance systems at the relevant scale, it is also dependent on equity and particularly equitable access to land and resources.

CHAPTER 4.4

Water Services in Urban Landscapes

Peter Bridgewater

4.4.1 Introduction

Many urban areas have been founded along the banks of great rivers, or in an area of freshwater seepage, where groundwater is (or increasingly was) close to the surface. There is thus a close cultural connection between urban settlements and wetland systems. The following vignettes give eleven different flavours of urban wetland ecosystems in urban areas, yet all are linked by common themes.

Landing at Rome's Fumicino Airport and taking the train, you pass through flat lands, with extensive stands of *Arundo donax*, and occasionally very low hills, with again dense stands of *A. donax* at the foot, or seepage line. In the distance scattered *Pinus pinaster* groves can be seen, and clumps of planted or escaped *Eucalyptus camaldulensis.* But the underlying ecological connection is clearly the wetland systems, including seepage lines. This ecological connection forms a visible map of the hydrological systems of the coastal plain.

North-east from Rome is perhaps the world's ultimate wetland city—Venice. Built in a functioning lagoon, Venice expresses all the delights and frustrations of a living wetland city. Venice is inscribed on the World Heritage List (UNESCO 2009b), but as a cultural site. Yet the major threats to this listing come not directly from human activities, but from natural events causing changes to the wetland system of the lagoon.

In Dubai, city of blowing sands and ever-higher concrete and glass towers, there is a wildlife refuge, a wetland, lined by mangroves (*Avicennia marina*). The inland edge of this reserve has little dense stands of *Phragmites australis*, and, occasional *A. donax*. The wetland is the Ras Al Khor Wildlife Sanctuary, a Wetland of International importance listed under the Ramsar Convention on Wetlands (Ramsar 2009a). In a few special places there are constructed wetlands, featuring primarily *P. australis*, sometimes supplemented by *A. donax*. These wetlands are supplied by wastewater from showers and kitchens, and primarily treated sewerage. And away from the development, native halophytic shrubs that represent highly ephemeral wetlands dominate the 'desert'. While *A. donax* is an alien species and *P. australis* is rare in the sand plain they do form stable wetland systems, provided the supply of water is kept up. They provide also a very important ecosystem service of removing nutrients and other pollutants from the waste water.

In Perth, Western Australia, there are many kinds of wetlands, ephemeral, and open water fringed by macrophytes. The Becher wetlands, a group of wetlands, interspersed with the sprawling development, is described in detail by Semeniuk (2007). These wetlands include lakes, ephemeral wetlands, and small wooded swamps. Included in the overall wetland landscape of the Perth region are the salty shores of the Swan River, and (too many) obliterated wetlands, now visible only through old aerial photographs. Again we find clumps of *A. donax*, *P. australis*, with *Eucalyptus rudis*, a western member of *E. camaldulensis* group, lining some swamps and riverbanks: compared with Rome, therefore, a sort of reverse exchange of invasive alien species.

And in Brisbane, on the other coast of Australia, as you land at the Airport you notice abundant mangrove forest—much of which was created during the building of the new airport in 1988, admittedly after extirpating some existing wetland habitat.

Airport expansion here continues to be controversial, but the government is demanding continued attention to conservation and management of biodiversity (The Australian 2007).

Hangzhou, China, has a wonderful wetland park (Xixi) of over 3,000 fish ponds, each one small and self contained, with fringing vegetation which includes *P. australis*, and on the drier ridges Mulberry (*Morus alba*) and Persimmon (*Diospyros kaki*) trees. In the centre of this area is a centuries old Buddhist temple, where the monks spent time in contemplation, writing evocative poems. The Park has an educational centre established, and has become a focus for enjoyment and education, as well as ecosystem productivity. The Xixi wetlands management programme included restoration and recreation of wetland ecological systems, rebuilding the wetland food chain, re-establishing wildlife ecosystem diversity and stability, restoring the water purification function of the wetlands, and the prevention (and decrease) of eutrophication (XiXi 2009).

Kampala, Uganda, lies on the shores of Lake Victoria, and largely covers the hills rising up to the east of the lakeshore. However, in low-lying parts of the city there are extensive papyrus swamps, some of which have been cleared, but the majority retained, as their function in moderating flooding is well recognized by the local community. They are also a source of water for some of the population. In addition, wetland edges are often used for food cultivation (especially Yam and banana), although this cultivation rarely penetrates far into the wetland, and helps provide food for nearby residents. One particular site, Lutembe Bay Wetland System (listed as a Wetland of International Importance under the Ramsar Convention (Ramsar 2005)) is dominated by *Papyrus*, *Phragmites*, and *Typha.* Threats come from the Water hyacinth, *Eichhornia crassipes*, an introduced submerged plant affecting open water in the area. The introduced species of fish Nile Tilapia, *Oreochromis niloticus*, *Oreochromis leucosticus*, and *Tilapia zillii*; and Nile perch, *Lates niloticus*, in Lake Victoria has led to the extinction of several *Haplochromine* species in the vicinity of the Lutembe Bay Wetland System.

New York, with its complex of islands and fringing riverine edges is also a wetland city. The edges of the Hudson River, again with *P. australis* dominant, provide a range of ecosystem services for the metropolis, as well as habitat for a range of wildlife, including migratory species. In New York City environs tidal wetland losses result from both human caused and natural disturbances. These disturbances include:

- dredging
- watershed development
- filling
- eutrophication.

But wetlands are not just lost, they are also gained. For example, Shinnecock Bay showed a gain of 161 acres of tidal wetlands as a result of a landward movement of the tidal wetlands boundary from 1974 to 1995 (NYDEC 2009). Moriches Bay showed a gain of approximately 100 acres of tidal wetlands as a result of a landward movement of the tidal wetlands boundary from 1974 to 1998. Loss of wetlands because of permitted and unpermitted human activities was too small to be detected. The main cause of wetlands destruction has shifted from human caused factors, such as filling, to natural factors such as storms and flow restrictions.

At the delta of the vast Parana River, seaward of Buenos Aires, are less densely populated municipalities, encompassed by the Delta del Paraná Biosphere Reserve (UNESCO 2009a). This coastal freshwater delta was declared as a biosphere reserve in 2000 and is located just north of Buenos Aires. Many species are at their southernmost limit of distribution. The flooded riverbeds are dominated by *Schoenoplectus californicus* and *Scirpus giganteus*. The Biosphere Reserve also contains low forests, forest ecosystems, and secondary forests with Black Cottonwood (*Populus* spp.) and several *Salix* spp. The latter are exploited for commercial purposes. The establishment of the Biosphere Reserve aims at revitalizing the economy of the region at the same time as conserving the natural and cultural values of the area. This biosphere reserve is dedicated to wetland protection, but also to its sustainable use, and education about its important function. While direct inhabitants are probably fewer than 3,000, it is an increasingly popular destination for citizens of Buenos Aires.

Finally, 8 km from the centre of London, there is a recently developed wetland complex, again featuring *P. australis* and many other species, but developed from artificial reservoir basins. The site is also a Site of Special Scientific Interest under UK legislation, supporting nationally important wintering populations of shoveler (*Anas clypeata*) and an assemblage of breeding birds associated with lowland waters and their margins (Natural England 2009). In addition to the nationally important numbers of shoveler, the site also supports significant numbers of wintering gadwall. Barn Elms Wetland Centre comprises a mosaic of different wetland habitats with the majority of the site comprising areas of standing open water, grazing marsh, and *Phragmites* reedbed. Other significant habitats include *Salix* woodland and scrub. The Barn Elms reservoirs were constructed in 1886 but became redundant in 1989. Wetland habitat creation was initiated in 1995 and the site is now being managed as a nature reserve, as well as an educational and visitor facility.

These eleven urban areas and their associated wetland ecosystems have many of the themes that characterize urban wetlands in the twenty-first century, that is:

- an association with human activities, especially development and culture;
- invasive species,
- migratory species feeding and breeding ground;
- a focus for education;
- being recognized by international environmental agreements;
- an ability for the wetland to be restored or constructed; and
- a role in providing a focus of sustainability in the urban system.

This chapter sets those themes in the context of the emerging paradigm of ecohydrology and the application of the ecosystem approach of the Convention on Biological Diversity.

4.4.2 Wetlands and water in the urban environment

The hydrological cycle (Fletcher & Deletić 2008) supports and links all components of the environment, including the urban environment (see Illgen, Chapter 1.4). Water flows throughout the environment, from the atmosphere, on the surface and below the surface of the land. By environment I mean the various biotic and abiotic elements that comprise marine, terrestrial (including aquatic) and subterranean (which includes aquifers, cave systems, and the saturated zone of the soil horizon) ecosystems, as well as the atmosphere. The hydrological cycle links all these components of the broader environment, and this means that water resources are linked, via the water itself, to all the other components of the broader environment (such as soil, biodiversity, and air).

Water itself appears as liquid, solid, or gaseous (vapour) forms in the environment, depending on the particular situation. In the atmosphere water is usually as the vapour or liquid form, or occurs temporarily in the solid form as hail or snow in winter seasons. In terrestrial ecosystems, water is in the vegetation and the unsaturated zone of the soil horizon. Such water becomes part of the evapotranspiration cycle—the term 'dash water' is used to describe this part of the water cycle. Marsalek *et al*. (2007) note that the direct and indirect impacts of different ecosystems or components of the water cycle need to be quantified with respect to local climate, urban development, cultural, environmental and religious practices, and other socio-economic factors.

Water in aquatic, marine and subterranean ecosystems appears in its liquid form, where it is usually termed 'dot water'—this includes water held in aquifers, or in the saturated zone of the soil horizon. Aquatic ecosystems are those in which water is generally fresh or brackish (but in inland arid areas may include hypersaline systems). Coastal marine ecosystems include the estuarine and near-shore marine aspect of water, while the offshore marine ecosystem's primary influence on the hydrological cycle is through global, continental, and regional weather patterns.

In urban systems, water can be both dot and dash, but there are also substantial amounts of water which are grey (meaning pathogen free but not necessarily cleansed of all nutrients and chemical pollutants) and black (meaning untreated water from human sewage systems, or untreated industrial wastewater). These forms of water are human in origin and need various degrees of treatment, which has typically been mechanically or engineering based. However, new technologies allow a mix of hard and soft engineering approaches to deal with these types of water—where soft engineering refers to the construction or use of organic wetland systems as treatment sites for all grey and possibly some black waters.

Figure 4.4.1 shows the way these 'coloured waters' move through the hydrological cycle. The figure is simplified to make the main pathways clear, and the many indirect impacts are omitted.

There are biophysical, biochemical, and ecological links within and between each of the coloured waters. Ecological processes play a critical role in regulating the hydrological cycle, and they are themselves affected by biophysical and biochemical processes occurring within the hydrological cycle. Here, the structural, functional, and compositional aspects of biodiversity play a variety of roles, at several different scales, in governing linkages within and between the coloured water components of the hydrological cycle. Additionally, ecological functions and processes linked to the hydrological cycle both affect people as part of associated socio-cultural systems, and are, in turn, affected by human activities.

Water in the hydrological cycle is also affected by natural and human-induced processes of change to land, water, and wetlands (see Illgen, Chapter 1.4). These can be due to changes in the topography and morphology of the landscape, which primarily affect the 'blue water' component of the hydrological cycle, or due to changes in vegetation and land cover, which primarily impact on 'green water' through affecting infiltration and evapotranspiration rates and patterns.

Changes in land and water environments affect the rates and pathways by which water moves within the hydrological cycle, and also affect the quality of the water in its various forms and places. Connections between the hydrological cycle and the broader environment are bi-directional, in that direct impacts on the non-water aspect of the environment can affect water, while direct impacts on water (such as abstraction or waste discharge) can affect the broader environment as well. This is

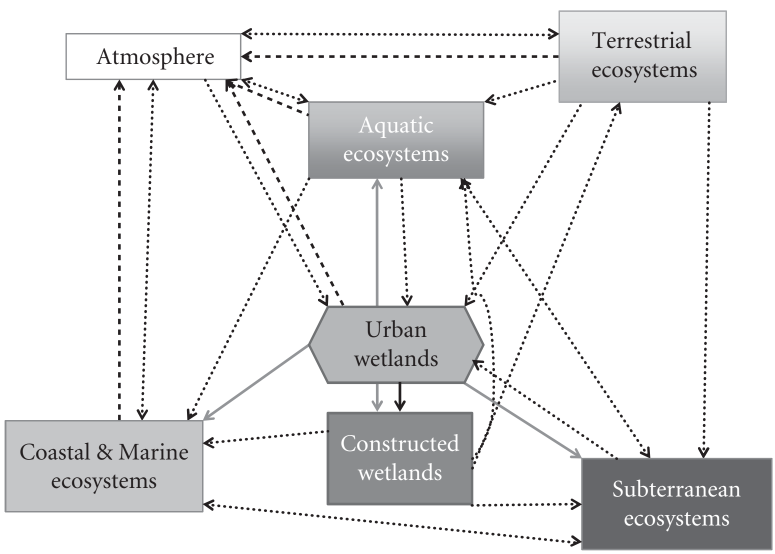

Figure 4.4.1 Relationship between green waters (dashed lines), blue waters (dotted lines), grey waters (grey lines), and black waters (black lines) (see text for definitions) and their flows through ecosystems, including urban and constructed ecosystems. The constructed ecosystems are wetlands designed to treat grey and black wastewaters specifically, but grey waters from urban areas flow to other wetland ecosystems. These flows are complex, interlinked, and may contain surprises in poorly managed landscapes

nowhere more evident than in the urban environment, where ecosystems suffer most from misuse of the water component.

In most countries, the conventional water sector, which is engineering in focus (and usually in management), deals with water primarily as a commodity. Water in urban environments is delivered to people through some kind of infrastructure such as pumps and pipes, and is sourced from either or both groundwater or surface dams. This water is thus derived exterior to the component urban ecosystems.

In urban areas, most residents see water as a service which is provided by the local authority, and for which they 'pay'. They do not see it as a resource that plays a key role in both their immediate, as well as the wider, environment, much less driving the functioning of key ecosystems, including ecosystem components of the urban landscape and its ecological infrastructure. The problem with the conventional approach to management of water as a commodity is that many of the values that people place on water, aside from just having an adequate supply when they turn on a tap, are dependent on water being a component of healthy, functional ecosystems.

In some urban areas, water is derived from ecosystems a distance away (e.g. New York) but in others (e.g. Perth, Western Australia) water is derived from groundwater sources. In the case of distant supplies the immediate nexus between supply and need is not obvious, but where water is abstracted from groundwater at a rate greater than replenishment a range of issues can arise, including, inter alia, subsidence of the urban area (e.g. Bangkok, Mexico City). Other cities using a mix of ground and surface water resources can none-the-less be affected by pollution issues—e.g. in London and Paris water can have higher than desirable levels of nitrate pollution, as the runoff from farmlands around these urban areas recharges the ground water reserves that are drawn on for domestic water supply.

4.4.3 Ecohydrology

Ecosystems, particularly those in which water is a critical component or the main component, are typically resilient and can withstand a certain degree of impact, including abstraction of water, use of food and fibre resources, and transport of goods through the system. However, exceeding these limits changes the structure and function of an ecosystem irreversibly, leading to irreparable changes in the range, availability, and quality of the ecosystem services formerly provided, including protection, production, and purification of water supplies. Such changes in ecosystem services can be irreversible, or at the very least impose strong constraints on ecosystem management.

In many cases wetlands, which act as provisioning of ecosystem services for urban environments, are especially under pressure—whether inside the urban periphery or external to it. A case in point here includes the Wetland of International importance (Ramsar Site) known as the East Calcutta wetlands (Ramsar 2002). These wetlands are essentially resource recovery systems, developed by the local people over time, using wastewater from the city. In the recovery process it treats the wastewater and has saved the city of Calcutta from constructing and maintaining a wastewater treatment plant. It is also the only metropolitan city in the world where the government has introduced development controls to conserve the water-bodies. These wetlands, however, are still under an intense encroachment stress of urban expansion. This task of conservation, therefore, needs further consolidation.

The quality of any living system cannot over time exceed the quality of the environment in which it is found. Of course, this is a process that is subject to many feedback processes: in today's terminology the drivers of ecosystem change. The Millennium Ecosystem Assessment (MA 2005b) defined a new conceptual framework (MA 2003), placing emphasis on the management of the environment to deliver ecosystem services, and through those services to enhance human well-being (McDonald and Marcotullio, Chapter 4.1). Well-being is more than simply human health, and reflects a more holistic approach. But to deliver better human health outcomes, we need to have healthy ecosystems—that is, ecosystems that are able to continually deliver services to people and the biosphere.

One approach links ecology and hydrology—sometimes combined (as in UNESCO 2009c) as ecohydrology (Bridgewater 2002; Wagner *et al*. 2008).

Ecohydrology has objectives that can be largely defined by the people and societies dependent on the ecosystem and associated water resources.

For wetlands there are four key points that underscore exactly what ecohydrology is, viz:

- Integrating water and biodiversity science at management relevant spatial and temporal scales.
- Understanding that ecological change is inevitable, and the role of people in managing change.
- Understanding the role of ecosystem services.
- Using ecosystem properties as indicators of change.

In linking these four points the natural and the social sciences must be brought together, preferably using the ecosystem approach developed by the Convention on Biological Diversity (CBD) (see Alfsen *et al*., Chapter 4.3). The ecosystem approach has, as its key feature, the relationship between people and the rest of the biosphere, and how that relationship is managed. The CBD website has full details of the evolution and current status of the ecosystem approach (CBD 2009), but see also Shepherd (2004) for another perspective. This organizing framework deals with 'natural' wetland systems and is also applicable to artificial wetlands constructed inter alia for sewage treatment, and for amelioration of flood plains. The role of such so-called artificial systems in ensuring continued functioning of linked ecological systems, and maximizing the expression of wetland biodiversity in urban and peri-urban systems, is particularly important.

The need for maintaining and enhancing urban and suburban populations of wildlife has thus greatly increased in recent decades (Washington 1978). A key issue is that urban planners often have insufficient awareness and expertise in wildlife matters (Davey 1967; Geis 1980). This lack of knowledge is often compounded by inadequate support from resource agencies for the development of appropriate strategies for wetland ecosystem conservation and management and their subsequent implementation. The solution to this dilemma is to encourage greater collaboration between agencies responsible for biodiversity and ecosystem management and municipal planners (Greer 1983) and to ensure planners are more alert to the link between wildlife (biodiversity) and the delivery of ecosystem services (MA 2005c).

Restoration of wetland to encourage particular features is an essential part of urban wetland management. Eglington *et al*. (2008) show that shallow, small-scale flooded areas are of critical importance for breeding waders. Management tools such as foot drains, coupled with appropriate hydrological management, provide a means of retaining water throughout the breeding season. While their paper refers to a broader landscape approach, there is every reason to use these techniques in an appropriate urban environment. Installation of these features is relatively simple, but maintaining sufficient water levels within the system is critical, especially in the face of increasingly unpredictable water supplies associated with climate change—and that is exacerbated in urban environments.

Similarly, Smart *et al*. (2006) emphasize that 'wet' features, of critical importance for breeding redshank which are common on coastal marshes, can be deliberately established on inland sites. Coastal marshes are less and less common and frequently threatened by dynamic coastal processes, whereas inland marshes are more abundant but largely unsuitable for breeding waders at present. They analyse the scope for improving the management of inland marshes for breeding redshank. As habitat suitable for breeding redshank frequently supports a range of other wader species, they propose that this information can also direct management efforts to improve breeding wader populations in the wider countryside.

4.4.4 Healthy wetlands, healthy people

For urban areas, links between nature and health are often seen as important—and often in combative terms (see Tzoulas and Greening, Chapter 5.2). Concern about health and the environment is essentially concern about the relationships that exist between people and the rest of the biosphere, and people frequently handle these relationships poorly. The need to integrate more fully the goals of conservation and ecosystem management and health ethics for a sustainable society is becoming ever clearer (Honari *et al*. 1999 and other refs), and this is especially true for wetlands.

From the perspective of human health, wetlands (as defined by the Ramsar Convention (Ramsar 2009b) have a real identity crisis. They are often seen simply as human health hazards, with malaria, bilharzias, and a whole host of other parasitic diseases typically associated with them. Two centuries ago, the dank surroundings of lakes and, worse, swamps were enough to provoke people into believing that to be simply close to such a landscape feature was to risk catching a fever. Urban wetlands suffered especially as people were uncomfortable at living next to what was seen as a source of disease.

Recently, there has been an upturn in the rate of emergence or re-emergence of infectious diseases associated with wetlands, and those in urban fringes are especially concerned. Factors contributing substantially to this trend include:

- intensified human encroachment on natural environments;
- reductions in biodiversity (including natural predators of vector organisms);
- habitat alterations that lead to changes in the number of vector breeding sites or in reservoir host distribution;
- niche invasions or interspecies host transfers;
- human-induced genetic changes of disease vectors or pathogens (such as mosquito resistance to pesticides or emergence of antibiotic-resistant bacteria); and
- environmental contamination by infectious disease agents (WHO 2007).

Water-related diseases affect over 2 billion people a year. Providing clean water and sanitation to poor communities would take pressure off their need to unwisely use wetland ecosystems, reduce wasteflows, and improve freshwater and coastal water quality.

Many of the people and sites affected adversely by ecosystem changes are highly vulnerable and ill-equipped to cope with further loss of ecosystem services. Ecosystem changes, with an increasing risk of non-linear changes, including accelerating, abrupt, irreversible changes, potentially have a catastrophic effect on human health. The increased likelihood of these non-linear changes arises, in part, from the loss of biodiversity and growing pressures from multiple direct drivers of ecosystem change.

In May 2002, at a speech at the American Museum of Natural History, the UN Secretary-General outlined a so-called WEHAB initiative, by identifying five major areas for the World Summit on Sustainable Development, areas where concrete results are both essential and achievable. The WEHAB areas were: **W**ater, **E**nergy, **H**ealth, **A**griculture, and **B**iodiversity. The initiative recognized for the first time the critical importance of biodiversity in delivering services in each of the other sectors. And by including water, biodiversity, health, and agriculture it also brought together key concerns for the Multilateral Environmental Agreements.

Although a WEHAB Working Group was established, and published *A Framework for Action on Biodiversity and Ecosystem Management* in August 2002, this initiative has largely disappeared from view, and even the url no longer works! Crucially, the WEHAB working group paper highlighted the need to shift focus from the proximate causes of biodiversity loss to the underlying causes. It focused on two key action areas: integration of biodiversity—and principles of sustainable development—in country development programmes and economic sectors; and halting the loss of biodiversity and restoring, if possible, biodiversity in degraded areas, as part of reversing loss of environmental resources. While these principles are applicable everywhere, they are especially important for wetland systems.

Importantly, reflecting the close linkage between the WEHAB framework and the Millennium Development Goals (MDGs) (UN 2009), the two action areas in this WEHAB paper are built upon and consistent with targets of MDG 7 to 'ensure environmental sustainability'. The action frameworks provide indicative targets or milestones, with examples of activities—a 'menu' for further development of activities. While the linkages between biodiversity and the health aspects of development are still little understood, Harvard (2007) has an example of some key work in this regard.

Ideally, we should be building on the biodiversity linkages evident between the five WEHAB areas.

And yet, vital though it seemed at the time, WEHAB has not found much favour since the lead-up to the Johannesburg summit. With the theme of 'Healthy wetlands, healthy people', it represents a latent framework for the wetland biodiversity community to develop and use. However, we need to develop specific strategies, tools, and ways of measuring success—such work is being undertaken through the Ramsar Convention Science and Technical Review Panel (Ramsar 2009c). This will include monitoring ecosystem functions in all parts of the world and developing environmental assessment and indicators.

Human health is not just about being not sick, however. The Millennium Assessment uses rather the term 'human well-being'. Solving issues of poverty and management of natural disasters are critical to achieving human well-being, as recognized by the last Ramsar Conference of the Parties (Ramsar 2009d). For the poor, food security depends to a large extent on biodiversity, through direct consumption of wild foods, wild plants for farm production, medicines, fuel, and the trading in species and products.

Conversely, loss and change of biodiversity can increase hunger and food insecurity. Wetland ecosystem degradation means less water for people, crops, and livestock, lower crop yields, and higher risks of natural disasters. Nevertheless, the relationship between biodiversity and poverty is complex and not linear. This is exemplified through human—wildlife conflicts, increased mobility of pests and diseases, and introduction of invasive species, innocently or deliberately. And nowhere more obviously than in urban wetlands.

4.4.5 Future research directions

Wetlands in an urban environment pose research and knowledge management challenges not just from the physical and biological environments, but also from the socio-economic and cultural environments. Elaborating research programmes that seek to integrate these worlds, as well as testing the resilience limits of each, are essential to enhance our understanding of urban wetlands.

Additionally, understanding the physical and biological effects of above and groundwater flows is another key element. This is of course, essential and special in urban areas that are close to or atop karstic systems—but it applies more generally as well. Wetlands are often surface expressions of the groundwater components—and may even be indicators of the health of such groundwater systems. (Semeniuk 2007).

Accordingly, the list below is a sample of 10 key areas which need further study, or which are embryonic research areas deserving of future attention.

1. Understanding the range and distribution of urban wetland types, including lakes, rivers, swamps, and groundwater aquifers.
2. Identifying how different urban wetland types are reacting to current and projected rates of groundwater abstraction and how recharge from surface wetlands to groundwater can be improved.
3. Examining the consequences of global change (including climate change) on the loss and degradation of urban wetland biodiversity, including species that cannot relocate and migratory species that rely on a number of wetlands at different stages of their lifecycle.
4. Building an understanding of urban wetland ecosystem connectivity in space and time in the urban environment.
5. Testing how much continuing loss and degradation of urban wetlands is leading to a reduction in the delivery of wetland ecosystem services, and examining the demand for these services, and the consequences of their reduction/loss on human health.
6. Examining the science–policy interfaces necessary to ensure the cross-sectoral focus that operates on urban wetland ecosystems can be developed to allow policy-makers and decision-makers to conserve and manage wetland ecosystems and their services in the context of achieving sustainable development and improving human well-being.
7. Urban wetlands deliver a wide range of critical and important services (e.g. fish and fibre, water supply, water purification, coastal protection, recreational opportunities, and increasingly, tourism) vital for human well-being. Understanding the economic valua-

tion of these services is critical to engaging in the broad debate about management of ecosystems, the biodiversity that supports those ecosystems, and how to maintain the natural functioning of wetlands.

8. Urban wetland loss and degradation has primarily been driven by land conversion and infrastructure development, water abstraction, eutrophication, pollution, and over-exploitation. Research into the balance between these threats is needed to ensure long-term conservation and management of wetland ecosystems.
9. Since management of wetlands and water resources is most successfully addressed through integrated management at the river (or lake or aquifer) basin scale; understanding how wetlands in a fragmented urban environment can persist and successfully deliver ecosystem services.
10. Research into the application of the wise use principle and guidelines of the Ramsar Convention (Bridgewater 2008), and its role in maintaining ecosystem services of the wetland in urban environments.

CHAPTER 4.5

The Role of Ecosystem Services in Contemporary Urban Planning

Johan Colding

4.5.1 Introduction

Urban spatial planning has traditionally meant planning 'for development'. Due to past decades' pressing challenges related to the global biodiversity crisis and climate change, the planning mode has shifted towards 'sustainable development' of cities. Today urban planners interested in achieving sustainable development, or 'sustainable cities', adopt different approaches in plans and designs for cities. Although planning nowadays includes sustainability as a central criterion when laying out roads, streets, buildings, and other components of the built environment, it is often argued that conventional planning practices ignore the natural configuration of the land (Benedict & McMahon 2002). As a consequence, ecosystems are rapidly becoming fragmented, transformed, or entirely lost, causing loss of ecosystem services, leading to, impoverishing water and air quality, for example. This eventually erodes ecosystem resilience, or the capacity of natural systems to buffer and reduce disturbances like heat waves, flooding, pollution, and anthropocentrically induced management mistakes.

In many countries today, the pace of urban land development far exceeds the rate of population growth. For example, the amount of urbanized land in the United States increased by 47 per cent between 1982 and 1997. During the same period, the population grew by only 17 per cent (Fulton *et al*. 2001). Moreover, across the largest 100 metropolitan areas in the US, employment decentralization has become the norm, with only 22 per cent of urban residents working within three miles of city centres and over 35 per cent working more than ten miles from the centres. As a consequence, air pollution from traffic emissions increases, adding to greenhouse gases, and eventually leading to climate change. This has led many planners to realize that the problem is not growth itself, but the pattern of growth (Daniels & Lapping 2005).

Urban sprawl has been identified as America's leading land-use problem (Freilich 1999). There are several definitions in the literature of what urban sprawl is, but it generally refers to the spread of urban congestion into adjoining suburbs and rural areas, leading to the loss of ecosystems or as 'low-density, dispersal automobile dependent land-use patterns' (Litman 2009).

There exists a whole arsenal of strategies that modern-day policy-makers and planners can adopt to steer away from unsustainable urban growth in and around cities. Of key influence for integrating ecology in urban planning has been Ian McHarg's work on comprehensive, ecologically-based planning and design (McHarg 1969). The applicability of systematic land-use planning for determining areas for development, or for conservation, involving the system of map overlays of different categories of natural features (e.g. hydrology, geology, soils, vegetation, and wildlife), represents a prominent feature of this approach. Similarly, systematic conservation planning has more recently emerged as a range of methods to determine, implement, and manage a set of areas containing desired conservation targets with the minimum expenditure of resources (Gordon *et al*. 2009).

In this chapter, I review two of the most prevalent planning strategies proposed to combat urban sprawl:

smart growth theory and green infrastructure planning. The former is predominantly derived from frustration over the failure of American planning projects, and is becoming increasingly adopted among planners in North American and European metropolitan regions. The latter is predominantly proposed by ecologists and conservationists, and has been influential in conservation planning in many countries (see also Pauleit *et al.*, Chapter 5.3). By elucidating some of the characteristic features and propositions of these seemingly disparate approaches to combat urban sprawl, I set these in communication with insights pertaining to the generation and management of urban ecosystem services, as identified in recently emerged empirical studies. I discuss the implications of the two approaches, as well as shedding light on urban designs that hold the potential to be developed into more exhaustive frameworks for governance of urban ecosystem services. I conclude by summarizing the major insights of this chapter.

4.5.2 Urban sprawl and ecosystem services

Urban development is claimed to generate some of the greatest local extinction rates of species, and frequently eradicates large proportions of native flora and fauna (McKinney 2002). Land-use in urban areas has a particularly strong influence on biodiversity; some scholars predict that it will likely have the largest effect on terrestrial ecosystems in the coming century (Sala *et al*. 2000). As recent studies of satellite data indicate, land-use continues to intensify in formerly occupied areas (e.g. urban areas) often overlapping with areas rich of biodiversity (Ricketts & Imhoff 2003). This is because humans tend to settle in areas with high ecosystem productivity, on lands suitable for agriculture, or in low elevation and coastal areas with high levels of biodiversity (Colding 2007).

The uncontrolled spread of urban congestion into adjoining suburbs and rural areas often leads to a net loss of natural habitats, and consequently to the erosion of many ecosystem services (see also Pauleit and Breuste, Chapter 1.1). On the other hand, urban biodiversity usually peaks at the suburban scale of cityscapes, where species tend to be 'urban adapters', confined to forest edges and adjacent open lands. Species within this category often exploit many resources, including human-subsidized foods (McKinney 2002). Such species tend also to be less sensitive to the presence of humans and pets. It is important to recognize that while biodiversity often peaks at suburban levels, urban sprawl may lead to the loss of overall regional biodiversity in the urban landscape (Colding & Folke 2009).

There has so far been hardly any attempt to link urban planning with the sustainable governance of ecosystem services. It can be argued, though, that many ecosystem services provided by natural systems resemble those services that urban planning strive for, that is, facilitating and distributing services for the general public, freely enjoyed on a day-to-day basis. Like public greenspaces, ecosystem services are strategically critical to the health and quality of life of a city environment. While provisioning services (products obtained from ecosystems like food and fibre), are often privately consumed, they represent priced goods that may be enjoyed by a multitude of urban residents at markets. On the other hand, regulating services (benefits obtained from regulation of ecosystem processes like air and water filtration), cultural services (non-material benefits obtained from ecosystems, like spiritual enrichment, cognitive development, recreation, and aesthetic experiences), and supporting services (necessary for production of all other ecosystem services), are considerably more difficult to evaluate in economic terms. In and around cities, ecosystem services provide a range of 'free' services enjoyed by a greater set of urban dwellers. Since they are not adequately priced (and valued), they run the risk of becoming mismanaged, adversely affecting most dwellers. It is, therefore, essential to develop institutions and design systems for their sustainable management.

Given that urban sprawl represents one of the leading land-use problems in many city regions (Freilich 1999), both 'smart growth planning' and 'green infrastructure planning' have become popular in many countries in coming to grips with this phenomenon. They are more closely described in the following.

4.5.3 Green infrastructure planning

Having its roots in the planning and conservation efforts that started 150 years ago, 'green infrastructure' is an approach to combating the adverse effects on biodiversity resulting from urban sprawl (see also Pauleit *et al*., Chapter 5.3). Green infrastructure can be described as 'an interconnected network of green space that conserves natural ecosystem values and functions and provides associated benefits to human populations' (Benedict & McMahon 2002: 12). Much of the foundation of green infrastructure planning draws on the principles of island biogeography theory (MacArthur & Wilson 1967), inspired by three patterns long recognized as part of island ecology:

- that larger islands tend to support more species;
- that remote islands tend to support fewer species;
- that there is often a turnover of species on islands, with newcomers replacing other species that become extinct (Forbes *et al*. 1997).

The application of island biogeography principles to urban settings has been widely used in the planning and design of terrestrial nature reserves. When transferring these principles to terrestrial systems, the key assumption made is that 'habitats' or reserve sites themselves cannot function as isolated parts, or 'islands', when surrounded by land of different types that is hostile to the species living within the reserves (Forbes *et al*. 1997). The surroundings of a reserve, the 'matrix', or the 'background ecological system', has the highest degree of connectivity of a landscape, and largely determines to what extent organisms may move or spread to other natural habitat patches. In the design of habitat networks, connectivity is the central objective to achieve in green infrastructure planning. Connectivity is a measure of how connected or spatially continuous a network is (Forman 1995). In green infrastructure planning, this can be achieved by way of habitat corridors, areas of land that facilitate species movement between larger habitats (Forbes *et al*. 1997). This involves protecting, creating, and restoring connections between parks, reserves, and other important ecological areas. Such habitat corridors may include hedgerows, road and railway verges, strings of open space sites, parkways, and riparian trails that function as a route for species movement. In order to avoid potential negative effects of habitat connections, wildlife biologists usually recommend that it is better to protect existing connections between habitats than to create new ones (Forbes *et al*. 1997). Green infrastructure planning, therefore, is said to work best when the framework pre-identifies both ecologically significant lands and suitable development areas (Benedict & McMahon 2002).

One prominent example of habitat networks that has long played a significant role in the development of urban, sub-urban, and even rural areas is greenways, defined as a 'network of land containing linear elements that are planned, designed and managed for multiple purposes including ecological, recreational, cultural, aesthetic, or other purposes compatible with the concept of sustainable land-use' (Ahern 1995: 134). Urban greenway systems date back to the nineteenth century, as part of an evolving landscape form that has its roots in the planning concept of medieval towns (Searns 1995). Besides ecological, recreational, and heritage motives, urban greenways need to comply with a core set of human dimensions to improve their success in a community, including cleanliness, naturalness, aesthetics, safety, access, and appropriateness of development (Gobster & Westphal 2004). Greenways are therefore mainly designed and managed to provide recreation and amenity rather than for the protection of biodiversity (Benedict & McMahon 2002).

The principles for green infrastructure are also used by planners in management of greenbelts and green wedges, that is, belts of predominantly undeveloped land that limit urban sprawl and connect rural areas with suburbs and urban areas. While the greenbelt idea originated from Ebenezer Howard's notion of a model 'Garden City' in England in the late nineteenth century, contemporary planners often apply the ideas of biological core areas, corridors, and buffer zones on greenbelts in master plans for cities. Proponents of green infrastructure planning often argue in favour of compact urban development to combat urban sprawl by advocating for the preservation of natural habitats in urban fringe and rural areas (see e.g. Benedict & McMahon 2002; Daniels & Lapping 2009). Hence, green infrastructure planning is largely compatible with smart growth planning, described next.

4.5.4 Smart growth planning

Due to the explosive sprawl witnessed in many peri-urban areas, matched by a decline or slower growth in the central cities and older suburbs, Katz (2002) describes how a new planning approach arose over a decade ago in the United States as a reaction to the widespread frustration with sprawling development patterns. This new thinking—commonly referred to as 'smart growth' or 'New Urbanism'—argues that metropolitan settings can grow in radically different directions if only major and deliberate government policies on land-use, infrastructure, and taxation are adopted.

The term 'smart growth' (abbreviated as SG in the following) became more widely popular when Maryland's 1997 legislation provided planning initiatives to promote compact development to combat sprawl (Daniels & Lapping 2009). In parallel with 'compact urban development' that concentrates growth in the centre of a city to avoid urban sprawl, SG advocates compact, transit-oriented, walkable, bicycle-friendly land-use, including mixed-use development with a range of housing choices. A dominant feature of urban designs involving SG is accessibility, the ability for urban residents to reach desired goods, services, and activities within close ('walkable') distance for example (Fig. 4.5.1). Planners adopting SG strive therefore to locate new development within already developed areas, encouraging in-fill development and redevelopment of older facilities and brownfields.

Proponents of SG planning argue that current planning strategies of land-use and transport patterns often reflect distorted market-oriented policies rather than reflecting true consumer preferences. Therefore, SG strategies are said to represent market reforms to 'correct' these distortions with the goal of increasing efficiency and equity, and making consumers and the economy better off overall (Katz 2002; Litman 2009). Strategies adopted to change current consumption behaviour may range from adopting various pricing mechanisms, such as additional cost fees for housing that requires more dispersed infrastructure and for parking in sprawling areas, to strict regulations and favourable tax policies, to investments in public transit (see Litman (2009)).

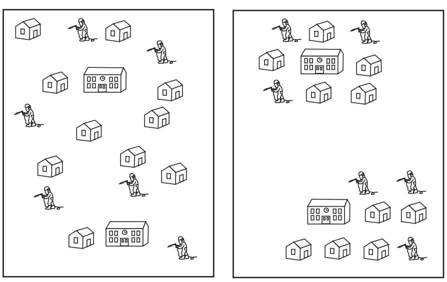

Figure 4.5.1 Urban sprawl versus smart growth land-use patterns. Overall density of housing, employment, and services is the same, but can be patterned in different ways. On the left side this density is dispersed (sprawling); on the right it is clustered and ordered so as to functionally support each other, as in 'compact cities'. Modified and adopted from Litman (2009)

While the SG framework predominantly emphasizes social benefits, policy-makers often point to environmental benefits to justify city compactions. Localization of new development within already developed areas, in-fill development, and redevelopment of brownfields are often claimed as efficient ways for land-use consumption. Such land-use developments are said to spare more remotely located ecosystems from being transformed into the urban fringe development that threatens prime farmlands, wetlands, and unique wildlife habitat (Litman 2009).

The SG framework is also often labelled as a land-use policy for combatting climate change (Chatterjee 2009). While compact growth has been shown to improve long-term air quality at a geographic scale, compatible with secondary pollution formation and transport due to reduced vehicle travel (Stone *et al*. 2007), very few studies have looked into this relationship at more depth. Even though smart growth may reduce greenhouse gas emissions in metropolitan areas, this proposition needs to be further examined. In a rigorous overview study by Susan Handy (2005), exploring how well the available evidence supports SG planning propositions about the relationships between transport and land-use, she concludes that these questions have not yet been fully resolved and that the ability to predict the impacts of smart growth policies still remains limited.

SG land-use development as a planning policy has also been questioned in terms of its feasibility and acceptability to the communities affected. Breheny (1997) asks whether compaction can be acceptable to the communities affected by it and questions to what degree high levels of urban compaction can be achieved in practice. For one thing there are limited brownfield and in-fill sites for housing development and these are also associated with high construction costs. Moreover, Breheny also doubts whether it is feasible to assume that companies will be persuaded to return to city core areas they once have abandoned. A recent study by Filion (2009) shows that high-density multifunctional development nodes in the Toronto metropolitan region have had limited capacity over recent years to attract new office and retail development, as well as meeting walking and public transit patronage objectives. Filion (2009) asserts that high-density suburban node developments are vulnerable to economic recessions.

4.5.5 Generation of urban ecosystem services

Recent urban ecological research reveals that the generation of many types of ecosystem services depends on locally managed green areas and the willingness of their stewards to manage and sustain these habitats (Colding *et al*. 2006). Such habitats predominantly represent semi-natural ecosystems, that is, ecosystems with a high degree of human impact that are managed by so called ‘green-area user groups’ by way of informal institutions (Colding *et al*. 2006). Green-area user groups involve groups and landholders that manage land individually or in cooperative form, such as in associations, clubs, or similar organizational units (Barthel *et al*. 2005; Elmqvist *et al*. 2004, 2008), and where management is based on leisure and recreational activities or conducted through civil interaction, rather than based upon profession (Ernstsson *et al*. 2009).

Semi-natural ecosystems have until recently been regarded as having little ecological value by ecologists, and have therefore been largely ignored in ecological inventories. They have therefore seldom been integrated in conservation planning frameworks. A good example of semi-natural ecosystems are private domestic gardens that often cover extensive parts of the landscape in urban settings and often form quite coherent greenbelts in cities suitable for migration and species movement (see e.g. Jeffcote 1993; Gaston *et al*. 2005; Colding *et al*. 2006). In a study of greater Stockholm, Sweden, Colding *et al*. (2006) showed that over 80 per cent of the green cover of suburban real estate (i.e. ‘low-building’ and ‘part-time’ summer houses) consists of garden cover when all buildings and impermeable surfaces were deducted. Furthermore, allotment areas, domestic gardens, and golf courses provided numerous ecosystem services (Table 4.5.1.) and covered nearly 18 per cent of the surveyed land area, representing well over twice the area covered by protected areas and over half of the land demarcated as green wedges. Studies of other cities reveal a similar pattern, with domestic gardens covering as much as 23 per cent of the land area in Sheffield (Gaston *et al*. 2005) and as much as 27 per cent in Leicester, England (Jeffcote 1993). These semi-natural urban landforms also house a considerably rich number of flora and fauna, including rare and threatened examples (see e.g. Maurer *et al*. 2000; Thompson e*t al*. 2003).

Smaller habitat parcels, like allotment areas, community gardens, and cemeteries, have also been found to sustain considerable pollinator and invertebrate diversity in cities and contribute to the maintenance of ecosystem services like pollination, seed dispersal, and insect-pest regulation (Cane 2001; Colding *et al*. 2006; Colding in press; Anderson *et al*. 2007). Urban golf courses provide important habitats for species in urban settings. In greater Stockholm, over a quarter of all available permanent, freshwater ponds are located on golf courses and contribute to sustaining metapopulations of both threatened amphibians and macroinvertebrates (Colding *et al*. 2009). More broadly, urban golf courses have been shown to positively contribute to biodiversity and, if managed appropriately, provide a measure for restoring and enhancing biodiversity in urban settings and promoting critical ecosystem services, like species migration, pollination, natural pest control, and even water purification (Colding & Folke 2009). Educational institutions may sometimes harbour the largest and last remaining green areas in highly urban developed settings and can be extremely significant in terms of biodiversity (e.g. Patwardhan *et al*. 2001).

4.5.6 The simplification of the urban landscape

Land-cover maps often classify biological patches in built-up areas as physical ones, thereby simplifying the fine-grained texture of the urban landscape

Table 4.5.1 Potential ecosystem services provided by three semi-natural green areas in Stockholm, Sweden. Modified from: Colding *et al*. 2006

	Allotment areas	Domestic gardens	Golf courses
Provisioning services			
Fire wood		X*	
Food (fruits & vegetables)	X*	X*	
Ornamental resources (flowers)	X*	X*	
Cultural services			
Aesthetic values	X	X	X
Inspiration	X*	X*	X*
Nature education	X*	X*	X
Recreation	X*	X*	X*
Social relations	X*	X	X*
Regulating services			
Air filtration	X	X	X
Erosion regulation	X	X	X
Noise reduction	X	X	X
Nutrient retention (in ponds)			X
Pest regulation	X	X	X
Regulation of microclimate	X	X	X
Surface water drainage	X*	X*	X*
Supporting services			
Habitat for flora & fauna	X*	X*	X*
Soil formation	X	X	X
Seed dispersal	X	X	X
Pollination	X*	X*	X
Water cycling	X	X	X

* Indicates accentuated service in respective land-use.

and the small-scale pattern of juxtaposed semi-natural land-uses. For example, much semi-natural land, including all domestic gardens are classified as 'built-up' lands in the official land-classification statistics of Sweden (Colding *et al*. 2006)—a situation mirrored in most city-regions of the world (Foresman *et al*. 1997). Such oversimplified land classification results in cities assumed to have less green cover than actually is the case. Furthermore, it discounts the essential landscape processes that are linked to semi-natural land-uses in suburban areas and which contribute to the support of organism groups and populations in larger natural habitats like nature reserves due 'ecological land-use complementation' (Fig. 4.5.2). Such oversimplification also shapes peoples' attitudes toward urban lands that results in an unfortunate divide between urban areas for biodiversity conservation and areas used for other purposes, when, in reality, species utilize a number of different habitats in the urban landscape (Melles *et al*. 2003). This helps create impressions of the urban landscape being 'binary', consisting merely of developed and undeveloped natural lands.

Recent studies show opportunities to improve intensively managed landscapes, for example urban and agricultural areas dominated by human activities, through greater engagement of ecologists and other urban practitioners in the process of ecological landscape design (Lovell & Johnston 2008). Such an approach encourages the exploration of multifunctional solutions to meet the demands of growing populations, while minimizing the negative impacts of human activities on the environment (Lovell & Johnston 2008). One critical objective in the creation of such multifunctional landscapes is to increase heterogeneity in the spatial pattern of the landscape itself (ibid) such as through ecological

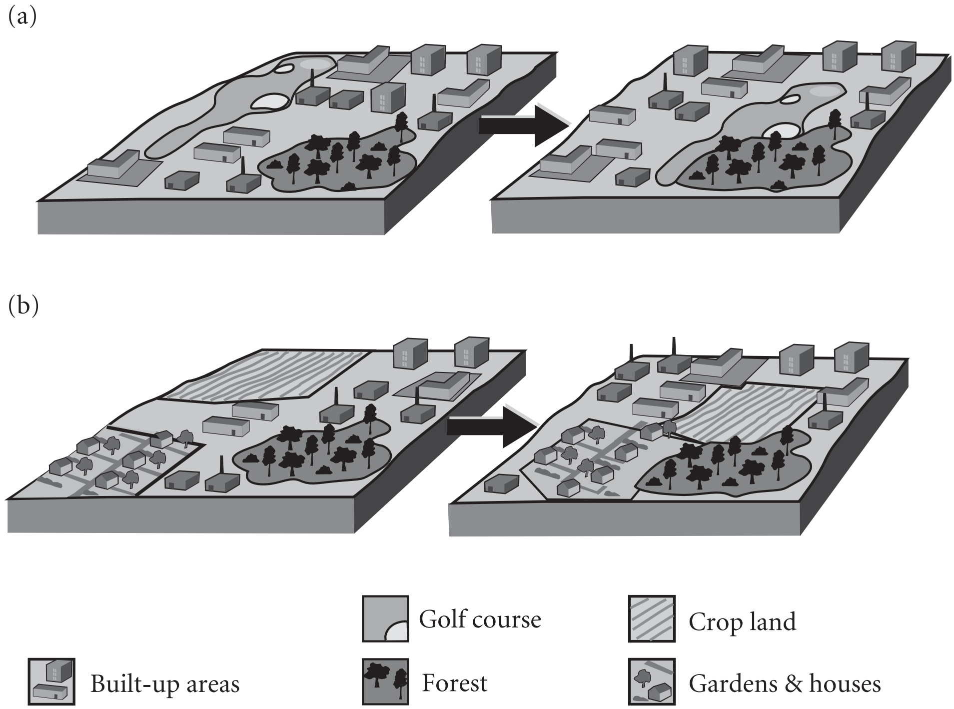

Figure 4.5.2 By adopting 'ecological land-use complementation' in urban spatial design, planners can promote ecosystem services. Situation (a) pictures a golf course with freshwater ponds that is located adjacent to a forested area. This design has greater potential to promote amphibians, relative to a design when the golf course is located in isolation surrounded by urban built-up land. In this sense, the golf course and forested area complement each other, providing necessary habitats for amphibians to breed, forage, and over-winter. Similarly, in (b) when urban gardens are clustered adjacent to forest patches and crop fields, pollinators can be promoted. In situation (b), pollinators use gardens for collecting pollen and nectar resources, use adjacent forest habitats as nesting sites, and perform important pollination of food cultivars on adjacent crop fields. Such design promotes 'response diversity' to environmental stresses among different species of pollinators. Modified and adopted from Colding (2007)

land-use complementation (Colding 2007). By this token the developed matrix in the urban landscape holds considerable potential to be designed and configured in ways that improve conditions for both humans and other biological organism groups. This, however, requires that semi-natural green-spaces in a multitude of forms be preserved and nurtured in developed areas.

There exists a multitude of ways for designing both existing and new green sites in compact cities to provide a range of ecosystem services (see e.g. Jim 2004). For example, ecological engineering, which operates at the interface between technology and environment, can be explored to design and restore ecosystems. Mitsch & Jorgensen (1989) emphasize that ecological engineering is about designing societal services such that they benefit society and nature, and that such designs be system-based and integrate society with its natural environment. Ecological engineering plays a critical role in the restoration of ecosystems that have been substantially disturbed by human activities such as environmental pollution or land disturbance.

4.5.7 Implications of smart growth and green infrastructure planning

Both the green infrastructure and smart growth planning paradigms can be argued to reinforce the stereotypic view that developed urban landscapes consist of lands of low ecological value, that is, the 'built' environment, and that nature is best preserved near and beyond the urban fringe. The smart growth planning paradigm does this on the grounds that compaction reduces per capita land consumption. I have here, however, highlighted the fact that much urban and suburban land-use positively contributes to the generation of ecosystem services and that opportunity exists to improve intensively managed landscapes to further bolster this potential. However, compaction severely reduces the opportunity for people to directly engage with nature. In the SG planning paradigm, a greater fraction of the urban populace will inevitable become 'landless' due to the fact that residents will increasingly come to live in multi-family dwellings without associated gardens. Hence, much of the urban population is removed from the opportunity to own land, to have an input into the use of urban land, and arguably, the ability to understand or have concern for land conservation outside and inside cities (Kendle & Forbes 1997). It will also be extremely hard to encourage the existence of smaller parcels of semi-natural land, such as allotment areas, community gardens, or other forms of urban agriculture, in a compact development scenario. Sharing the concerns of Breheny (1997) regarding feasibility and acceptability, it will be ethically difficult to exclude non-gardeners in such property rights arrangements. It would also be difficult for governments to subsidize such land-uses through public means as is presently done in some countries.

While public greenspaces may exist in densely populated areas, certain dimensions to public domains are particularly important for understanding the enclosure behind public lands in cities (Lee & Webster 2006; Colding in press). One is 'congestion', referring to the degree of competition within a public domain, or the numbers of individuals who jointly consume it, and the range of tastes amongst those individuals (or groups) (Lee & Webster 2006). When public domains become congested, they need governing in such a way that use rights become clear and enforceable; however, to design, create, and administer such a system of rights is costly in terms of transaction costs (Colding in press). When congestion generates excessive costs there is likely to be pressure to reform property rights and subdivide the public domain either into private domains or smaller public domains (e.g. club goods).

Another dimension of public domains is the 'separation of attributes' which is likely to be established if it is cost effective and sufficient demand exists for it. For example, the rights regarding the different attributes of a park can be separated and allocated to various groups of consumers, such as recreational space for sports, habitats for wildlife, restaurants etc. In a congested public domain, markets and governments will strive towards a separation of these rights according to different attributes (Lee & Webster 2006). A telling example of the separation of attributes is public parks that have degraded due to underfunding in Stockholm city during economic recessions. In conjunction with restoration of these parks, local government agencies open up several types of private establishment, such as cafés, amusement areas, etc., to finance the costs of restoration. This inevitably results in the successive loss of greenspace.

The green infrastructure planning paradigm arguably also discounts the role that semi-natural land-use may play in cities. It typically focuses on preserving larger tracts of land beyond the urban fringe, and the designation of land for nature preserves or greenbelts. The emphasis on linear corridors as links in such habitat networks hardly ever considers cohesive domestic garden belts, nor golf courses, although they often represent biological assets in urban settings (Colding & Folke 2009). Suffice to say, this planning paradigm has been shaped by insights derived from ecological studies of more pristine ecosystems and the island biography theory.

The designation of protected areas, green belts, and green wedges has so far been the central tenet in green infrastructure planning. Protected area management is, however, costly for most governments both in terms of land acquisition and management (Colding *et al*. 2006). Many parts of London's protected green belt have been severely degraded due to lack of money partitioned for management, and as a result have been largely avoided for recreation by local inhabitants (Greater London Authority 2001). In response to this, local boroughs initiated the creation of community forests that encompass several hundred hectares of greenbelt land for residents living nearby to manage and maintain.

It is important to note that informal management of urban green areas for the most part bear their own cost in terms of management expenditure and transaction costs. Such management is mainly voluntarily based. In contrast, most parks and protected areas are managed through public means, with government expenditures, making such management more vulnerable to economic recession and political power shifts (Colding *et al*. 2006).

4.5.8 The pedagogic role of nature in cities

Insights developed in environmental psychology show that ecologically impoverished metropolitan areas contribute to increased 'environmental generational amnesia' among city dwellers (Miller 2005). People that are not interacting with ecosystems early and regularly are less likely to support necessary environmental efforts in society (Kaplan *et al*. 1998), such as economic measures to combat climate change. Hence, from a planning perspective, it is important to consider the pedagogic responsibility that cities hold for mitigating further ecological illiteracy among urban populations. In this context many locally managed, semi-natural ecosystems represent important 'learning arenas' for ecosystem services, as they provide visible and measurable examples of human interaction with dynamic ecosystems (cf. Grimm *et al*. 2008). To gain the much needed, broad-based public support for a sustainable use of ecosystems, inside and outside cities, the places where people live and work need also to offer opportunities for meaningful interactions with functioning ecosystems (Miller 2005). In this respect, and in order to help mitigate the growing disconnection of urban residents from nature (Pyle 1978), novel property right designs and land-management approaches, of which some have been dealt with here, need to be developed that may foster the pedagogic role that engagement with nature can foster.

A number of environmental organizations and neighbourhood associations are demanding a voice in zoning and planning decisions that affect their communities (McDaniel & Alley 2005); however, few approaches exist for their inclusion in biodiversity management activities. Research shows that active land management can contribute to promoting understanding about the feedback links between ecosystems and people, by increasing environmental knowledge among urban populations where such knowledge tends to be low (McDaniel & Alley 2005; McKinney 2002). Kaplan and Kaplan (1989) argue that people who do not experience nature early and regularly are less likely to develop strong emotional ties that motivate costly conservation efforts in society. Against this background it is relevant to ask whether urban compaction, as advocated by proponents of smart growth planning and many green infrastructure planners, is socially feasible. In relation to this, Fyson (1996) showed that public preference for house types and their locations is very much the opposite of that advocated by proponents of smart growth planning. Also, marketing surveys carried out by housebuilders in the UK reveal a strong preference for houses with gardens, with people being twice as satisfied with their location in rural areas than those in urban/city areas (Breheny 1997). Likewise, overall satisfaction with housing was lowest in urban centres and highest in the most rural areas. While the compaction logic suggests a need to switch from low-density to higher-density houses and flats, attitude surveys suggest that people overwhelmingly prefer houses, particularly those with gardens (Breheny 1997). Even though preferences may be subject to change given new governmental policies and incentives, compact development advocators have a real dilemma in redirecting such preferences. To some people it would be the same as violating strong norms and community sense.

4.5.9 Concluding remarks

While the concept of 'ecosystem services' (coined by Ehrlich and Ehrlich (1981)) is nowadays frequently used in ecology and increasingly so in economic assessments, it is barely found in the urban planning-oriented literature to date. This is likely because only a fraction of ecological studies have been devoted to urban systems (Collins *et al*. 2000). Furthermore, there exists a time-lag before new ideas are 'filtered' into planning processes. However, as suggested in this chapter, there exist strong motives for integrating a wider set of urban residents into management of ecosystem services. Against this background, it may be unwise for planners to rely too strongly on one, single planning strategy for improving urban sustainability. Both the two major planning strategies reviewed in this chapter propose compact development growth to combat the adverse effects of urban sprawl. However, and as shown here, many types of ecosystem services are generated in the developed landscape, and also in sprawling, suburban settings. Moreover, the significance of creating meaningful opportunity for a greater populace of urban residents is important to combat ecological illiteracy among burgeoning urban populations. Not least to build incentives for measures that involve potentially high costs for avoiding biodiversity loss and unwanted effects due to climate change. Such a meaningful opportunity should also be provided for in land-uses of the urban, developed matrix. As discussed here, the urban matrix is often more heterogeneous than often recognized, contributing to habitat diversity and thereby increasing landscape diversity.

Many urban dwellers prefer to live in locations and houses with access to gardens. Hence, it is hard to justify the compact growth scenario in the smart growth and green infrastructure planning paradigms. While human activity destroys ecosystems at an alarming rate, people are also important for generating and sustaining ecosystem services. More studies need to be conducted to assess whether the environmental benefits of planning compact cities overweigh those of dispersed settlement growth. Until we gain more knowledge, a desirable planning strategy would be to foster approaches and urban designs that qualitatively improve the urban landscape. How such a landscape should ideally be designed to more efficiently preserve the urban landscape and its associated services is a pressing issue. I agree with the proponents of compact growth that the pattern of urban growth needs to be redirected. However, displacing community residents and steering population growth into the labyrinths of dense city cores, with little access to natural habitats, by way of top-down planning initiatives, seems utterly old-fashioned. A more viable strategy would instead be to nurture and support the civic initiatives to self-organize around various urban activities, such as around urban agriculture, community gardening, and partaking in the design and management of urban parklands. Planners should also facilitate for integrating 'urban commons' into comprehensive planning (Colding, in press). Such common property systems are rare in cities presently, but hold the potential to contribute to sustaining and greening the world's cities.

As comprehensively described by Jim (2004), there exist a whole arsenal of green designs that can be adopted in compact cities to provide a range of ecosystem services. Such activities will likely boost social capital, promote sense of place, and revitalize degraded neighbourhoods. This will likely also create more attractive cities that foster environmental stewardship, provide new job opportunities, and contribute to an overall economic development of cities that is more in tune with the life-supporting processes that sustain life itself. We are only at the beginning of developing strategies and tools to more sustainably govern ecosystem services in cities, and there exists a gap in knowledge (MA 2005). Policy-makers, urban designers, and planners need therefore to facilitate to a larger extent more experimental designs in cities. Such strategies may gain from insights on adaptive comanagement of ecosystems, and lessons provided from sustainable natural resource management, to increase the potential for adaptive learning and to avoid the vulnerability traps during the process of urban redevelopment.

Acknowledgements

This research was funded by the Formas Urban Net Program, the Swedish research council Formas. Thanks also to Mistra (the Foundation for Strategic Environmental Research) for support to the Stockholm Resilience Centre.

Summary

Thomas Elmqvist

The chapters in this section examined how the constantly evolving urban landscapes around the globe form complex mosaic of human modifications, metabolic flows, networks, and built structures. These chapters contribute to our knowledge of how urban ecosystems work, how they change, and what limits their performance. This adds to the understanding of ecosystem change and governance in an ever more human-dominated world. Today, cities are facing enormous challenges and urban ecosystems may have a large role in facilitating a transformation to a future beyond fossil fuels and reducing impacts of climate change. Ecosystems provide flexibility in urban landscapes and help build adaptive capacity to cope with, increased temperature and changing precipitation, for example, through multiple ecosystem services that promote human well-being. The concept of ecosystem services has proved useful in describing how biodiversity and ecosystems are linked to human wellbeing. Such services to urban residents include benefits obtained from regulation of ecosystem processes, like air and water filtration, non-material benefits obtained from ecosystems, like spiritual enrichment, cognitive development, recreation, and aesthetic experiences. Specifically the importance of wetlands in the urban landscape is highlighted and the often large benefits generated from restoring degraded wetlands and water systems. Ecosystem services are often bundled, and a specific patch of urban vegetation may simultaneously contribute to reduced air pollution, reduced noise, and improved general perception of health, and are of importance in reducing impacts related to climate change.

To maintain these services in the face of uncertainty and change requires a deeper understanding of the resilience of urban systems. Enhancing resilience of a system often means encouraging flexibility and adaptive capacity in the forms of redundancy, inclusiveness, monitoring, and preparedness for multiple futures. In the urban landscape the general principles of adaptive governance (Folke *et al*. 2005) need to be translated into a set of methodologies, discourses, and planning tools that recognize: (1) in an urban and urbanizing planet, cultural and biological diversity is key to resilience of social, economic, and ecological systems, (2) knowledge (either scientific or local) is key to management, (3) education is the main conduit for mainstreaming an empowering communities, and (4) adaptive management is not only a matter of building flexible institutions and governance systems at the relevant scale, it is also dependent on equity and particularly equitable access to land and resources. Adaptive governance has to merge with developments in urban planning such as smart growth and green infrastructure planning. Policy-makers, urban designers, and planners need to facilitate to a larger extent more experimental designs in cities, and gain from insights on adaptive co-management of ecosystems and lessons provided from sustainable natural resource management, to increase the potential for adaptive learning and avoid the vulnerability traps during the process of urban redevelopment.

The chapters in this section have the aim of contributing to our understanding of urban ecosystem services, urban resilience, and the pathways to sustainable urban development. They generally point to the need of placing human well-being at the core, breaking the artificial and largely culturally biased divide between the pristine and the human-dominated ecosystems, and contributing to the creation of a new language, with signs, concepts, words, tools, and institutions that gather rather than divide, broker conflicts rather than create them, and establish responsible environmental stewardship at the heart of public interest.

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SECTION 5

Urban Design, Planning, and Management: Lessons from Ecology

SECTION EDITOR: **Philip James**

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Introduction

Philip James

What challenges do urbanites face in the early twenty-first century? What contribution does a knowledge and understanding of urban ecology make in addressing these challenges? What might the landscapes of cities in the near and mid-term future be like? These are the central questions addressed in Section 5.

Urban ecology has been defined at two levels. Narrowly, it is concerned with the organisms, habitats, and ecosystems that are found within a city. More broadly urban ecology integrates various areas of science, the humanities, and arts in the planning of urban areas to improve the living and working conditions within cities (Wittig 2009). This Section takes the broader view; elsewhere in this book this is described as 'ecology of the city'.

Background

As I write this introduction, the UK's Royal Geographical Society in hosting a series of events under the banner '21st Century Challenges'. The first of these events, held in October 2007, brought Sir Bob Geldof and Kofi Annan together to discuss 'Africa in the 21st century'. Since then, the series has turned its attention to a range of issues including climate change, migration, energy resources, environmental hazards, food production, water resources, the future of the countryside, and the impact of globalization on developing countries.

In 2008–9, The Institution of Civil Engineers' Brunel International Lectures were delivered by Peter Head. The theme of these lectures was 'Entering the Ecological Age' (Head 2008). Associated with these lectures was a series of videos illustrating 'the ecological age'. The videos demonstrate how ecological features might be retro-fitted into city centres, suburban areas, and slums. These inspirational scenarios show how food can be grown (using roofs, gardens, and public spaces), energy produced (wind turbines, photocells) and cities cooled (increased vegetation), thus addressing two of the great challenges that face us in the twenty-first century. The visions, as would be expected from the title of the lecture series, are based on harnessing ecological knowledge.

In a report commissioned by the Austrian Federal Ministry of Agriculture, Forestry, Environment and Water Management (Lebensministerium 2004), a number of 'mega-trends' that characterize the development of urban areas across large territories were identified. Globalization is one such 'mega-trend.' Two important affects of globalization are increasing spatial division of labour and economies of scale in the international economy. These trends currently over-ride transport costs. Transport of goods and mobility of people continues to grow annually, adding to the pollution of the global environment, the depletion of fossil fuels, and congestion in urban areas and of major transport routes. New rural areas are opened up to urban development as transport infrastructures develop in response to economic pressures, resulting, in turn, in increased mobility and accessibility. The demand for easing long distance and international flow of goods currently over-rides local sustainability needs. Advances in information and communication technology (ICT) are resulting in the emergence of new, more polycentric, patterns of urban development. Demographic trends, specifically an ageing population and the growth of smaller and single person households, are adding to the demands for new housing and to pressures for suburbanization in rural areas and the requirements to improve the quality of the environment in inner city areas. Agricultural land, amenity space, and nature reserves are being lost, and at the same time increasing land values and property prices in cities make housing in areas which are accessible to employment opportunities and to services increasingly unaffordable for many sections of society. The decline of industrial activity in many cities has resulted in the need to invest in the regeneration of brown field sites and declining inner-city neighbourhoods. The concentration of social problems in inner-city neighbourhoods, and in isolated or over-sized public housing estates, increases problems of crime and personal security leading to further spatial segregation and the 'flight of the middle classes' to protected suburban enclaves. Growing prosperity and wealth and demands for an improved quality of life are reflected in the increasing consumption of land and space, demand for privacy and better living conditions, and access to green space.

Rates of ill health associated with obesity (e.g. diabetes, heart diseases, and cancer) continue to rise. Doulas Smallwood, Chief Executive of Diabetes UK is quoted as stating 'Research shows that losing weight can reduce the risk of developing type 2 diabetes by 58%. It is imperative that we raise awareness of the importance of eating a healthy, balanced diet and doing at least 30 minutes of physical activity a day if we want to make any headway in defusing the diabetes time bomb' (BBC 2009). The challenge here is clear: to design urban spaces that are conducive to people taking more exercise and provide the space for the provision of healthy food. The use of ecological knowledge in addressing the challenge of lifestyle is clear.

The effects of climate change are far reaching and will influence settlements, human health, and the distribution of species. Evidence reported by the IPCC (2007) suggests that the most vulnerable industries, settlements, and societies are those in coastal and river floodplains, those whose economies are closely linked with climate-sensitive resources, and those in areas prone to extreme weather events, especially where urbanization is occurring. The IPCC also report that there is an increased likelihood of malnutrition, diarrhoeal disease, cardio-vascular disease, and increased death, disease, and injury due to heatwaves, floods, storms, fires and droughts. The distribution of some infectious disease vectors will also change. However, in temperate areas, deaths due to exposure to cold may decrease. Evidence also suggests that species are moving northward. Climate change, if seen as being driven by a redistribution of carbon with an increased percentage found in the atmosphere, is an ecological problem: it is based on the cycling of material. Hence, ecology needs to be incorporated both into the science that addresses the causes of climate change and into the technologies (both hard and soft) that are being developed to mitigate the effects of climate change.

Developers, planners, and politicians are at the sharp end of meeting the challenges the myriad of challenges we face, and it is members of these professions who will map out a course for cities into the second half of the twenty-first century. However, those in these professions need to be equipped with sufficient ecological knowledge and understanding to steer an appropriate route into what Peter Head called 'the Ecological Age'. Section 5 provides that guidance, illustrating how knowledge of ecology has already been, and how it can be, incorporated into city planning and management.

Ian Douglas and Joe Ravetz present an overview of ecology of the city taking a holistic view of the relationship between ecological and social and cultural features present within urban environments. Within this chapter a 'framework for urban ecology' is presented. Crosscutting themes are identified, A series of examples of good practice and successful projects are set out. These provide clear guidance for all involved in planning and designing the development of other cities and towns. Green infrastructure is identified as a key development and the chapter concludes with a set of best practice guidelines for green infrastructure planning.

The health and well-being of those living and working in urban areas is a paramount concern. Kosta Tzoulas and Kim Greening provide an overview of the role that urban greenspace can play in public health approaches. Such activities aim at creating the broad social and environmental conditions that promote human health and well-being.

These two authors discuss how urban ecosystems can make a significant contribution to public health. And point out that understanding the dynamic interaction between environmental, ecological, and social systems in urban areas is key to the understanding of public health determinants. Within the chapter the authors provide a typology of public health activities across three spatial levels, from policy to individual practice. The contributions of urban greenspace to the social well-being and to the physical and psychological health of individuals are addressed in separate sections in this chapter. Overall this chapter highlights how urban ecological knowledge, if integrated into the planning, design, and management of cities and of healthcare systems, can play a principal role in improving the physical, biological, and social urban environments.

Stephan Pauleit and his co-authors review the main principles of green infrastructure planning, and explore whether and how these are related to urban ecological theory. They go on to present two case studies (Seattle, Washington, USA and Greater Manchester, UK) which allow them to study green infrastructure planning in practice. In presenting these case studies, the authors of this chapter demonstrate how high stormwater flows and increasing temperatures associated with climate change can be mitigated by using ecological principles. The authors also discuss work in Greater Manchester on the development of the Ecological Framework that seeks to provide an overarching structure within which specific conservation orientated interventions can take place.

Jon Sadler and his co-authors review the science underpinning the management of urban systems, the planning systems and policies that exist to help manage urban areas, and mitigation techniques and tools used to compensate for habitats lost to urban development. These authors point out that there are multiple social and institutional barriers to building for biodiversity in cities because they were constructed without biodiversity in mind. Recent understanding of the value of green infrastructure has led to an increasing realization that greenspaces are not only valuable for plants and animals, but also for people, and that to fully benefit from linked habitat networks and ecosystem services cities must be managed at large spatial scales. They recognize and state that the development of truly ecologically sustainable cities requires the wide-scale alteration of physical habitat, legislation, and cultural perceptions and is therefore an enormously challenging problem that seems a distant and ambitious goal.

One of the linking themes of this Section is the inter-dependency of social and ecological systems. This theme is developed by Zipperer and co-authors, who explore and evaluate current models that link these two systems. The application of the concept of complex adaptive systems to urban landscapes is reviewed and a socio-ecological model based on complex adaptive systems and structuration theory is presented. The chapter concludes by suggesting two metrics, sense of place and land cover, as integrators between ecological and social systems.

John Box considers mechanisms for enhancing biodiversity within cities. Importantly he points out that the benefits of biodiversity are most relevant at the community level and hence there are difficulties in translating these benefits into direct financial benefits to individuals and to businesses. Taxation and financial savings are discussed as mechanisms that can offer financial incentives, but difficulties with these models are described. Three drivers for change are identified: a) economic benefits directly affecting businesses and people; b) legislation, regulation, and official guidance; and c) using targets to encourage those individuals and organizations pioneering change by providing evidence of tangible achievements to change behaviour and overcome cultural norms. These drivers imply a necessary change in human behaviour. The likelihood of these changes is discussed.

Taken together, these chapters, by describing case studies from across the world, clearly demonstrate how ecological knowledge and understanding is being used to shape our cities and towns. The authors also identify significant challenges that must be faced if we are truly to enter ‘the ecological age’.

CHAPTER 5.1

Urban Ecology—The Bigger Picture

Ian Douglas and Joe Ravetz

5.1.1 Introduction

This chapter takes a holistic, integrated view of the interplay of natural, ecological, social, and cultural processes, and how this creates the great variety of plant and animal assemblages occupying urban greenspaces, and determines the benefits that those assemblages provide for both individuals and communities, and biophysical, and ecological systems. It also shows that although we are relatively ineffective in securing as many of these benefits as we could, there are a host of examples of good practice and successful projects. Furthermore, this chapter reminds the reader that most of the world's urban greenspace users in the twenty-first century are likely to be living in poor circumstances, where the urban ecosystems and wildlife that matter most to them are urban food production systems, the feral dogs that eat the rubbish dumped near their dwellings, the microbes and other organisms in their drinking water, and the disease vectors that threaten their children with malaria and dengue fever.

So, in this chapter we outline a framework for the human dimension of urban ecology. First, we look at the fundamentals of urban ecology, as relationships between ecosystems and urban forms and activities. Then we look at the global dimension, with some critical perspectives on globalization and other crosscutting themes: this provides insight into the role of urban ecology in relationships of power and ideology. Thirdly, we look at social and economic functions, and linkages between people, communities, and their urban ecosystems. This leads into the practical issues of design, planning, funding, and management: and finally a set of guidelines for best practice in green infrastructure planning.

5.1.2 A wider framework

Clearly there is more to urban ecology than parks and gardens. We can identify a 'hierarchy of urban ecologies', with a broad view of the relationships between urban systems and ecosystems (Roberts *et al.* 2009):

- Ecology *in* the city—ecosystems within the urban form and fabric: particularly the unique urban or peri-urban habitats and biodiversity.
- Ecology *of* the physical urban system: this includes a wide range of environmental patterns, and physical flows, in and through the city, and through urban systems on a regional, national, or global scale.
- Ecology *of* the social-economic urban system: a further range of human activities which revolve in some way around ecosystems and environmental processes. This includes industrial ecology, ecological design, ecosystems markets, and so on.
- Ecology *throughout* the human–environment system—social, economic, cultural, political patterns, and relationships, in the 'human ecology' sense of a community of interdependent processes in both human systems and ecosystems.

This broad overview can be explored in many directions. For instance, the definition of 'urban' is a moving picture with many angles. We can explore layers in time and space, where global long-term processes have local short-term effects, and vice versa. We can explore human processes, into areas such as culture or ideology, which can often drive more tangible impacts such as land-use and food demand. Above all, the urban ecology concept is about relationships at the system level: this is essential to understanding global pressures and crisis points, as much as local urban greening (see also McDonald and Marcotullio, Chapter 4.1, Alfsen *et al*., Chapter 4.3).

5.1.2.1 Global perspectives

Urban ecologies can be divided crudely into those in 'developing' and 'developed' worlds—the 'South' and the 'North' (see also Alfsen *et al*., Chapter 4.3). In very simple terms, the early twenty-first century developing world includes the 80 per cent of the population who consume 20 per cent of world resources: and the developed world includes the 20 per cent—about 1.3 billion people—who consume the 80 per cent remainder. In 2009, twice as many people live in cities of the 'South' than in those of the 'North'. By 2030 there may be four times as many. The future may indeed be a 'planet of slums' (Davis 2005; Neuwirth 2006).

For many in the developing world, cities are often places of poverty, disease, and struggles for power and resources—where the 'environment' is represented by polluted water, deforestation, or fear of landslides and floods. In contrast, for many, the developed-world city can be a place of conspicuous consumption, leisure and culture, illusions and icons—where the environment is a commodity to be bought and sold, or seen from a car windscreen or tourist hotel.

In both worlds, the urban environment is clearly a 'biophysical' system of water, air, land-use, energy flows, and biological elements. It is also a 'human' system, with complex patterns and processes, which might be explained by economics, sociology, political science, psychology, and culture. The causes, the impacts, and the solutions for biophysical problems are generally found in these human processes and pressures.

Generally, the more affluent cities of the North import increasing amounts of materials and products from countries with less regulation and lower wage costs. Despite more organized infrastructure, the rate of per capita consumption and the use of new and untested technologies in the North still produces environmental problems such as transport emissions, endocrine disrupters, biomedical waste, and food chain accumulation of heavy metal toxins. Meanwhile, standards and expectations continue to rise, as environmental quality is linked to economic competitiveness, property values, and social well-being (Ravetz 2006).

In contrast, the less affluent cities of the South are typically driven by global markets to the lowest production cost and environmental standards, containing hazardous and highly polluting industries. Their urban areas often have inadequate public services and infrastructure and are unable to cope with constant migration from rural areas. They thus have growing unplanned and un-serviced settlements, in locations with high risks of flooding, landslides, and contamination (Hardoy *et al*. 2001).

In reality, all cities contain both affluence and poverty in varying proportions; in particular the rise of the middle classes in emerging nations such as India and China is changing the picture rapidly. However, there is a structural difference between opposite ends of the spectrum. At one end is a typical North American city with, in 2005, a per capita income of $35,000, and an 'ecological footprint' of over 10 global hectares per person. At the other end is a typical Sub-Saharan city, with a per capita income of less than $350, and a footprint of 0.5 global hectares per person. Ironically, in 2005 the USA was the richest and most unequal country and had the lowest life expectancy of any large developed nation (UN Habitat 2007).

5.1.2.2 Critical perspectives

Behind the details of urban design and greenspace is a much bigger picture. This can be framed as 'critical perspectives'—controversial debates, unfinished agendas, and theoretical battlegrounds (Roberts *et al*. 2009). The critical point of these is that existing structures of power and ideology are not accepted as inevitable and fixed—rather, they may critique and argue for change. Below is a summary shortlist:

- Globalization is a dominant force in cities around the world. It takes economic forms in the structure of business and finance; political forms in the regulation of trade; and cultural forms through media and information and communications technology (ICT). There is also a counterpart strand—'localization'—where the cultural identities of people and places are re-constructed (Sassen 1994; Soja 2000).
- The paradigm of liberalization involves privatization, franchizing, cost recovery, and market segmentation into profit-making and other public services in many cities. Liberalization has been forced on many developing nations and cities through the process of structural adjustment, driven by loans and investment funding packages (Stiglitz 2002; Harvey 2005). It has adverse consequences, as shown by the 2008–09 global financial crisis.
- The consumption and affluence culture is a major driver of cultural identity and community. The result can be new perceptions of urban environments, particularly through leisure and tourism, and the investment or dis-investment which takes place as a result.
- The 'risk society' concept is a powerful way to explain structural changes due to uncertain risks from environmental change and different social groups. This produces both the need for security against crime, civil disorder, and perceived risks of migrants; and the importance of trust between citizens, organizations, and governments (Beck 1995). One response is a defensive partitioning of the cities into gated communities: another is the agenda for the 'liveable city' and the 'sustainable community', with public open space, safe public transport, and a multicultural community (Roberts 2008).
- The political urban ecology agenda starts and ends with poverty and livelihoods, as above. But the problem of poverty is not so much a simple lack of income, but a cumulative causation of stress coming from employment, housing, education, health, as well as environmental factors, adding up a pervasive conditions of 'social exclusion'. In response, there are many solutions in both North and South, from community participation and empowerment, to asset and livelihood approaches, and microfinance schemes (Yunus 2008).
- The social construction of reality, power, and ideology is deeply embedded in the apparently well-intentioned policy agenda of managing urban ecosystems. We can look at the 'discourse' of the ruling 'coalitions' as the organizer of values, objectives, and resources to achieve them, and the provision of urban infrastructure or greenspace is a good example of this (Hajer 2003; Brand & Thomas 2005; Kaika *et al*. 2005).

There are many more perspectives to be explored, elsewhere—the point is that many kinds of human urban ecology questions can be explored and deconstructed with such critiques.

5.1.3 Social and cultural issues

Urban ecosystems influence our mental and physical health and our psychological well-being in complex ways (see Tzoulas and Greening, Chapter 5.2). Particular natural settings may have offered such specific advantages during our evolutionary history for the survival of particular groups such that natural selection favoured individuals who developed and retained positive responses towards those settings (Ulrich 1993; Sadler *et al*., Chapter 5.4). Throughout human history, people have reacted to natural environments in different ways. While some people are excited and challenged by clumps of wild vegetation, others are afraid of what might lurk in those dark, unmanaged areas. Many people find natural, or wild, areas to be unattractive and that they induce negative reactions. Thus a wide variety of individual responses to green areas, both positive and negative, is to be expected. Direct behavioural evidence of such negative reactions is limited because the use of wildlands for recreation is an activity chosen by individuals, and thus those who dislike them avoid them. Behavioural surveys conducted among adult visitors in urban nature inevitably examine an already self-selected group likely to have positive attitudes to wildlife. Students attending compulsory field classes represent a broader range of attitudes. Bixler *et al*. (1994) collected examples of negative reactions by urban students on field trips observed by park naturalists and teachers of environmental science. Some of the attitudes found were generalized fears of the woods; of wildlife; and of insects and spiders; disgust reactions to the dirtiness of the environment; and discomfort from extreme weather conditions.

Urban greenspaces have to meet the needs of diverse social groups, from providing the first contacts with nature for young children, to the tree-climbing and den-building of pre-teens, to teenage adventure, to recreational running and biking for young adults, dog-walking for the middle-aged, and leisure walking for senior citizens. Actual use varies with socio-economic characteristics and access to open spaces and other recreational facilities. Spatial access to attractive, public open space usually leads to increased walking (Giles-Corti & Donovan 2002). Increased attractiveness promotes more open space use. For example, residents in social housing in Kirby, Merseyside, UK made much more use of a large grassed open space when it was planted with wildflowers.

Cultural contrasts in using urban greenspace are marked (Risbeth 2001). Some enjoying peace and quiet, others seeking pleasant places for large, raucous family picnics. Investigations at Lincoln Park on the Lake Michigan foreshore in Chicago showed that people of European descent were more involved than other groups in active individual pursuits such as walking, cycling, and jogging (Gobster 2002). Other ethnic groups dominated those involved in passive, social activities such as picnicking, sightseeing, socializing, and attending festivals and parties.

5.1.3.1 Overlapping multiple benefits across urban areas

The view of urban spaces as meeting the needs of people has long been the dominant theme of the creation and management of vegetated areas in cities (Table 5.1.1). The ancient Egyptian villa gardens and Babylonian walled gardens illustrate how in ancient time the nobility tried to maintain direct contact with nature (Ulrich & Parsons 1992). Now the view of green urban infrastructure for multipurpose benefits incorporates not merely benefits to humans, but a range of biodiversity and environmental values, from insect habitats to flood mitigation. For example, water spaces face multiple competitions between motorized and non-motorized water sports, angling, and habitats for fish, amphibians, and birds. As Canada geese (*Branta canadensis*) illustrate, what is good for one species is not always good for others. Canada geese create problems for wetland site managers by the way their faeces contribute to the eutrophication of water bodies and by their impact on marginal and bankside vegetation through grazing and damage to soil structure (Gosser *et al*. 1997).

5.1.3.2 Ecosystems and their services

The concept of an ‘ecosystem service’ looks at the functional relationships between natural and human systems (Millennium Ecosystems Assessment 2005). Urban ecosystem services (see also Section 4 and Pauleit *et al*., Chapter 5.3, Sadler *et al*., Chapter 5.4) include air filtering (gas regulation), microclimate regulation, noise reduction (disturbance regulation), rainwater drainage (water regulation), sewage treatment (waste treatment), food production, erosion control, biodiversity maintenance, recreation, health, and cultural values (Bolund & Hunhammar 1999) (Fig. 5.1.1 and Table 5.1.2).

In a crowded world, where the globally wealthy minority of people consumes land and other resources at a rate far above the capacity of the world to sustain all people at such a level, the task of managing urban areas to maximize ecosystem services must be to gain multiple benefits from every tract of land. Such efforts require collaboration between a variety of agencies, and a willingness to invest in the long-term future (at least the expected lifetime of a building, or piece of green infrastructure) by land and property owners, tenants, and investors. They also need access to intelligible, readily assimilated scientific information of what will grow where, what kind of substrate is required, what organisms will colonize a given area, and what type of ecological succession can be expected in that area.

Decisions have to be made about sharing urban ecosystems among species, including human beings.

Table 5.1.1 The nature and role of urban parks gardens and forests through time

Urban Development	Leading Actors	Types of garden	Main forms of park	Main urban forest functions
Ancient cities	Rulers and elites	Palace and villa private gardens	Limited public gardens (e.g. in Athens)	Hunting grounds (Persia, Assyria); parts of palace gardens (Rome)
Mediaeval city (political, religious)	Nobility	Gardens in monasteries and large private residences; gardens horticulture within city walls	Private parks around palaces	Hunting, subsistence
Mercantilist and Renaissance City	Nobility and bourgeoisie	Private gardens behind larger urban houses; much vegetable growing	Botanic gardens linked to medical schools	Recreation, prestige, production (for a few)
European Industrial City	Local governments and industrialists	Private gardens for the wealthy, intense peri-urban horticulture	Parks for people	Recreation (for all)
Tropical Colonial City	Colonial officers	Private gardens for the wealthy (colonial elite)	Formal parks, Botanic gradens	Nature conservation
North American city	Democratic city planners	Gardens in urban squares as 'symbols of nature'	Nature in cities; Pleasure grounds	Provision for areas of trees in early city designs
Traditional east Asian cities	Emperors and rulers	Formal symbolic palace gardens	Parks in palace grounds: later opened to public	Glades of trees in palace grounds; sacred forests near temples/ monasteries
Early-twentieth century cities	National and local governments	Garden city suburbs; street tree planting, sprawl greenspaces	Open space systems; multi-purpose facilities; sports grounds	Bird sanctuary and tree protection orders;
Late-twentieth century cities (1970–2000)	National and local governments; civil society organizations	Domestic gardens, municipal gardens; allotments, urban farms in parks	Rejuvenation of old formal parks; urban nature reserves, natural areas; sports grounds giving way to informal recreation	Recreation, nature conservation, environment, landscape, production
Post 2000 cities	National, regional and local governments	Renewal of interest in local and home food production in affluent societies	Promotion of biodiversity and green infrastructure planning and management	Health benefits and ecosystem services emphasized; multi-functional uses promoted
Tropical cities in emerging economies	National and Municipal Governments	Small domestic gardens, high use of plant pots; informal use of land for urban agriculture especially by poor	Formal parks as national symbols; others as pleasure grounds including lakes	Protected forests on urban periphery, or significant topographic features, including coastal mangrove reserves

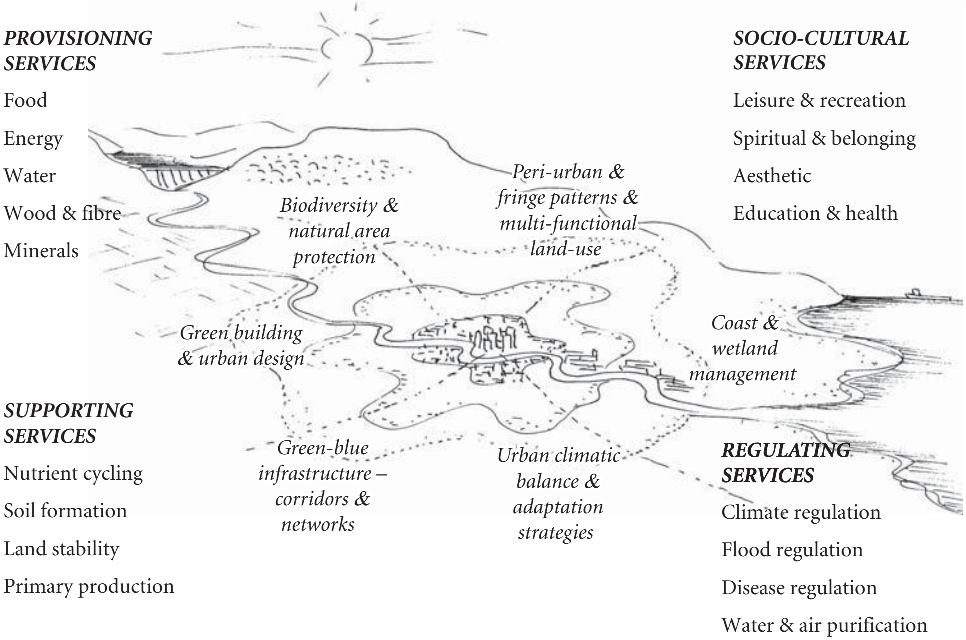

Figure 5.1.1 Urban ecosystem services. Four main types of ecosystem services with urban and peri-urban responses. Source: Millenium Ecosystem Assessment (2005), Ravetz (2000)

Different community groups and institutions have conflicting goals for particular tracts of urban greenspace. Some of these bodies can exert strong influence through ownership and land tenure, but other bodies, from local planning authorities to wildlife trusts, have stewardship responsibilities that can use either legislation or persuasion to protect particular urban ecosystems or habitats. In the range of concerns about urban environmental change, from the battles to save individual inner-city trees to conflicts over new airport runways in urban greenbelts, the multi-faceted character of interplay between institutions and organizations is readily apparent.

Once developments have occurred, ecological changes produce new problems of maintenance of green areas, not only in the face of damage and deterioration through human use, but in the face of invasive species, pests, climatic extremes, and interspecific competition. For example, many restored former brownfield areas have suffered through lack

Table 5.1.2 Urban ecosystem services

- 'Provisioning services'—tangible goods which ecosystems provide directly. This could be fresh water for consumption or production; food for consumption; forest and crop plantations for energy and fibre.
- 'Cultural services'—more intangible experiences which are offered or enabled by ecosystems. Landscapes, uplands, community forests, and urban greenspace are valued for aesthetic and recreational qualities: reservoirs, canals, and urban water courses enable social relations and cultural identity.
- 'Regulating services'—benefits from ecosystems concerning regulation of natural processes. Wetlands, dunes, and floodplains for flood and flow regulation; vegetative cover for erosion regulation; peat bogs for carbon sequestration, are all examples of the regulation functions, which urban development ignores at its peril.
- 'Supporting services'—these underpin the provision of other ecosystem services. Soil formation is essential to other services; wetlands, aquifers, and riparian habitats for water cycling; soil for nutrient cycling.

of maintenance. Elsewhere, growth of trees may threaten houses, through risk of falling branches or subsidence during drought, and health and safety arguments may mean a loss of vegetation.

For some, urban greenspace may have an economic value, enhancing property prices and also potentially offering an economic return, for example through forest product sales from urban forests. Others may be forced to forgo economic gains by demands for greenspace provision in new housing or retail developments that reduce the number of homes built in a certain area, or cut the car parking spaces available. The search for win–win situations suggests that some urban ecosystem service benefits may be gained while also meeting economic demands. Techniques such as green roofs and sustainable urban drainage systems bring benefits to both human residents and urban ecosystems.

Using the categories of ecosystem services employed by the Millennium Assessment (Hassan *et al*. 2005), those provided by a simple set of urban greenspace types (Table 5.1.3) can be readily understood. The main message conveyed by this tabulation is that all types of urban greenspace bring multiple benefits and require planning, management, and maintenance, that is, mechanisms that maximize those various benefits. For example, urban agriculture, whether food production by the poor in deprived areas of tropical cities or fruit growing in affluent suburban gardens, provides not only food, but gives exercise that improves mental and physical health, while also helping to reduce urban heat island effects, absorb carbon dioxide, and allowing rainwater to infiltrate. Care has to be taken that excessive fertilizer applications do not lower groundwater quality and that harvesting does not deplete soil nutrient stocks.

5.1.3.3 Urban geography and the spatial dimension

The urban greenspace agenda involves dealing with spatial levels ranging from the domestic window box to regional-scale green corridors. This multilevel system is now framed as 'green infrastructure' or 'green–blue infrastructure' (see Colding, Chapter 4.5, Pauleit *et al*., Chapter 5.3). This is in line with ecological systems thinking on 'nested holons', that is, coherent systemic behaviour on a multi-scalar axis (Waltner-Toews *et al*. 2009). Experience shows that success at one level is linked to other levels—rather than a set of free standing open spaces, urban green infrastructure works as a connected web of habitats. And such webs are rarely static, but rather continuously changing with the restructuring of the physical city, its economy and industry, its society and lifestyles.

Such restructuring works at each level—regions, conurbation, districts, neighbourhoods, streets, blocks, and dwellings—to form new patterns of urban 'eco-morphology' (Hough 1984) (Fig. 5.1.2). An obvious example is the way in which suburban areas often contain greater biodiversity than the chemically intensive farmland surrounding them, and yet many people still aspire to a house with a view of trees and fields. If the city is to retain its population it must offer such opportunities for ecocystem–human linkage. Urban streets and squares, allotments, country parks, and remote wilderness each have a role to play in this multiscalar web. The edges of each zone are typically the most diverse and active, for both ecological and human interest—so the connectivity and interpenetration of each habitat type with others is crucial.

At the city–region scale, green wedges and fingers of river valleys, railways, and canal corridors tend to be degraded by roads and other infrastructure, so the interface between urbanized and natural landscapes needs several layers of planning. Green corridors along roads and other routes should link the 'stepping stones' of countryside and urban parks in a range of sizes. Different degrees of urbanized cultivation should link new and ancient woodlands, intensive and extensive habitats, and leisure uses.

At the local level, traffic calming of residential areas enables planting of trees in streets and public spaces on a large scale, and reconnects each local centre to its catchment with a detailed network of green routes. These link with green fingers and wedges, surrounding each neighbourhood with stepping stones of open land for allotments and community woodland. One practical problem is the crossing of major roads, where a combination of

Table 5.1.3 The ecosystem functions of different types of urban greenspace

Ecosystem service functions	Regional parks, green grids, greenways	Street trees	Communal and neighbourhood open spaces	Gardens	Green roofs	Sustainable urban drainage sytems	Wetlands	River corridors
Food: Crops	Urban food production		Allotments and urban faming possible	Family food production	Potential for vegetable cultivation			Urban vegetable plots
Livestock	Managed grazing possible			Raising chickens and rabbits for food		Managed grazing possible	Water birds including ducks	Managed grazing possible
Fishing (capture)							Can include recreational fishing ponds	Angling a major participant sport
Aquaculture	Ponds can be incorporated			Garden ponds (usually ornamental)		Ponds could be used for fish	Potential if water quality good	Flood basins could be used
Wild plant and animal food sources	Harvesting of wild fruits and green plants	Provide food for birds		Birds attracted to gardens	Bird habitat		Waterbirds	
Fibre: Timber	Managed urban forests						Possibility of use of willows and similar wetland trees	
Wood fuel	Coppicing provides wood for cooking	Trimmings for composting or fuel		Trimmings for composting or fuel			Trimmings for composting or fuel	
Genetic Resources	Preservation of species lost from farmland	Possible to have diverse species	Wildlife reserves can harbour rare species	Often high biodiversity	Possible to have diverse species	Possible to have diverse species	Wildlife reserves can harbour rare species	Diverse habitats encourage many species
Biochemicals			Can promote preservation of traditional medicinal plants	Can promote preservation of traditional medicinal plants				Potential for varies biochemical production
Freshwater	Water			Rainwater collection on garden sheds and greenhouses	Rainwater collection systems possible	May reduce pollution	Help to cleanse water and delay run-off	Storage and conveyance of water, interaction with groundwater

(*Continued*)

Table 5.1.3 Continued

Ecosystem service functions	Regional parks, green grids, greenways	Street trees	Communal and neighbourhood open spaces	Gardens	Green roofs	Sustainable urban drainage sytems	Wetlands	River corridors
Air quality	Reduces heat island effect, removes some pollutants	Reduces heat island effect, removes some pollutants	Reduces heat island effect, removes some pollutants	Reduces heat island effect, removes some pollutants	Reduces heat island effect, removes some pollutants	Reduces heat island effect,	Reduces heat island effect,	Reduces heat island effect,
Climate	Absorbs CO_2	Absorbs CO_2	Absorbs CO_2	Absorbs CO_2	Absorbs $CO_{2;}$ reduced heat loss to atmosphere	Plants absorb CO_2	Plants absorb CO_2	Plants absorb CO_2
Water regulation	Help reduce some flood flows	Some impact on runoff rate, depending on canopy size and foliage	Aids infiltration to groundwater	Aids infiltration to groundwater	Reduces rainwater discharge	Reduce stormwater peak flows	May act as flood detention ponds	Can include flood detention ponds
Erosion regulation	Good ground cover reduces erosion risk		Good ground cover reduces erosion risk			Reduce gullying and channel erosion, retain sediment	Retain sediment	Restoration can stabilize channels
Water purfication and waste treatment						Helps in cleaning water flowing to streams or groundwater	Can be highly effective in removing pollutants	Can incorporate wetlands
Disease regulation	Removal of forest vegetation can lead to spread of leptospirosis in tropics			Plant diseases can be introduced by importation of exotic species			Risk of dengue fever in tropical cities if drainage blocked	
Pest regulation	Opportunities for natural enemies to provide pest control							
Pollination	Possibility of reducing decline in pollinators							Diverse habitats favour pollinators

Natural hazard regulation	Helps to reduce geophysical hazards, protecting hillsides, coastlines, and river banks							Significant flood management role, especially peak run-off storage
Spiritual and religious values	Possibility of maintaining sacred groves		Gardens in religious buildings and compounds	Water in ceremonial gardens highly important				Some rivers have great religious significance
Aesthetic values	Forests often highly attractive	Street trees improve landscape quality	Parks often have high aesthetic value as an integral part of a city	Gardens and gardening often highly valued by individuals	Improves appearance of buildings			High quality river scenery
Human health benefits	Mental and physical benefits	Mental and physical benefits	Mental and physical benefits	Mental and physical benefits	Mental and physical benefits	Mental and physical benefits	Mental and physical benefits	Mental and physical benefits
Social relations	Group re creational activities		Group re creational activities	Family activities				Group re creational activities
Cultural Heritage	May include historic woodland	Specific trees may have historical associations	May include historic buildings	Classic ornamental gardens part of heritage				May include historic bridges and other structures
Recreation and eco-tourism	Passive recreation	Can enhance tourist enjoyment	Can include sports grounds	Gardening as a relaxing hobby	Relaxation and garden maintenance	Possible to use for sport and passive recreation	Can include areas for water sports	Water recreation possible
Soil formation	Renewal of soil nutrients	May receive fertilizer	May receive fertilizer	May receive fertilizer	May develop a soil	May receive fertilizer	Gley soils develop	
Photo-synthesis	Effective	Effective	Effective	Effective	Effective	Effective	Effective	
Primary production	Produces wood and plant matter	Produce wood	Produces wood and plant matter	Produces wood and plant matter	Produces plant matter	Produces plant matter	Produces plant matter	Growth of aquatic and floodplain plants
Nutrient Cycling	Usually maintained: depends on harvesting	May receive fertilizer or compost	May receive fertilizer or compost	May receive fertilizer or compost		May receive fertilizer or compost		
Water cycling	Maintains natural water cycle	Plays role in water flux	Transpiration important	Transpiration important	Transpiration important	Transpiration important	Evaporation important	Important for conveyance, storage, infiltration, and evapotranspiration

priority crossings, permeable surfacing, wildlife tunnels, and pedestrian overpasses can enhance the 'connectivity' of the network (Sukopp 1995).

5.1.3.4 Changing benefits through time

Contaminated land and the industrial legacy

Every piece of urban land has a land-use history. Many urban greenspaces occupy former built-up areas. Their soil often reveals a set of significant legacies from the past in terms of the soil texture and chemistry (see Sauerwein, Chapter 1.3). Often the soil beneath urban open spaces and gardens contains heavy metals and relict petroleum-derived compounds derived from petroleum at depths of around 50 cm to 1 m (Douglas *et al*. 1993). Potentially these industrial 'hangovers' pose problems for biodiversity, garden planting, and the growing of vegetables. Most importantly, present land-use does not always indicate what is just below the surface. Encouragement to develop brownfield land means that new retail parks, warehouses, and even residential areas have been established on old industrial land. While twenty-first century regulations require identification and treatment of possible contamination, many earlier developments were built of former factory sites, landfills, and mine waste dumps without adequate investigation of potential hazards.

Derelict, under-used, and neglected land (DUN) and urban ecology

Many patches of urban greenspace are temporary, while land is idle between abandonment and demolition or reconstruction. While lacking the permanency and protection of the all-important designated urban nature reserves, they often have high biodiversity, as illustrated by the high species richness on some abandoned industrial sites in Düsseldorf, Germany and London, England. Often such temporarily vacant sites may be accessible to only a few people. Temporary use licences and temporary adoption by community or wildlife groups of such areas offer opportunities for large gains. Coupling these to subsidies, perhaps to use urban food production at some sites to help in improving local physical and mental health, will extend the benefits of such transitional urban ecosystems.

Practical issues in remediation and reclamation

Remediation involves knowing what went on at a given site. It requires a clear and detailed account of the immediate past use of the land (including an inventory and map of where and how chemicals and hazardous substances were handled and stored) together with information on previous land-uses and manufacturing and storage processes. Soil and groundwater samples collected on a systematic basis help identify potential contamination and associated impacts, risks, or hazards. Many contaminants will migrate through time, passing by subsurface water movement into ponds, lakes, and rivers. Three-dimensional modelling in GIS can help predict likely paths of contaminant movement. Sometimes it is prudent to try to isolate the contaminants on site by bunds and subsurface traps, with a view to eventual removal and treatment.

On the surface, the patterns of plant invasion can indicate where problems might lie. Contaminated sites often support characteristic plant species, some of which are able to accumulate high concentrations of heavy metals in their tissue. Cultivating such plants on low to moderately contaminated industrial waste sites can provide a clean, cheap alternative to the removal of contaminated soil for clean-up off-site. In addition to the removal of contaminants, the technique also offers containment of leachates and maintenance/improvement of soil structure, fertility, and bio-diversity. Nevertheless, it needs greater understanding of how plants absorb, translocate, and metabolize heavy metals, the identification of genes responsible for uptake and/or degradation of the contaminant, decreasing the length of time needed for phytoremediation to work, disposing the biomass so produced, and protecting wildlife from feeding on plants used for remediation. Greenspace restoration requires detailed knowledge of both site conditions and the alternative treatments available. Selection of an inappropriate technique can lead to rapid degradation of the site and a loss of both ecosystem services and cultural values.

Remediation and reclamation as the new urban ecology

Remediation is usually considered to be an entirely positive step, freeing up land for redevelopment with the advantages of reducing pressure on

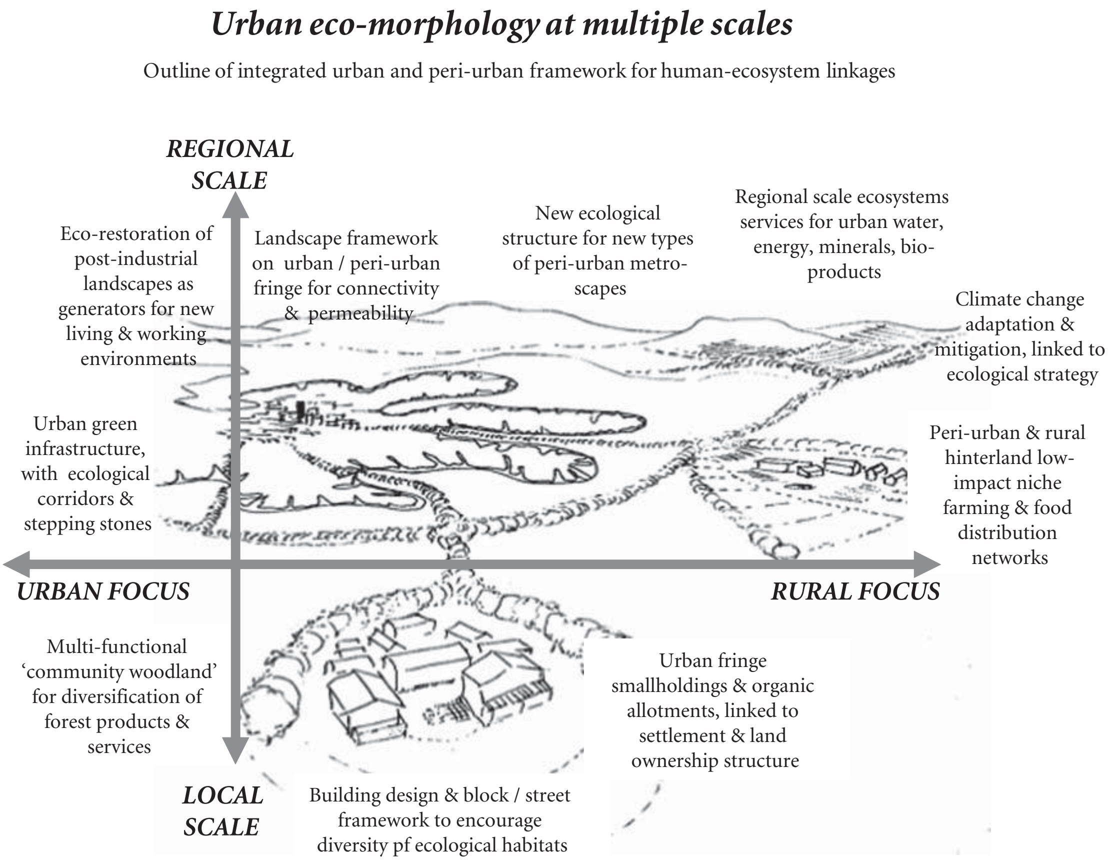

Figure 5.1.2 Multi-scalar urban ecological networks

greenfield sites and possibly improving the area within which the site is located through the removal of, or reduction in, the effects of blight. In terms of urban ecology, remediation can promote greater biodiversity. Often, however, removal of contaminants can change the diverse flora and fauna of a varied industrial site into the monotonous and regulated plantings of a modern office or warehouse park where the chemical variety that gave opportunities for diverse plant cover has been removed.

Former industrial land can provide great opportunities for wildlife in an urban setting. For example, Detroit River International Wildlife Refuge, along some 80 km of the lower Detroit River and western Lake Erie between the USA and Canada, supports 300 species of birds, including 30 species of waterfowl, 23 species of raptors, and 31 species of shorebirds, plus 117 kinds of fish…all within an urban area of six million people. Nature can be left to take its course, or creative conservation can be used to re-introduce plants and animals that had left urban areas. In this way, the new urban ecology of helping biodiversity to increase by allowing diverse habitats to develop promotes nature in the city for the benefit of all.

5.1.4 Ecological planning and investment

5.1.4.1 Good design and urban ecosystems

In all the efforts to manage green infrastructure and to extend the role of nature in urban life, good urban design is important. Social, economic, and environmental factors are all encouraging a loss of urban greenspace. The intensification of urban land-use is a world-wide phenomenon, with urban land prices and growing numbers of private cars leading to larger residential buildings and parking areas on urban land plots (see also Pauleit and Breuste, Chapter 1.1). Former private household gardens are being paved or built upon at a growing rate. Urban design has the opportunity to counteract this trend by insisting on the provision of grassed, blocked parking areas, green roofs, and strategic tree planting. Cities such as Vancouver, Canada offer many good examples.

Ecosystems can be protected through good house design and minimization of waste, of heat loss, and emissions. Adjustment can be made between the needs of domestic pets and the protection of wild creatures in cities. Street trees can be enhanced and replanted. Increasingly urban areas can become oases of green in a warmer world. Imaginative water supply systems that have household or community level wastewater treatment mechanisms offer ways of using grey water to enhance the urban green. The techniques and technology for innovative urban design that enhance urban nature exist, from the use of wildflowers in urban planting (Scott 2008) to the establishment of green roofs (Oberndorfer *et al*. 2007) and interlinked green infrastructure (see also Sadler *et al*. Chapter 5.4). The economic and social values of the urban environment still conflict with immediate human needs for shelter, and individual concerns for enhancing wealth and personal prestige. The profits of development companies not only take precedence over providing greenspaces; they often force people into houses with high maintenance costs, inefficient energy use, and inappropriate internal spaces. Better design has to challenge the conventional on all grounds, not merely on aesthetic tastes.

The great urban parks of the nineteenth century, such as Central Park, New York, or Phillips Park, Manchester, England, established to help improve the 'Health and Comfort of the Inhabitants', still play a large role in urban life: but they have to be seen as parts of larger urban greenspace networks (now termed the green infrastructure) that link to river valleys and other greenspaces of various sizes. Hierarchies of natural areas can be incorporated into urban design, but often such spaces in suburban developments are mere awkward residual patches that are difficult to get to and poorly sited for use by local residents. Good design requires participation of residents and careful thinking about the variety of uses of the available greenspaces that should be catered for (see Colding, Chapter 4.5).

5.1.4.2 Urban greenspace and green infrastructure strategies

Urban greenspace design strategies (planning and maintenance of the green infrastructure, Pauleit *et al*., Chapter 5.3) extend in large urban agglomerations to the country parks and national parks that abut onto the suburbs and which fill the spaces between cities (as for example does the Peak District National Park in England). These peri-urban ecosystems straddle administrative boundaries and may require special thinking across jurisdictions, state, and provincial boundaries. Many, such as the Tatra Mountains in Europe, may require international agreements. Designations of preserved countryside can be made for diverse reasons under a single government, so examination of all the benefits of designation and planning regulations have to be considered when protecting natural areas or areas of cultural heritage. These now have to be coupled with the environmental restrictions that may be placed on land-use in order to protect aquatic ecosystems or atmospheric resources, such as the systems of land-use management and control envisaged by the European Water Framework Directive.

Among the examples of good practice and good legislation is the work of Greenspace Scotland, which sees urban greenspace as having a crucial role to play in key elements of sustainable development, including transport, enterprise and lifelong learning, environmental quality, health, and social justice, which go well beyond traditional approaches to greenspace management. Greenspace Scotland has succeeded in developing an integrated approach to greenspace management rarely seen at national level, because it involves, and is supported by, a wide range of government departments (Greenspace Scotland 2009).

At the city and metropolitan region level in the UK, policies for protection of the green infrastructure at regional scale for spatial planning have been adopted by Glasgow and the Clyde Valley (http://www.gcvcore.gov.uk/structure_plan/structure_plan.htm) and London (Mayor of London 2002). In the USA, among many metropolitan area schemes are the Chicago Wilderness Biodiversity Conservation Plan—a regional biodiversity conservation initiative developed by a public, private, and non-profit alliance that includes green infrastructure concepts and principles (Moskovits *et al.* 2002); the Twin Cities Minnesota Metro Greenways regional green infrastructure network; and the Portland, Oregon Metro Greenspace Program—a regional conservation initiative involving a partnership of state, regional and local government agencies and non-governmental community organizations (Bryant 2006).

One of the largest green infrastructure strategies is PlaNYC, an ambitious 30-year 'smart growth' strategy, which plans for 900,000 new residents. It is branded as a carbon-reduction strategy because of the comparatively lower climate emissions per capita of New York residents. There is a full green infrastructure strategy, including a re-imagining of the public realm, street-tree planting, ensuring all residents live within 10 minutes walk of a play/greenspace, a range of 'destination parks', waterway cleaning, and wider urban–rural linkages (PlaNYC, 2007).

5.1.4.3 Criteria for design

Good design assumes some acceptance of a universal, national, or local sense of what is good. The arbiters of what is good are seldom identified. For some people wind turbines are good design and environmentally beneficial, for others they are simply obnoxious blots on the landscape. Houses that are pastiches of the homes our newly affluent great-grandparents bought 120 years may be considered ideal. For others they are simply boring, inefficient responses reproduced by profit-oriented developers. Clearly, in talking about urban ecology and the encouragement of biodiversity, vegetation, and wildlife in cities, we have to address the whole process of the attitudes of individuals and institutions in the planning, design, construction, operation, maintenance, and eventual demolition or reuse of urban buildings, infrastructure, and associated greenspaces (see also Box, Chapter 5.6). Alternatives have to be found to the present crisis-focused campaigning, that only sometimes succeeds in protecting greenspaces.

Many authorities require the provision of new green open space as a part of any new development. For example, the Luton and South Bedfordshire (UK) GI (Green Infrastructure) Plan states that 'New development will be expected to aid the delivery and management of GI by ensuring that green open space is provided for in the emerging preferred urban extensions and other new developments' (Bedfordshire and Luton Green Infrastructure consortium 2007). The planning system in South Australia has a public open space policy covering the incorporation of both 'usable' public open space and environmental open space into all new developments (http://www.planning.sa.gov.au/go/overview/planning-reforms-2008/new-regional-plans-for-sa/new-30-year-plan-for-adelaide/features-of-all-new-development). The government will promote innovative measures, including green roofs and living walls, to retain and even increase biomass. Water sensitive urban design will also be a feature of all new developments. Targets for retention of native vegetation will be set at regional and statewide levels. In addition to such requirements on developers, efforts have to be made to ensure that the open space so provided is truly usable, meets multiple demands, and provides a variety of ecosystem services. Ideally, there should also be an endowment for maintenance, to ensure that local authorities or landowners can sustain the quality of the green infrastructure. One of the causes of a slow uptake of sustainable urban drainage systems that use green infrastructure is that there is all too often no clear funding for maintenance of the system after the developers have handed over the site.

5.1.4.4 Economics of green infrastructure

The economic dimension is often downplayed in urban ecology policy and analysis, although it has great potential to justify both investment and ongoing maintenance of green infrastructure. Some interesting economic approaches have emerged in recent decades, which are suitable for urban ecology issues, often at the edge of the conventional economic system:

- 'Welfare economics' looks at the distribution and redistribution of benefits and costs between social groups, some of which may be measured by money (Pigou 1952). For instance, public investment in a park or greenspace could be assessed for benefits and costs to different groups, compared with other uses for the same money, or with different rates of interest and return. This could also look at alternatives to public investment with free access—for instance, charging for entry in a cost-recovery model; a levy on nearby housing in a social investment model; or selling-off catering franchises in an added value model.
- 'Institutional economics' follows this with a wider focus—the interactions of organizations, management, labour, consumers, and so on. Some key concepts include 'transaction costs', property rights, ownership, and stewardship (Williamson 1985). Urban ecology questions can be a good demonstration of this. A public park, for instance may be cared for, or suffer vandalism from, neighbours, depending on the institutional stewardship, with large implications for running costs and benefits (Jacobs 1997; Paavola & Adger 2006).
- 'Environmental economics' seeks to measure environmental resources and impacts through the medium of money, using various methods to construct shadow markets to assign money values for rivers, forests, landscapes, and so on (Common 1996). For urban ecology, there are many studies which aim to define the value of greenspace, or tree-lined streets: these often find that the total benefits far exceed the capital and running costs (but the money is still hard to find (Willis and Garrod 1993)).
- 'New economics' focuses at the other end of the scale, on the informal and 'social economy'. This looks at patterns of non-monetary trading or reciprocity in neighbourhoods, between social or kinship networks, and it observes how the dominance of commercial activity can often generate unemployment and dependency (Simms 2009).
- Generally, there is a reexamination of the meaning of true wealth and prosperity, particularly for benefits such as greenspace which are often beyond the market (Jackson 2009).

All these economic dimensions can combine to inform practical issues in urban ecology. The current green infrastructure strategy and research programme in the North-west of England, for example, shows a rich mixture (TEP Consultants 2008):

- A direct cost–benefit calculus for the effects of green infrastructure on economic output, investment, avoidance of negative costs such as health damage, and the management of risks such as flooding.
- Then there is a wider welfare economics concerning the total costs and benefits to different groups, particularly those with high levels of deprivation.
- Now being researched is the institutional economic case for partnership agencies, who are often perceived to be more responsive and effective than local authorities.
- This relies on an environmental or ecological economics analysis of the benefits of greenspace on property values, public health, and general amenity.
- Now emerging is the new economics approach to mobilization of communities and social enterprise, so that local people will feel involved and contribute active help and passive surveillance of the greenspaces and habitats.

Overall, this shows a creative approach to the goals of the green infrastructure agenda, by actively linking with the economic logic of the 'grey' infrastructure of major roads, industrial estates, airports, and other urban utilities (EcoTec 2008). At a national and global level, the TEEB programme (The Economics of Ecosystems and Biodiversity) has provided a definitive statement for the linking of ecosystem values with fiscal policy and cost-benefit assessment (TEEB 2010). It remains to be seen how far this approach can be taken in practice.

5.1.4.5 Funding and fiscal policy

Taxation, subsidy, and regulatory measures can play an important role. CABE Space (2009) has shown how values can be placed on urban parks, demonstrating that two of England's most well-known public open spaces are worth more than £50 million each. Highbury Fields in London was valued at £49 million, excluding the on-site public swimming pool, or £53 million including the pool. Sefton Park in Liverpool was valued at £105 million excluding the Palm House, or £108 million including it. However, these values underestimate the total value of these greenspaces to society. For instance, the valuation methodology does not include: the cost of designing a landscape; its biodiversity value; its value as a way of mitigating the effects of climate change, such as flooding; and various other aspects of value that parks can bring to communities. Putting values such as these before councils can help persuade them to allocate greenspace maintenance expenditure on a greater scale than when assets have been depreciated of time to extremely low values, just £1 in the case of some UK local government councils.

The importance of setting up maintenance funding and the idea of endowments for open space in new developments are emphasized in CABE Space's conclusions about paying for urban parks in England (http://www.cabe.org.uk/public-space/parks/paying-for-parks):

- It is important to set up dedicated funding and management arrangements from the start.
- Successful urban greenspace funding is often underpinned by a strategic approach to funding and management.
- The success of funding models is inextricably linked to the physical, political, and social context within which the greenspace is located.
- Market-driven models are more applicable in areas of high housing demand, allowing these areas greater flexibility.
- The skills and capacity of those running greenspaces have a clear impact on the quality and sustainability of spaces.
- Legislative reform in England has created a flexible statutory environment, encouraging more sophisticated funding mechanisms.

For the longer term, endowments provide secure funding for urban greenspaces from the interest gained on investments in assets such as property or the stock market. For instance, the Corporation of the City of London manages around 4,000 hectares of greenspace in and around London using funding that comes primarily from historical property investments.

5.1.4.6 Maintenance, public attitudes, and social cohesion

Even if adequate funding is available, the quality of open space and its ability to provide provision of ecosystem services will depend on people: both those with statutory responsibilities for it and those who live around it and use it. Sometimes it would appear that potential users of urban open space are put off by the way some of its neighbours use the edge of the open land as a convenient rubbish dump—a localized version of the 'tragedy of the commons' (Hardin 1968). Walking along river valleys in the hilly eastern parts of eastern Greater Manchester, UK, it is often possible to see the back fences of suburban gardens at the top of the incised river valley slope. Below many of those fences is a stream of debris, ejected from the private gardens into the public open space. Such behaviour challenges environmental managers and campaigners to find ways to make the appearance of the open space as important as the appearance of the private house and garden. Changing attitudes and developing a greater sense of personal responsibility is important. Many primary schools have a nature corner and pupils in reception class quickly develop a sense of care, both for nature and for the cleanliness of the school playground. Yet many do not carry that sense of care for the immediate environment into later life. In some cultures, environmental care is enforced by strict littering penalties and close policing of littering and fly-tipping. Behaviour has been changed in others by a combination of by-laws and social pressure, for example, dog waste is picked up by dog owners instead of being left to foul English parks and roadside verges as it did 50 years ago.

A sense of personal responsibility plays an important part in sustaining the quality of urban greenspace. People commit themselves to their local environment in differing ways. The small community groups of 'Friends of the little park' are key elements in providing that combination of acting locally while looking at urban greenspace across conurbations and nations. The owners and supporters of individual elements within the urban open space system are a key part of the stakeholder network needed to maintain the multifunctional urban greenspace infrastructure.

Another important component in attitudes is to have greenspace values and principles in the minds of decision-makers and officials, particularly those involved in urban planning and development. The leadership of national governments in promoting the development of regional and local green infrastructure strategies is praiseworthy. However, the key is in the implementation of those strategies on the ground by hard-pressed local governments who often have tight budgets and an excessive number of central government targets and directives to meet.

5.1.5 Conclusions

This brief review shows some of the most topical human dimensions in managing wildlife and greenspace in the city. The one overarching message is that urban ecology is embedded in the life and structure of the city and the wider city-region. The social, economic, cultural, political, design, and engineering dimensions are all crucial for success—or conversely, to understand the causes of failure. Likewise, the management of the wider urban environmental metabolism is essential to the ecological integrity of the whole system. Given the compound pressures of climate change, water shortage, food stress, and rapid urbanization, it is clear that our common future depends in part on the success of urban ecology and green infrastructure.

Some key principles for green infrastructure (Benedict & McMahon 2001) have become well established (note that in many European countries this is extended to include 'green–blue infrastructure'):

- Green infrastructure should be the framework for conservation and development.
- Design and plan green infrastructure *before* development.
- Linkage between greenspaces and habitats at a variety of scales.
- Green infrastructure functions across multiple jurisdictions and at different scales.
- Green infrastructure is grounded in sound science and land-use planning theories and practices.
- Green infrastructure is a critical public investment.
- Green infrastructure involves diverse stakeholders.

Now, the need is to increase the thinking about preserving the green and enhancing wildlife habitats, while providing cities and towns that are fit for people to live in, and giving their children the excitement and the health and educational benefits of encounters with the urban wild! Growing-up requires contact with nature, the opportunity to enjoy the world, to appreciate flowers, to see animals in natural habitats, to hear birdsong, to watch the changing seasons, to wonder at the diversity and interdependence of life. For the majority of the world's children that experience has to start where they live: in the urban environment. Many live in poverty in inadequate housing with little opportunity to enjoy parks and public open spaces. That should not stop those who understand the beauty and potential of cities from making access to nature for all the world's urban children their goal and future target.

CHAPTER 5.2

Urban Ecology and Human Health

Konstantinos Tzoulas and Kim Greening

5.2.1 Introduction

The aim of this chapter is to provide an overview of the role that urban greenspace can play in public health approaches, which are aimed at creating the broad social and environmental conditions that promote human health and well-being. Cardiovascular disease and mental health problems are major public health challenges facing urban populations in the twenty-first century. Public health approaches to dealing with these issues may be 'downstream', focusing on treating the symptoms of ill health through the acute health-care system; or they may be 'upstream', focusing on creating the social, economic, and environmental conditions which promote health and prevent illness and disease. McKinlay (1979) first used the term 'looking upstream' in an address to the American Heart Association. It refers to the analogy of a rapidly flowing river to represent ill health, whereby many patients fall into the water only to be rescued downstream by doctors who have no time to look upstream to consider why the patients are falling in. He characterized 'downstream' activity as short term and related to specific problems at the individual level, and upstream as long term and related to the prevention of problems and the promotion of health.

Urban ecosystems can make a significant contribution to 'upstream activity' in public health. Urban ecosystems are inextricably linked to providing and maintaining the broad environmental conditions that are central to the liveability of cities (i.e. climate, hydrology, soils, and the natural and built environments; see Sections 1 and 4). Understanding the dynamic interaction between environmental, ecological, and social systems in urban areas is key to the understanding of public health determinants. So, knowledge of urban ecosystems could be used to inform 'upstream' public health interventions.

Urban ecosystems comprise mosaics of habitats including forests, grasslands, formal parks and gardens, churchyards, incidental greenspace, private gardens, and wetlands (see Section 2), as well as numerous ecological processes that determine their functioning (see Section 3). In this chapter, the term 'urban greenspace' is used as an umbrella term for all the different, natural and human maintained, urban habitats and their ecological processes. Urban greenspace provides opportunities for urban residents to be in contact with the natural environment. Pretty *et al*. (2005) suggested that there are three ways in which people can gain health benefits from nature. The first way is through active participation, such as sports, gardening, or conservation work. Secondly, people gain health benefits by the presence of nearby nature which is incidental to their personal activity (e.g. cycling or walking near a greenspace). Thirdly, people can gain health benefits by simply viewing natural scenes.

Evidence presented in Chapter 5.1 (Douglas and Ravetz) suggests that human interaction with urban greenspace may have a key role in the prevention of disease and the promotion of health and well-being. Indeed, urban greenspace plays a central role in a number of public health activities focusing on improving human physical health, and psychological and social well-being. Table 5.2.1 outlines examples of such public health activities across different spatial levels.

The contributions of urban greenspace to the social well-being and to the physical and psychological

Table 5.2.1 Examples of public health activities that involve contact with greenspace and corresponding ecological knowledge that could be integrated into their design

Spatial level	Example of public health activities	Relevant ecological knowledge
City/ wider population	Environmental, nature conservation and public health policy and/or legislative frameworks; Ecosystem management approaches to public health (EcoHealth);	Application of ecological and landscape ecology knowledge in habitat creation and restoration and in green infrastructure planning;
City/wider population & Neighbourhood	Walking for Health Initiative; Green Gym®; Riding for the Disabled; Sure Start; Casting for Recovery; Mosaic; Forest Schools;	Knowledge of the distribution and abundance of habitats and species (including infectious diseases); Use of applied veterinary knowledge; Use of nature conservation knowledge and skills;
Neighbourhood	Natural play areas; Community allotments; Nature trails;	Use of applied ecological knowledge of plants and habitats (i.e. in horticulture, gardening, and vegetable growing); Creation of interconnected networks of urban greenspaces; Creation of natural play areas, school gardens, green roofs and green walls;
Neighbourhood & Individual	Horticultural therapy; Sensory gardens; Hospital gardens; Prison gardens;	Quality of management and maintenance of greenspace; Creating therapeutic landscapes and healing gardens; Incorporating green elements into indoor and outdoor healthcare facilities.
Individual	Green exercise routines; Pet therapy; Post traumatic stress recovery activities.	

Key: The list of examples and of relevant ecological knowledge is not exhaustive and is mainly drawn from examples within the UK; there may be significant overlap between the physical, psychological, and community health outcomes of different public health activities; **EcoHealth:** public health initiatives that incorporate community involvement as well as ecological, veterinary, and human health knowledge in ecosystem management; **Walking for Health Initiative** (previously known as Walking the Way to Health Initiative): encourages the enjoyment of natural spaces through participation in local health walks; **Green Gym®**: an initiative that integrates nature conservation, human health, community well-being and environmental improvements; **Riding for the Disabled:** a scheme that improves the lives of people with disabilities through the provision of opportunities for riding and/or carriage riding; **Sure Start:** a government programme to deliver the best start in life for every child by bringing together early education, childcare, health, and family support; **Casting for Recovery:** an outdoor programme which provides fly fishing retreats for women who have, or have had, breast cancer; **Mosaic:** builds links between ethnic communities and national parks; **Forest Schools:** an innovative educational approach to outdoor play and learning; **Natural play areas:** are greenspaces innovatively designed for children's play; **Community allotments:** plots of land leased from the local authority by people to grow their own fruit and vegetables; **Nature trails:** paths that run through greenspaces; **Horticultural therapy:** programmes that use planting and caring of plants as a therapeutic method; **Sensory gardens:** gardens specifically designed to stimulate all of human senses; **Hospital gardens:** greenspaces incorporated in the outdoor design of hospitals, which may or may not follow the principles of therapeutic landscape or healing gardens design; **Prison gardens:** greenspaces provided by prisons for inmates to care for; **Green exercise routines:** formal or informal physical activity taking place in greenspaces; **Pet therapy:** programmes that incorporate the caring of pets as part of therapy for psychological ill health; **Post traumatic stress recovery activities:** that incorporate contact with greenspaces.

health of individuals are addressed in separate sections in this chapter. However, these contributions are inextricably interrelated at both practical and conceptual levels. The contribution of urban greenspace to the prevention of public health problems, such as cardiovascular disease, some types of cancer, and type II diabetes, through the provision of opportunities for physical activity, is addressed in the first part of the chapter. This part of the chapter uses the Walking for Health Initiative in the UK as an illustrative example (previously known as Walking the Way to Health Initiative).

Contact with urban greenspace can also help to alleviate stress, restore attention capacity, and provide opportunities for people to self-regulate their emotions. The effects of urban greenspace on the psychological health of people may have a role to play in the treatment of depression and anxiety. These aspects and related practical schemes, such as green exercise and ecotherapy initiatives, are outlined in the second part of the chapter.

The third part of the chapter is concerned with the social determinants of health. Enhancing feelings of community well-being, social inclusion, and improving quality of life are effects of urban greenspace that could improve the liveability and enjoyment of urban areas. Green Gym® in the UK is a good example of an initiative that integrates nature conservation, human health, community well-being, and environmental improvements. Therefore, this chapter will highlight that urban ecological knowledge integrated into the planning, design, and management of cities and of healthcare systems and initiatives, can play a principal role in improving the physical, biological, and social urban environments.

5.2.2 Urban ecology and physical health

Greenery is seen as a factor that may increase walking and physical activity patterns in neighbourhoods (Jackson 2003). The structure, composition, and management of urban greenspaces can influence how they are perceived and used by people. Urban habitats that are not perceived as threatening can provide both physical and psychological health benefits to people who may be in passive contact with, and to those who are physically active in, them. The application of ecological knowledge is central to maintaining the structure, complexity, and composition of urban habitats that are necessary to meet both human aesthetic and nature conservation goals.

Irrespective of peoples' socio-economic background, contact with urban greenspace can improve life expectancy (Takano *et al*. 2002) and self-reported health (de Vries *et al*. 2003). These health effects may be related to the effects of urban greenspaces in reducing air pollution (Patrick & Farmer 2007) and urban noise (Gidlof-Gunnarsson & Ohstrom 2007), as well as in ameliorating the urban heat island effect (Whitford *et al*. 2001). Furthermore, urban greenspaces provide opportunities for people to engage in formal and informal physical activities which can have additional health effects to those gained from improved environmental conditions.

Sedentary lifestyles are associated with increased risks for morbidity and mortality. The World Health Organization (2002) estimated that, globally, physical inactivity causes 1.9 million deaths annually and that 22 per cent of these mortality cases are attributable to ischaemic heart disease; and between 10 and 16 per cent are caused by breast, colon, and rectal cancers, and diabetes mellitus. The proportion of mortality associated with physical inactivity is highest in the American, European, and Western Pacific regions. In Eastern European countries the proportion of all deaths that can be attributed to physical inactivity can be up to 10 per cent and in the rest of Europe and Northern America (including Canada) it can be up to 8 per cent (World Health Organization 2002). Empirical studies have shown that people engaging regularly in moderate to vigorous physical activity have a reduced risk of developing hypertension, ischemic heart disease, diabetes, colorectal and breast cancers, and depression. Furthermore, they are more likely to maintain healthy body weight and musculoskeletal health (World Health Organization 2002). Hence, incorporating physical activity in lifestyles is key to maintaining good health.

The minimum amount of physical activity required by adults to gain any health benefits is at least thirty minutes of moderate intensity physical activity on at least five days a week (moderate intensity physical activity is expenditure of 3 to 6 times the rate at which adults burn 4.187 Joule, i.e. 1 kcal, at rest; Dawson *et al*. 2006). People who do not reach these levels of physical activity are classified as physical inactive (having a sedentary lifestyle). Although physical activity can be taken at gyms, motivation to continue physical activity is more likely to be sustained if this is taken in the natural environment (Bird 2004).

There is some evidence that proximity to urban greenspace may be associated with increased levels of physical activity even after controlling for education levels, age, and sex (Humpel *et al*. 2004).

However, another study that also controlled for socio-economic characteristics found that the amount of greenspace in the neighbourhood did not necessarily make people more physically active (Maas *et al*. 2008). Even if the role of urban greenspace in increasing levels of physical activity is disputed, it has been argued that appropriately designed urban landscapes and gardens could be used to increase physical activity in sedentary and vulnerable patients or residents (Bird 2004).

The concept of green exercise refers to the synergy of physical activity in natural settings and the consequent gain in health benefits (Pretty *et al*. 2005). Consequently, green exercise includes any informal physical activity that takes place in natural settings, such as walking or cycling in a local park, undertaking conservation work in local woodlands, gardening in private gardens, mountain climbing or kayaking in rivers. Green exercise has been found to improve self-esteem and overall physical fitness; have a positive effect on feelings of anger, confusion, depression, fatigue, tension, and vigour; and help people achieve more than the recommended amounts of weekly physical activity (Pretty *et al*. 2005). Common types of green exercise that are used in targeted public health initiatives for improving the physical and psychological well-being of people include walking in greenspaces and undertaking nature conservation work.

Walking is a simple way by which people can increase their physical activity levels. Recognizing this, William Bird, a UK-based General Medical Practitioner, developed the concept of health walks in 1995, that is, a brisk walk taken regularly and with the purpose of improving health (Dawson *et al*. 2006). In 2000, this concept gave rise to the Walking for Health Initiative (previously known as Walking the Way to Health Initiative; funded by The British Heart Foundation and the Countryside Agency until 2005, and subsequently by Natural England). This initiative has been endorsed by the UK Department of Health. The aim of Walking for Health is to engage physically inactive people, and particularly those living in areas of poor health, in walking to improve their health. Characteristically, Walking for Health schemes have been using local greenspaces to undertake health walks.

A national evaluation of the changes to physical activity levels amongst adults who participated in Walking for Health schemes, undertaken by Dawson *et al*. (2006) on behalf of Natural England, found that the initiative was successful in attracting people who had been affected by ill health (physical or psychological) and in particular disadvantaged groups (i.e. living in areas of poor health, non-white groups, high socio-economic deprivation, and registered disabled). Furthermore, the evaluation found that the initiative played a key role in the rehabilitation and social and psychological support of people recovering from ill health, and that it helped participants achieve and retain the recommended levels of weekly physical activity. Overall, the Walking for Health Initiative made participants feel healthier and more socially connected (Dawson *et al*. 2006).

On the other hand, the Walking for Health evaluation found that the most frequent worries of people who did not engage in walking in local greenspaces were about perceptions of personal safety in terms of both crime and health and safety issues (e.g. tripping and falling; Dawson *et al*. 2006). Therefore, it is essential that urban ecologists and planners understand different social perceptions of personal safety and apply this knowledge in the development of urban greenspaces.

The planning of ecological networks, green infrastructure, greenways, and green hearts (see Pauleit *et al*., Chapter 5.3, Colding, Chapter 4.5), as well as local parks and greenspaces, is particularly well placed to create interconnected regional to local networks of greenspaces that people can use to walk in. Connectivity between different greenspaces is important as this could increase the potential length of different health walks. Incorporating paths, and/or steps, on slopes at various gradients into the design of urban greenspace could provide health walks of different intensity. Furthermore, the creation of different habitats designed according to local communities' needs, perceptions, and values would allow for differences in visual scenes and so promote a variety of aesthetic responses. These effects of contact with urban nature could enhance the social, physical, and psychological health benefits gained from purposeful brisk walking in urban greenspace.

5.2.3 Urban ecology and psychological well-being

Taking part in activity in urban greenspaces does not only confer physical health benefits but also brings about psychological health benefits. The report Ecotherapy – the Green Agenda for Mental Health (Mind 2007) identifies the need for greenspace to be a part of mental healthcare planning and assessment, and for General Medical Practitioners to consider referring patients for green exercise.

Mind, a UK based mental health charity, has introduced ecotherapy activities in many of its 200 local associations in England and Wales as an alternative to antidepressants (Mind 2007). This charity sees ecotherapy as a free, accessible, and clinically valid treatment of stress, anxiety, and depression through being active in nature (Mind 2007). A study of patients taking part in ecotherapy, that is, gardening, conservation work, cycling, and running found that such activities had a significantly positive effect on their psychological well-being (i.e. decreasing levels of stress and depression), and on their overall fitness and self-reported health (Mind 2007). Furthermore, improvements in self esteem, depression, and tension were greater from physical activity taken outdoors than indoors (Mind 2007).

The psychological health effects of nature are underpinned by a dual theoretical background based on stress recovery and attention restoration theories. The first of these theories stipulates that even passive viewing of natural environments after psycho-physiological stress produces stress-ameliorating effects which may ultimately confer health benefits (Ulrich 1984). The second theory suggests that contact with nature, or natural views, can confer psychological well-being benefits through attention restoration (Kaplan & Kaplan 1989). These two theories have been supported by numerous experimental studies (Sustainable Development Commission 2008).

Ulrich (1997) proposed the theory of supportive design for healthcare facilities. According to this theory positive distractions (i.e. access to greenspaces, natural views, or even indoor plants) in healthcare facilities could aid the recovery of patients (Ulrich 1997). This suggests that improvements in psychological health from (even passive) contact with nature may also lead to enhanced recovery from physical illness. If this is so, then incorporating greenspace elements in the design of healthcare facilities and green exercise in the treatment of patients becomes central. Hence, ecological knowledge could be used to inform the design, planning, and development of both outdoor and indoor healthcare facilities.

The stress reduction (Ulrich 1984) and attention restoration (Kaplan & Kaplan 1989) effects from contact with nature may be central to understanding other psychological well-being benefits gained from it. In a study on the self-regulation of mood, Korpela *et al*. (2001) found that visits to natural favourite places are associated with an increase in positive feelings, with forgetting worries and random thoughts, and with recovering attention focus and relaxing. Furthermore, adults with negative mood (Korpela 2003) and those with health complaints (Korpela & Ylén 2006) were more likely to visit natural favourite places than sporting, commercial, or other venues. These studies indicate that contact with urban greenspace can be a key factor in controlling negative mood as well as in improving overall self-reported health. Therefore, it is critical to create a variety of urban open and greenspaces that allow people to have informal and individual restorative experiences.

Studies have shown that the greener the play area of 7 to 12-year-old children with attention deficit disorder the better their attention focus was (Faber-Taylor *et al*. 2001). Another study found low-income children who had recently moved into neighbourhoods with natural views had higher levels of attention capacity than children who had moved to neighbourhoods that did not have as many natural views (Wells 2000). The Swedish concept of Forest Schools has been utilizing opportunities presented by local greenspace for integrating play activities, practical ecology, and therapeutic theory in the outdoors children's curriculum. This concept has been adopted in other Scandinavian and European countries. Forest Schools integrate outdoor play and learning and so they engage pupils and adults with the natural environment. Forest Schools could not exist without access to appropriate greenspaces or natural areas. Integrating green roofs, green walls, and school gardens, with neighbouring formal and informal greenspaces in the planning, design, and development of educational establishments could facilitate the development of innovative educational approaches such as Forest Schools. Additionally, promoting contact between pupils and the natural environment may have a beneficial effect on their educational achievement (Wells 2000).

In healthcare establishments in North America, healing gardens and therapeutic landscapes are used to promote the psychological and physical health benefits from contact with nature (Larson & Kreitzel n.d.). Healing gardens are associated with general healing, not necessarily referring to the cure from a given illness, and may be mainly associated with spiritual healing. On the other hand, therapeutic landscapes are specifically related by design to a particular aspect of a disease or healing process. These landscapes are designed to produce a measurable outcome, similar to the effects of medication, with regards to an illness (Larson & Kreitzel n.d.). Successful healing gardens and therapeutic landscapes include the following design principles: variety of spaces; prevalence of green material; encouragement to exercise; the provision of positive distractions; minimal intrusions; and minimal ambiguity (Larson & Kreitzel n.d.). Design principles like these could be incorporated into the planning requirements for the development of new healthcare facilities, or in the design briefs of new urban greenspaces.

5.2.4 Urban ecology and social well-being

Quality of life is a broad term that encompasses social, economic, environmental, and health factors. Greenspace can contribute to quality of life by enhancing feelings of psychosocial well-being and by improving the ambient environmental and social conditions. Shafer *et al*. (2000) evaluated peoples' perceptions of how greenways may contribute to quality of life and found that reduced air pollution and access to recreational facilities were amongst the most significant contributions to quality of life. The highest contributions were the opportunities for health and fitness activity (Shafer *et al*. 2000). However, the importance of greenspaces for the social well-being of urban communities may be as fundamental as providing recreational and fitness opportunities.

Social relationships and community feelings are central to personal well-being. Urban greenspace can enhance such feelings and consequently contribute to the well-being of individuals. For example, a study by Kuo (2003) found that well-maintained urban greenspaces in residential areas can foster social ties between neighbours, enhance community safety and children's play, and reduce antisocial behaviour and crime. Another study found that urban greenspaces can increase residents' feelings of attachment towards the community and their interactions with other residents (Kim & Kaplan 2004). Furthermore, Westphal (2003) demonstrated that, although dependent on local community and organizational characteristics, engaging people in urban forestry projects can be a tool for community empowerment at the individual, organization, and community level. Therefore, urban greenspace can be a main factor in strengthening community interactions and social support. These in turn are central to individuals' feelings of wellbeing (Rioux 2005).

A project that illustrates the benefits of community engagement through participation in local greenspaces is the Healthy Parks, Healthy People Campaign, run by Parks Victoria, Australia. This campaign aims to develop a community understanding of the value of parks to the health of the community. The slogan 'Healthy Parks, Healthy People' indicates that the health of local parks is related to the community's health and well-being (de Kievit 2001). The campaign involves good promotion and branding; partnerships with key health organizations; graded walks (for asthma and arthritis sufferers); publications in journals; a social isolation and post-natal depression scheme; children and families bike days; engagement and participation schemes (e.g. environmental interpretation, friends' of groups, and educational activities; de Kievit 2001). So, urban greenspaces can provide a stimulus for local community development, public health initiatives, and for developing healthier lifestyles.

Residential quality is an essential aspect of social well-being. Residential quality may be assessed on the basis of six evaluative scales comprising urban planning and design, security and social relationship, transportation and commercial services, residential atmosphere, environmental health, and facility management (Tu & Lin 2008). Within these six evaluative scales, design and quality of urban greenspace can be a key component. Peoples' appreciation of naturalistic (Fig. 5.2.1) or designed (Fig. 5.2.2) urban landscapes may differ, but people are more likely to prefer natural elements in urban areas, that is, natural versus designed and green versus built (Özgüner & Kendle 2006).

Lee *et al*. (2008) found that residents were likely to prefer neighbourhoods where the areas with trees were interconnected and varied in size and shape. On the other hand, there is also evidence to suggest that multilayered woodland edges and naturalistic looking landscapes in close proximity to where people live may not be preferred by urban residents (Jorgensen et al. 2007). Urban ecologists can play a key role in creating and maintaining urban habitats, but also in interpreting them and making them relevant to peoples' lives, especially so when they create natural looking landscapes in close proximity to residential areas. Therefore, urban ecological knowledge can have a central role in the design of high quality urban environments.

A good example of an initiative that uses nature conservation work to improve the health of people and the local environment is the Green Gym® organized by the British Trust for Conservation Volunteers (BTCV). The BTCV is a charity dedicated to local community engagement in nature conservation work. A Green Gym® is a scheme that supports 'people in gardening or local environmental improvement while providing opportunities for exercise and developing social networks' (Department of Health 2004, p. 79). Green Gym® schemes are endorsed by the UK Department of Health.

Green Gym® sessions are led by trained leaders and incorporate a range of outdoor practical projects and physical jobs, such as building stone walls, maintaining paths, or habitat creation or maintenance. Participants can join independently or by referral from their General Medical Practitioner. An early evaluation of the Green Gym® has shown this scheme to be successful, but there remains scope for physical activity specialists in the National Health

Figure 5.2.1 Naturalistic urban green space design; Leaf Street, Manchester, UK

Figure 5.2.2 Formal urban green space design; Hulme Park, Manchester, UK

Service of the UK to be more aware of the potential of greenspace in promoting physical activity. There is also a need for imaginative ways to promote and market urban greenspace to different age groups (Bird 2004). The concept of the Green Gym® is valuable because the activities involved have purpose, so exercise is secondary to environmental and social benefits (Bird 2004).

The national evaluation of the Green Gym® covered 52 projects across the UK that had been established between 2003 and 2007 (Yerrell 2008). Like the earlier evaluation (Bird 2004), the national evaluation also found that the main reasons for people joining were to be outdoors and to help in local environmental improvements, while improving health was one of the least frequent reasons for joining (Yerrell 2008). People who joined the scheme having already compromised health showed significant improvements, with those having the poorest health showing the most significant improvements (physical and mental health were measured with SF12, an international standard survey covering physical and social functioning, and general physical, emotional, and mental health, pain, and vitality; Yerrell 2008). Furthermore, participants reported that the scheme improved their feelings of self-worth and that they felt they had contributed to the local environment by habitat creation or biodiversity conservation (Yerrell 2008).

The BTCV Green Gym® demonstrates the dynamic synergy between ecology and health. Participants acquire and use practical skills in applied ecology to improve local habitats and maintain species. These improvements in turn provide health benefits to other users of greenspaces. Importantly, while the participants engage in practical conservation work they gain health benefits from being in contact with nature and from being physically active.

Imaginative and creative provision of greenspaces with high quality maintenance and community engagement is salient in creating neighbourhoods that people would like to be active in. Different configurations of greenspaces have been associated with different types of experiences. For example, people may visit forest-type landscapes out of curiosity or interest, and park-type landscapes with a local café to socialize (Cottrell *et al*. 2005). Urban planners have a key role to play in integrating urban ecological knowledge of habitats and species in urban design in order to create settings for a variety of social activities and experiences as well as for biodiversity.

5.2.5 Summary and conclusions

It is impossible to control the array of confounding personal, genetic, temporal, and cultural factors that affect health. So, the mechanisms that underlie the positive effects from contact with greenspace are not clear (Croucher *et al*. 2007). Further interdisciplinary research amongst ecologists, sociologists, biologists, public health practitioners, psychologists, and anthropologists is required to start clarifying the underlying mechanisms by which urban greenspace contributes to human health and well-being.

Some authors have argued that, despite the lack of clear causal relationships, there is sufficient evidence to conclude that urban greenspace can be a significant factor in enhancing public health (Tzoulas *et al*. 2007). This assumption is based on the speculation that environmentally induced changes in physiological, emotional, and cognitive processes may induce, or mediate, changes in well-being and health (Tzoulas *et al*. 2007). However, further empirical research is required to evaluate this assertion.

The contributions of urban ecology to physical health are mediated through opportunities for physical exercise. Physical exercise has both preventative and curative effects on, for example obesity, cardiovascular disease, and type II diabetes. Furthermore, physical exercise taking place in urban greenspaces can be more sustained than indoor activity and confers additional psychological well-being benefits. Psychological well-being benefits from taking exercise in urban greenspace may be facilitated through and/or include stress recovery; attention restoration; self regulation of mood; and children's cognitive, physical, and emotional development. Moreover, while many of these physical and psychological health benefits depend on direct engagement with urban greenspaces, many can be gained by passive viewing. The contributions of urban ecology to social well-being are based on enhancing feelings of community well-being and satisfaction; improving quality of life; and providing preferred environments. Nonetheless, further research that integrates social, ecological, biological, and health systems is needed to establish the physical and psychological mechanisms that control the contributions to health from contact with urban greenspace (Sustainable Development Commission 2008); and to establish how ecological knowledge can be integrated in the planning, design, and management of cities and of healthcare facilities and initiatives.

CHAPTER 5.3

Multifunctional Green Infrastructure Planning to Promote Ecological Services in the City

Stephan Pauleit, Li Liu, Jack Ahern, and Aleksandra Kazmierczak

5.3.1 Introduction: green infrastructure

'Green infrastructure' was first introduced in the USA at the end of 1990s—it has been defined as an 'interconnected network of protected land and water that supports native species, maintains natural ecological processes, sustains air and water resources and contributes to the health and quality of life for America's communities and people' (Benedict & McMahon 2006). This definition of green infrastructure advocates its broad-scale, regional application in the USA in order to support the protection of natural systems from disturbance and displacement by future urban development. In Europe, green infrastructure has recently been taken up in the United Kingdom (e.g. Carol & Kambites 2007). The Netherlands and the Scandinavian countries have had experience with the related concept of urban green structure planning since the 1980s (Tjallingii 2005). This 'European' interpretation of green infrastructure, practised internationally, relates to a fine-scale urban application where hybrid infrastructures of green spaces and built systems are planned and designed to support multiple ecosystem services. No further distinction will be made in the following chapter between these related concepts, and 'green infrastructure' will be used throughout despite the observation that term 'green structure' was already used in Europe prior to the emergence of 'green infrastructure'.

By using the term 'infrastructure', greenspace planning is aligned and put on a par with other infrastructures, for example, transport, communication, water supply, and wastewater systems. Thus, a well planned and managed urban green infrastructure arguably needs to be considered as an integrative part of the city which is indispensable for its functioning in a sustainable manner (Sandström 2002). More recently, green infrastructure has been identified as a specific strategy to support resilience in cities (Ahern 2007).

Generally, there is ample evidence of the ecological, social, and economic benefits of urban greenspace (e.g. Chiesura 2004; Tyrväinen *et al*. 2005; Tzoulas *et al*. 2007). Urban greenspace is also regarded as an important element for city image, quality of life, and branding (Lynch 1985). Nevertheless, greenspace planning still plays a rather weak role in the process of urban development and greenspace is under pressure, from in-fill densification (see Pauleit and Breuste, Chapter 1.1) for example. In this chapter we examine whether green infrastructure planning can improve this situation. Moreover, to what degree do ecological thinking and knowledge inform green infrastructure planning and management? And does the planning system promote green infrastructure planning in practice?

Giving answers to these questions is a difficult task because green infrastructure is still a fairly recent and evolving concept. Green infrastructure has been promoted by different organizations and has been adopted by an increasing number of cities and regions but no systematic assessment of its effectiveness has been published to our knowledge. Therefore, the chapter takes a twofold approach.

First, we will review the main principles of green infrastructure planning, and explore whether and how these are related to urban ecological theory. Second we select two case studies—one case each—from Europe and the USA to study green infrastructure planning in practice. These allow us to discuss whether green infrastructure promotes a change of practice in planning and landscape architecture, towards adoption of designs and management practice that build on ecological principles and knowledge to effectively promote vital ecosystem services in the city—thereby addressing current and future challenges of urban transformation, such as adaptation to climate change.

5.3.2 Concepts and principles of green infrastructure planning

There is no one definition or one approach to green infrastructure planning, and this is unlikely to ever exist in practice given the differences in national and local planning cultures and needs. However, a set of shared principles emerge from the literature on green infrastructure planning (e.g. Benedict & McMahon 2006; Ahern 2007; Cambites & Owen 2006; Li 2008; Colding, Chapter 4.5) which are summarized in Table 5.3.1:

- Multifunctionality: Many urban greenspaces are monofunctional, that is, they are designed and managed to serve one particular function, such as sports pitches. This does not mean that they cannot have other functions as well, for instance, the football pitches may also contribute to stormwater infiltration or mitigation of the heat island effect. However, these additional functions are often not explicitly taken into consideration in greenspace design and management. A multifunctional green infrastructure, on the other hand, would seek to explicitly define and combine different ecological, social, and economic functions when possible. One way of assessing the benefits derived from green infrastructure is the concept of ecosystem services (see Colding, Chapter 4.5; Ahern 2007; Table 5.3.2).
- Multifunctionality has several dimensions. First, a single greenspace can provide a multitude of functions for multiple uses. Second, the green infrastructure of interrelated greenspaces can perform functions that individual greenspaces cannot perform on their own, for instance providing complementary or supplementary habitats for wildlife (Colding 2007). Multifunctionality as a strategy can strengthen the role of greenspace in the city, because having more than one function will give the greenspace additional value, and broader public constituencies of support, thus making them more resistant to development pressures or changes in political leadership. In this sense multifunctionality is an important strategy for urban sustainability. However, there may also be limits to multifunctionality, as it cannot be expected that small greenspaces in densely built areas provide for intensive recreational use and at the same time be species rich habitats—particularly for disturbance-sensitive species.
- Connectivity: landscape ecological concepts, and research on the use of greenspaces by wildlife, highlight the importance of connectivity between greenspaces to enhance species dispersal and support metapopulation dynamics (see Colding, Chapter 4.5, Opdam *et al.* 2006). However, not every species will benefit from greenspace corridors (Simberloff & Cox 1987) and the greening of the matrix of the different urban land-uses may be more important for many species' survival in the city (Melles *et al*. 2003; Colding 2007). A dense network of greenspaces in close proximity to where people live and greening of the urban matrix are also most important for mitigating the heat island effect (Gill *et al.* 2007). Greenspace corridors can be important for the ventilation of the city and they facilitate access to, and recreational use of, greenspace (Spirn 1984). Of particular relevance are un-channelized semi-natural or restored stream reaches which may form a natural system of green and blue corridors in many cities (Tjallingii 2000). Thus, connectivity is an important criterion for the design of the green infrastructure, but simplistic ideas of green corridors are not adequate. Instead,

Table 5.3.1 Main principles of green infrastructure planning (Li 2008, modified)

Principles	Planning and management of urban green infrastructure (green infrastructure) needs to:
Multifunctionality	• Consider a broad suite of ecosystem services: abiotic, biotic, and cultural. • Consider combining different functions/uses whenever possible: multiple-functions of single greenspace, interconnected green structure, and integrated structures. • Prioritize among functions/uses and set up clear goals through comprehensive analysis and stakeholder involvement. • Conduct monitoring to learn which functions are operating as expected, in a learn-by-doing adaptive manner. • Improve awareness of the multifunctions of green infrastructure through communication and public participation/education.
Connectivity	• Consider physical and functional connections between green spaces at different scales and from different perspectives: e.g. recreation, biodiversity, urban climate, stormwater management, etc. • Base green infrastructure planning on thorough analysis of the urban green space resource and its functions.
Integration	• Consider integrating and coordinating urban green infrastructure with other urban (infra) structures in terms of physical and functional relations (e.g. built-up structure, infrastructure, water system). • Create beneficial relationships through communication and negotiation between different professions, administrations, and other actors.
Communicative and social-inclusive process	• Attempt to meet the needs and interests of all stakeholders. • Involve stakeholders in decision-making through coordination, cooperation between different professions, sectors at different levels, between public sector and private sector, and public participation.
Long-term strategy	• Adopt the sustainable development concept, considering long-term benefits instead of short-term economic gains. • Consider multiple uses, interactive structures, and balance between different stakeholders' interests, which will help achieve a long-term goal. • Allow adaptation through ongoing learning and discussion between different actors.

planning for connectivity needs to build on a more holistic view of the multiple green and blue structures of a city, their specific functional character and quality, and their spatial characteristics and requirements.

- Integration concerns the interaction and links between urban green infrastructure and other urban structures. Looking at the totality of all greenspaces as an integrative green infrastructure elevates it to a level matching that of overall built-up structure and infrastructure. Instead of looking at 'green' and 'red' as separated entities, the new approach means that these are increasingly viewed as integrated partners. Integration through site scale design often leads to better expression of the multiple functions of each structure, and stimulates innovative solutions for green infrastructure as in the Green Street Program of Portland Oregon (Metro 2002).
- Communicative and socially-inclusive planning and management: Since green infrastructure includes various types of greenspaces—public, institutional, and private—and as it interacts with other urban structures, many stakeholders, or actors, are involved. In addition to urban green other interests should be considered; different opinions should be involved during the process. Therefore, a partnership approach should be adopted for green infrastructure planning (Cambites & Owens 2006). This is where

Table 5.3.2 Key abiotic, biotic and cultural ecosystem services of a green urban infrastructure (Ahern 2007, adapted)

Abiotic	Biotic	Cultural
Surface:groundwater interactions	Habitat for generalist species	Direct experience of natural ecosystems
Soil development process	Habitat for specialist species	Physical recreation
Maintenance of hydrological regime(s)	Species movement routes and corridors	Experience and interpretation of cultural history
Accommodation of disturbance regime(s)	Maintenance of disturbance and successional regimes	Provide a sense of solitude and inspiration
Buffering of nutrient cycling	Biomass production	Opportunities for healthy social interactions
Sequestration of carbon and (greenhouse gasses)	Provision of genetic reserves	Stimulus of artistic/abstract expression(s)
Modification and buffering of climatic extremes	Support of flora:fauna interactions	Environmental education

This table articulates what a green urban infrastructure can explicitly do to contribute to sustainability.

knowledge of communicative planning comes into play and assists the process of mutual learning and mutual understanding of the benefits and costs of land-use options. Economists, ecologists, and social scientists need to collaborate more to promote better insights into the trade-offs involved in land-use change decisions and make their work more accessible to collaborative planning and management. In practice, decision-makers, planners, and the public need to communicate about the benefits and losses of different land-use scenarios in a concrete sense (de Groot 2006; Steinitz *et al*. 2002).

- Strategic, long-term oriented approach: Long term benefits of urban greenspace often conflict with short-term economic gains. The benefits from single-use landscape change usually only make narrow economic sense for private or corporate interest groups, which place the costs on a broader group of stakeholders and future generations. In the short term, these programmes may be rational with respect to public or private policy objectives. But in the longer term, they may result in both economic inefficiency and the erosion of ecosystem services (de Groot 2006). Therefore, green infrastructure planning should be based on a long-term vision. Instead of preparing a static plan, it is aimed towards achieving overall, long-term goals, while at the same time allowing new inputs through on-going learning and discussion between different actors in an adaptive planning mode (Kato & Ahern 2008).

5.3.3 Green infrastructure planning in practice

The following case studies will explore how green infrastructure planning is applied in practice. Selection was based on the authors' personal familiarity with the case studies. Each case will discuss a particular theme:

- Seattle, Washington, USA: Green infrastructure for stormwater management and ecology.
- Greater Manchester, UK: synergies between climate change adaptation and ecological functions of green infrastructure.

The two case studies are thus complementary in as much as they explore at different scales—city region and site—how the green infrastructure can contribute to urban ecology and at the same time promote selected ecosystem services in the city.

5.3.3.1 SEA Street Seattle, green infrastructure for stormwater management and ecology

Green infrastructure in the USA is an emerging planning and design concept with origins relating to governmental environmental legislation. Green infrastructure is most frequently related to urban water resources, more specifically stormwater management. Many US cities have combined sewer overflow problems (CSO) that are nearly impossible, or very expensive, to cure with conventional engineering solutions (Braden & Johnson 2004). Many cities have been fined by the US Environmental Protection Agency (USEPA) for non-compliance with the Clean Water Act (1972) due to routine CSO discharges. The USEPA has thus motivated cities to pursue innovative stormwater policies and practices that favour infiltration and surface flow. More recently, cities in the Pacific Northwest have been faced with a new regulatory issue—the presence of endangered species (Salmon, *Salmo* spp.) in urban watersheds! The Endangered Species Act of 1973 requires a habitat protection response from these cities and, with Salmon as the focal species, water quality and stable flow regimes are recognized as the key challenges. Here, stormwater management is the starting point. This legislation has motivated cities to implement innovative stormwater management which, in the hands of creative planners and designers, becomes a multipurpose green infrastructure (Braden & Johnson 2004). While motivated by CSO reduction and habitat protection, cities recognized the additional collateral benefits that attend stormwater solutions—often without additional cost. These collateral benefits serve to strengthen the economic argument for green infrastructure for stormwater management, and build coalitions of public support for other benefits including wildlife habitat, recreational opportunities, and enhancing community character and quality of life.

The city of Seattle, Washington, is a leader in the implementation of innovative green infrastructure in the United States, primarily in the area of urban stormwater management. From 1972 to 1996 Seattle lost 13 per cent of its urban tree canopy, and stormwater runoff increased by 7.5 million cubic feet (Viani 2007). These trends motivated the Seattle Public Utilities Department to establish a natural drainage system programme (NDS) that mimics natural processes to slow stormwater runoff, increase infiltration, and improve water quality. The first pilot project of the NDS, the Street Edge

Figure 5.3.1 The Street Edge Alternative (SEA Street) in Seattle, Washington is a natural drainage system (NDS) designed to direct street run-off from a narrowed street pavement to planted infiltration swales. Note the adjacent street in the background showing the pre-SEA Street condition with wider road pavement and conventional stormdrains. Photo: Jack Ahern

Alternative (SEA Street, Fig. 5.3.1), was established in 2001 on 2nd Avenue in the Pipers Creek Watershed in north Seattle. The SEA Street as green infrastructure has been successful in multiple respects: it achieved its expected reduction in stormwater runoff, it has been accepted by the community, it has produced additional collateral benefits, and it served as a model for additional, similar green infrastructure experiments across the city of Seattle.

The SEA Street project is also an example of low impact development (LID)—a development model based largely on surface drainage. LID emphasizes source control and non-structural microscale management strategies to reduce or eliminate stormwater runoff in urban development through retention, detention, and infiltration of stormwater (NRDC 2001). While LID focuses on hydrology and drainage, it also advocates for an integrated approach to urban development to provide multiple functions including aesthetic improvements, biodiversity and habitat, and reduced construction costs. LID can be understood as a precursor of green infrastructure.

The primary goal of the SEA Street project was to reduce the runoff from major storm events, defined as the maximum precipitation expected during a 24 period for a given recurrence interval. The SEA street was designed to manage the amount of runoff generated by a 24 hour rainfall event which statistically occurs every second year (the 2 year, 24 hour storm) (4.25 cm) to pre-development levels, and to safely convey 100 per cent of the runoff from the 25 year, 24 hour storm, in accordance with city drainage requirements. To achieve these hydrologic goals the project designers narrowed the street to 5.5 m wide (4.25 m plus two unpaved road verges of 0.6 m each), representing 11 per cent less impervious surface than a conventional street in Seattle. The SEA Street includes planted swales and basins to create a long flow path with high surface roughness to increase the hydrologic time of concentration and to promote stormwater infiltration. Swales and basins along the SEA Street are planted with native and non-invasive ornamental species of trees, grasses, sedges, and rushes to create an aesthetically-pleasing 'wetland' environment and to promote infiltration and water quality remediation in the plant root zone. Plantings were designed in close collaboration with residents, resulting in a sense of ownership and stewardship. Since the plantings were installed, neighbours have been actively involved in voluntary care and maintenance of the SEA Street plantings.

Three years of hydrologic monitoring by Seattle Public Utilities and the University of Washington found that 98 per cent of wet-season, and 100 per cent of dry-season, stormwater runoff has been eliminated by the SEA Street project (Horner *et al*. 2002). The success of SEA Street has helped inspire and inform subsequent Natural Drainage System designs in Seattle and in other cities (Vogel 2006; Girling & Kellett 2005). Also in Seattle's Pipers Creek Watershed, subsequent projects by Seattle Public Utilities include the Broadview Green Grid and the 110th Street Cascade, an innovative NDS solution on a steeply sloping street including a series of small check dams and basins to detain and infiltrate stormwater. The natural drainage system programme (NDS) was also applied at the 50 hectare High Point housing project which includes permeable paving, surface swales, and retention and infiltration areas integrated into the design of the moderate-high density community—a leading contemporary model of low impact development.

The SEA Street is also notable for providing multiple benefits and a rare demonstration of 'learning by doing'. In addition to improving hydrological performance, the SEA Street has slowed vehicular traffic, while providing adequate space for emergency and delivery vehicles. The slower street has encouraged more pedestrian use (+400 per cent) and has encouraged more neighbourly social interactions. The project has also raised awareness of the urban watershed issues that it was designed to address. Because the project was proven to be effective through scientific water quality and flow monitoring, it has served as a model for other projects in Seattle, and other cities. The project has received numerous awards and has been published internationally—demonstrating the potential of a small project to serve as a defensible and visible model.

The SEA Street project has become a model for innovative green infrastructure in the US and internationally. The Seattle Public Utilities agency, as project managers, approached the project as a 'Safe-to-fail' design experiment (Lister 2007). To innovate, the SEA Street planners and designers acted with professional responsibility, while implementing practices that were untested for this specific application. The project was coordinated with the University of Washington, where engineers collaborated on design and monitoring to rigorously verify the projects' hydrological effectiveness. Seattle Public Utilities engaged the local community throughout the planning, design, and implementation of the project as a model of communicative and socially-inclusive process.

In 2006, the University of Washington's Green Futures Research and Design Lab conducted a city-wide planning charrette. The project, titled 'Envisioning Seattle's Green Future', engaged a diverse group of public, private, and not-for-profit interests to guide Seattle's future development over the next 100 years. The charrette was organized according to the city's watersheds, employed green infrastructure as a strategy to integrate open space, transportation, urban development, and urban hydrology. The final report provides a unique demonstration of the potential for green infrastructure to guide and inform multipurpose long term urban planning.

5.3.3.2 Greater Manchester, UK: synergies between climate change adaptation and ecological functions of green infrastructure

In the UK, the renewed interest in spatial planning has been beneficial to the adoption of green infrastructure planning in regional and urban planning. However, one of the challenges in implementing such an approach is the development of an evidence base to assess functionality of the existing greenspace resources and map potential future functions. In this respect, there is a danger that the demands of policy are moving ahead of science. With reference to Greater Manchester (UK), we will discuss how this evidence may be developed and applied to address some important challenges for sustainable urban development in this city region in transition: coping with climate change and developing an ecological framework for habitat creation and restoration. The case study will also discuss how this information is used to formulate a green infrastructure strategy and finally assess the role of the planning system in green infrastructure planning.

Greater Manchester (GM) is a conurbation of over 2.5 million people in the north-west region of England. It covers 1,276 km^2 and comprises ten local authorities (Fig. 5.3.2), who are independent planning entities but who also work together as the Association for Greater Manchester Authorities (AGMA) on key strategic and policy issues which impact Greater Manchester. These issues include planning and housing, transport, environment, and health.

Greater Manchester stretches from the Pennine uplands in the north and east, to the mossland and farmland to the south and west (Ravetz 2000). Heavy industry has provided a legacy of reservoirs, canals, subsidence flashes, railway sidings, and industrial waste which have been colonized by plants and animals. In other areas of Greater Manchester, intense agriculture has replaced traditional farming. Yet, despite these changes, habitats of more natural origin also survive (e.g. woodland, moorland, and raised bogs) (GMEU 2001). Many of the greenspaces in Greater Manchester are protected from development by having been designated as nature conservation sites or are excluded from development by being included in the green belt (Figs 5.3.3 and 5.3.4).

The green infrastructure concept gained exposure in Greater Manchester initially because of its role in adaptation to climate change. The Adaptation Strategies for Climate Change in the Urban Environment project (ASCCUE), carried out by the Centre for Urban and Regional Ecology at the University of Manchester (2003–06) investigated the potential of the green infrastructure to help Greater Manchester to adapt to climate change, and in this section we will focus on the findings related to adaptation to higher temperatures and heat waves. These, combined with the urban heat island effect, may be dangerous to human health and well-being, as demonstrated by the 70,000 excess deaths resulting from heat wave in Europe in 2003 (Robine *et al*. 2007).

In the ASCCUE project, the Greater Manchester area was firstly classified into 29 'urban morphology types' (UMTs). This classification is based on land-use characteristics. For each UMT the mean

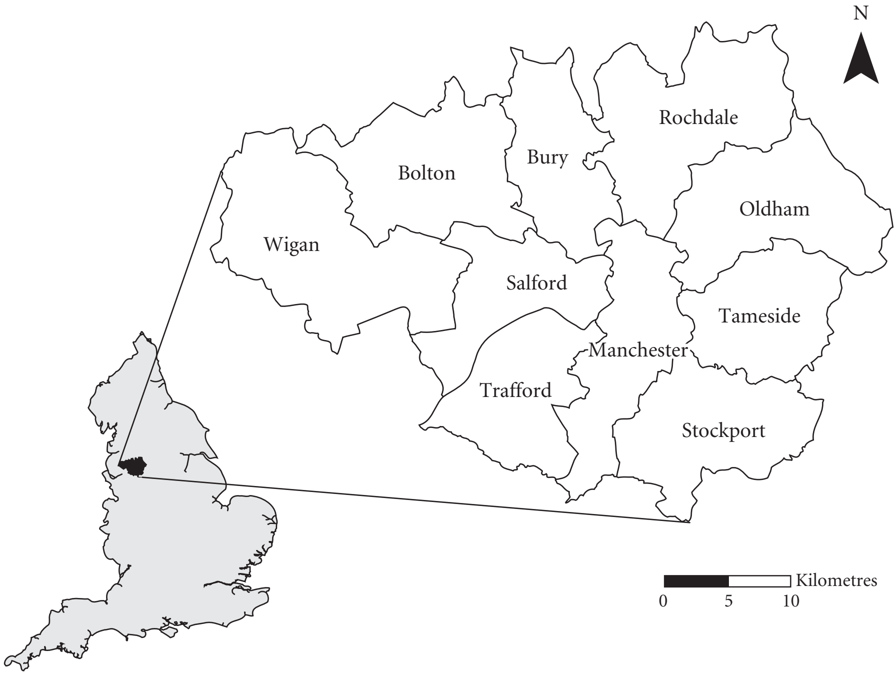

Figure 5.3.2 Greater Manchester: location in England and administration. Base map is © Crown Copyright/database right (2009). An Ordnance Survey/EDINA supplied service

percentage cover of built and vegetated areas was estimated by interpretation of aerial photographs (see also Pauleit and Breuste, Chapter 1.1). Secondly, surface temperatures were modelled to study the relationship with green cover—both with regards to today's climate and under scenarios of climate change. Finally, contrasting scenarios of urban densification vs. urban greening were applied to study the effect of varying degrees of green cover on the adaptive capacity of the city (Gill *et al*. 2007).

The findings of the ASCCUE project showed that while the percentage of green cover differed between the UMTs, the minimum greenspace cover on average was 20 per cent. Also, on average 59 per cent of the 'urbanized' area of Greater Manchester (excluding farmland; see Fig. 1.1.5 in Pauleit and Breuste, Chapter 1.1) consisted of surfaces covered by vegetation or water and 40 per cent of the vegetation cover in 'urbanized' areas of Greater Manchester were present in the UMT described as 'medium-density housing'. The surface temperatures closely followed the pattern of green cover in the urban area. In town centres and areas of retail or industry (20 per cent of vegetated areas) the surface temperatures reached 31.2 °C on a hot summer day. By contrast, in woodlands (98 per cent of vegetation) the surface temperature was only 18.4 °C and 24.0 °C in medium density housing areas where 50 per cent of land cover is vegetation (Gill 2006; Gill *et al*. 2007).

While climate change will lead to an increase of temperatures in every part of the city, greenspace will significantly buffer some of the predicted warming. By 2080, under a high CO_2 emissions climate change scenario, surface temperatures would rise to 35.5 °C in the town centres (+4.3 °C), 27.3 °C in medium-density residential areas (+3.3 °C), and 21.6 °C in woodlands (+3.2 °C) (Gill 2006; Gill *et al*. 2007). If the amount of greenspace decreased by 10 per cent in town centres and other densely built-up areas, the surface temperature under the high emissions scenario could rise by as much as 8.2 °C by 2080 from the present 31.2 °C. On the other hand, an increase of green cover by 10 per cent would keep

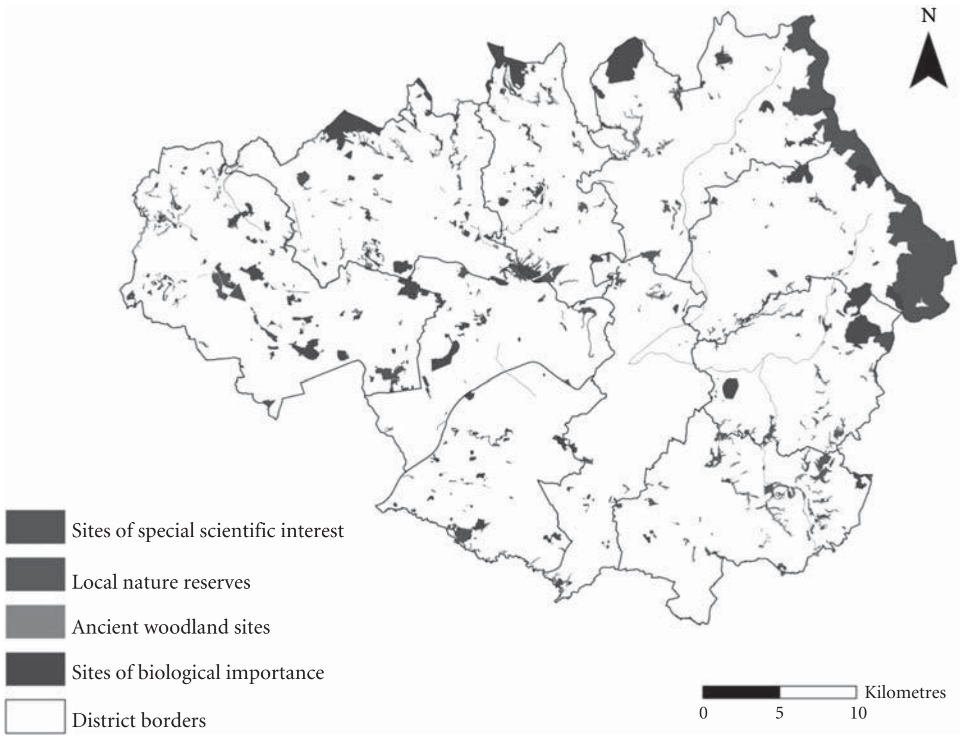

Figure 5.3.3 Areas of nature conservation in Greater Manchester. Based on Natural England (2007) and GMEU (2006) Base map is © Crown Copyright/database right (2009). An Ordnance Survey/EDINA supplied service

the maximum surface temperatures at nearly the same level as the 1961–90 baseline conditions (Gill *et al*. 2007). Therefore, increase in greenspace, especially in the densely built-up areas, is advised in order to limit the threat to human health and well-being caused by high temperatures.

Another project focusing on green infrastructure in Greater Manchester is the Ecological Framework. This project was championed under AGMA's auspices by the Greater Manchester Ecology Unit and carried out by the University of Salford. The aim was to develop a spatial and strategic guide to habitat creation and repair at the Greater Manchester scale through the land-use planning system (Richardson *et al*. 2008).

The Ecological Framework represents a departure from the traditional patch-corridor-matrix approach to landscape scale conservation, that is, actions aiming at biodiversity protection by creation of habitat corridors. Instead, the Ecological Framework approach identifies four Biodiversity Opportunity Area types sharing similar vegetation and land-use characteristics (Richardson *et al*. 2008). Three types of Biodiversity Opportunity Areas were identified as places with the highest potential to sustain biodiversity in urban areas and were as follows:

- the areas with the highest percentage of natural and semi-natural land cover (such as trees, shrubs, rough grassland, and water);
- areas with high concentrations of domestic gardens; and
- areas of high complexity of habitat mosaics (Kazmierczak & James 2008).

The fourth type of biodiversity opportunity was distinguished in the remaining areas of the conurbation as 'areas of local action'. Even though biodiversity may be currently low in these places, enhancement of existing habitats can result in sig-

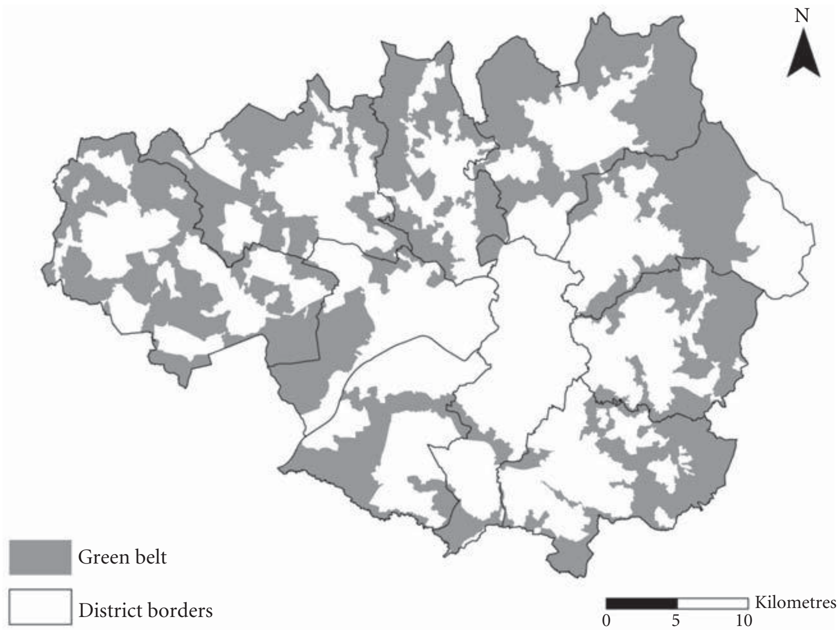

Figure 5.3.4 Green belt in Greater Manchester Green belt data courtesy of Greater Manchester Ecology Unit. Base map is © Crown Copyright/ database right (2009). An Ordnance Survey/EDINA supplied service

nificant improvements to biological diversity as these areas contain at least 20 per cent vegetation (Gill *et al*. 2007). The Biodiversity Opportunity Areas were mapped with the use of Geographic Information Systems and the focal statistics tool was applied in order to identify the highest concentrations of the natural areas, gardens, or habitat mosaics (Kazmierczak & James 2008; Fig. 5.3.5).

The spatial Ecological Framework is underpinned by a strategic framework of place-specific policy recommendations. These advocate protection of the most natural areas from development; maintenance of large garden areas by protecting them from in-fill development, and enhancement of habitat mosaics through mixed land-use planning. The policy also covered the densely built-up 'areas of local action', and advocates greening of these with small-scale green infrastructure actions, such as pocket parks, street tree-planting, green roofs, and green walls. The Ecological Framework is, at the moment of submission of this chapter, being consulted on with local authorities in Greater Manchester.

The two projects outlined above separately addressed two particular issues. Yet, the projects show important synergies and significant progress towards implementation of other strands of green infrastructure thinking. Firstly, they offer a comprehensive approach to the entire urban greenspace resource. In either of the studies there is no distinction made between private and public greenspace or between different greenspace types. Instead, the focus is on the qualities of urban areas that are important for provision of ecosystem services: respectively, cooling and biodiversity enhancement. Secondly, both studies have covered in their scope the entire area of Greater Manchester, providing classification into either UMTs or Biodiversity Opportunity Areas. Therefore, both research projects emphasize that green infrastructure is not limited to the narrow spectrum of open spaces managed by local authorities, but is a huge resource and an omnipresent part of urban areas.

Juxtaposition of the findings of these two projects shows that green infrastructure can help to

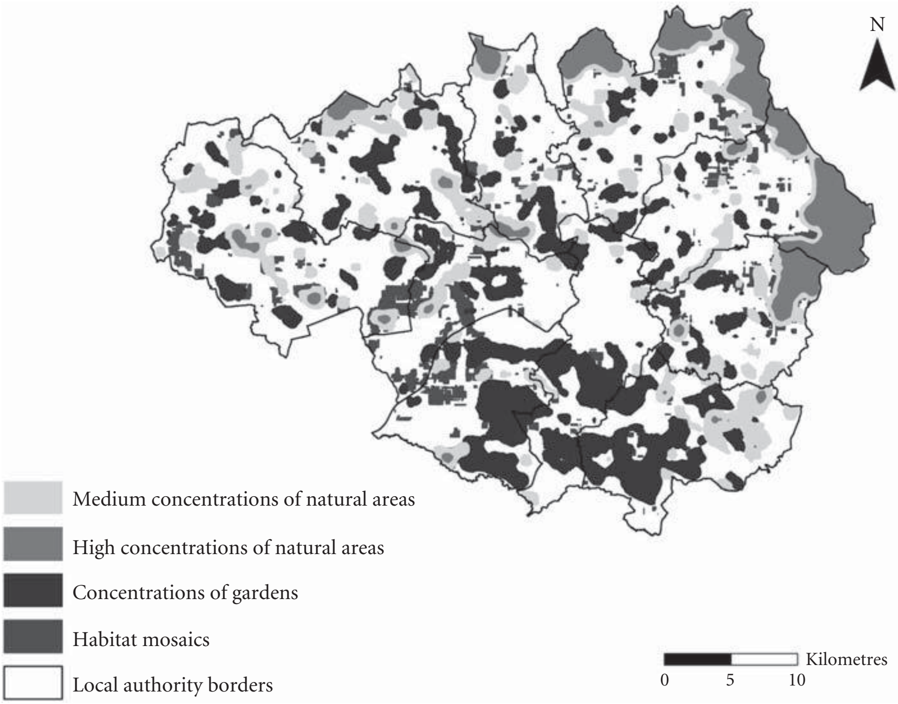

Figure 5.3.5 Biodiversity opportunity areas in Greater Manchester

simultaneously deliver both climate adaptation necessary for human well-being and biodiversity objectives. For example, the vegetated surfaces in medium-density residential areas were recognized as forming the majority of the evapotranspiring areas in the 'urbanized' part of Greater Manchester in the ASCCUE project, and was hence an important cooling asset of urban areas (Gill *et al*. 2007, see also Pauleit and Breuste, Chapter 1.1.). These areas are represented mainly by gardens, which were found to have a high biodiversity value during the literature review leading towards the Ecological Framework project. Also, the areas containing the highest percentage of vegetation such as trees and shrubs, therefore providing shade and cooling, are strongly associated with the presence of major bird sites in GM (Fig. 5.3.6).

Both projects emphasize the importance of maintaining sufficient greenspace within the built matrix and call for greater incorporation of greenspaces into new developments. Green areas on the outskirts of the conurbation provide cooling, but can also support many species, while greening of streets and creation of green roofs and green walls in densely built up areas not only contribute to cooling but also provide biodiversity benefits. Such synergies need to be emphasized in order to provide sufficient evidence of the green infrastructure benefits and to promote a consistent approach to green infrastructure in spatial planning, which in the UK will be one of the main mechanisms for the delivery of green infrastructure on the ground. This evidence would thus help to find a better balance between the goals of the compact and the green city which are equally promoted by planning policies.

Green infrastructure has only been present in the UK for about five years, and therefore has been picked up only by the planning policies that have been produced in this time. Separate planning policies for sustainable development, climate change adaptation, biodiversity protection, and open space for recreation and sport exist, but there is no integrated planning policy for the development of a multifunctional green infrastructure. This lack of

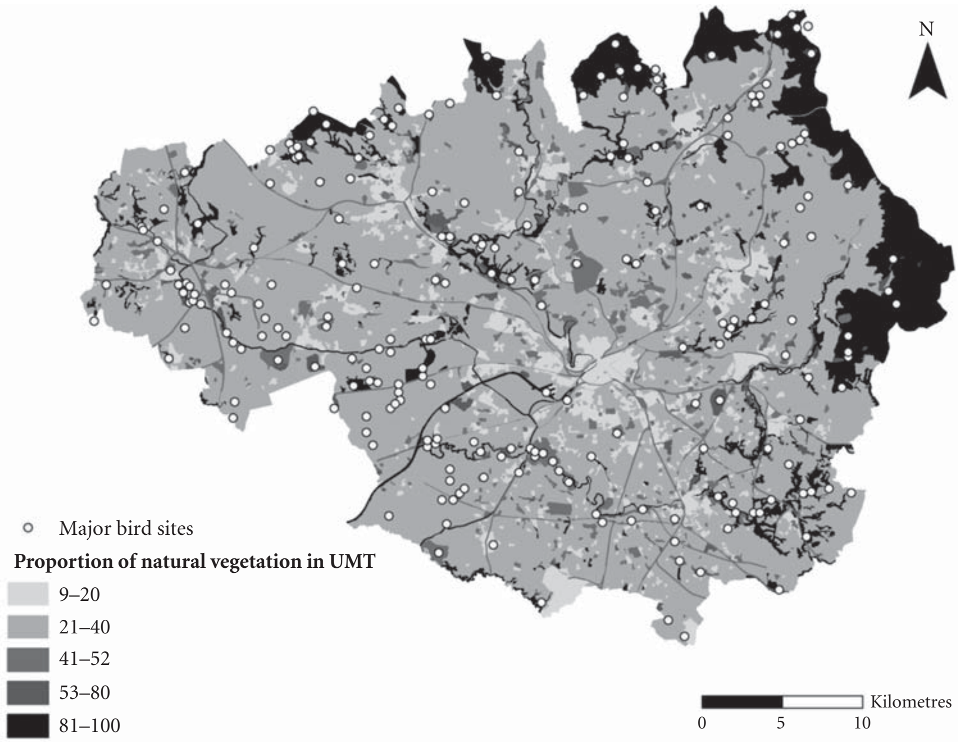

Figure 5.3.6 Association between the proportion of natural vegetation (trees, shrubs, rough grass, and water) in the land cover of Urban Morphology Types (%) and location of major bird sites in Greater Manchester (Smith 2008)

strong support for green infrastructure at the national level in UK impedes the implementation of the concept on the ground. This is somewhat balanced by the inclusion of green infrastructure concept in the Regional Spatial Strategy for the North West (NWRA 2008), which also guides local planning (Gallent & Shaw 2007). The Regional Spatial Strategy explicitly requires a green infrastructure approach to be adopted in all land-use, development, community, economic, and regeneration strategies. Regional Spatial Strategy also states that an integrated approach to biodiversity, landscape, and heritage is essential, rather than approaching them as separate policy areas, in accordance with the green infrastructure thinking. This highlights the importance of the synergies between the two projects in Greater Manchester reported above.

The green infrastructure approach in Greater Manchester was strengthened in July 2008 as the area was declared as a national New Growth Point, pledging to deliver high housing development targets in accordance with the government's vision of sustainable development (TEP 2008). The conditions attached to the government's offer of New Growth Point Status stated that every Growth Point must prepare and implement a Green Infrastructure Strategy to protect and enhance existing key environmental assets and to provide new greenspaces designed to deliver a wide range of environmental and social benefits. Therefore, there is a strong case for creation of coherent green infrastructure plans at the conurbation level, which makes up for the absence of green infrastructure focus in the national policies.

AGMA, in order to meet the green infrastructure requirements of a New Growth Point, commissioned the Environment Partnership consultancy (TEP) to advise how green infrastructure might be embedded into the spatial planning policy and practice in the conurbation. This exercise included translating the concept of green infrastructure into local spatial planning terms, identification of the existing resources, and signposting the priority areas for green infrastructure. Both the ASCCUE project and Ecological Framework, together with other evidence for wider social and economic benefits of green infrastructure, were included in this document. While it was recognized that the existing urban fabric constrains the ways that green infrastructure can be planned, implemented, and managed, TEP emphasized the role of the existing 'skeleton' of green space resource: green belt and nature conservation areas (see Fig. 5.3.2 and 5.3.3), as well as floodplains, parks, and open spaces. TEP recommended that AGMA produces a framework for green infrastructure (a document identifying priority areas for investment and enabling joint programmes) as an early action to influence spatial and infrastructure planning across the whole city region. This should be formalized into a strategy (expansion on the framework document, including a focused series of individually-owned actions, shared across several green infrastructure funding and delivery agencies) once the overall extent and timescale of growth is clear.

5.3.4 Conclusions

Green infrastructure holds considerable potential to advance adoption of ecological practice in urban planning and design. While its roots may be traced back to the visionary landscape architects and thinkers in the nineteenth century, such as Frederik Law Olmsted who designed the 'Boston Fenways' (Turner 1998), it is a major advance compared with conventional green space planning and design which is mostly concerned with aesthetic, recreational, and spatial functions of greenspace. As the first parts of this chapter have shown, green infrastructure planning is indeed strongly informed by ecological theory and principles, such as the concepts of multifunctionality and ecosystem services, habitat connectivity and complementarity. As a planning approach, green infrastructure thinking is related to contemporary planning theory, that is, the collaborative approach to planning. Importantly, green infrastructure recognizes the need for adaptive planning and management of complex urban ecosystems. Therefore, it provides a conceptual framework which appears to be well founded to promote ecologically informed planning in urban areas.

The concept of green infrastructure can help to build bridges between different disciplines. 'Infrastructure' thinking provides a common ground for ecologists and landscape planners to join forces with urban planners, engineers, and social scientists in order to collectively address the major challenges of contemporary urban development, such as the restructuring of waste and stormwater systems, wetland restoration, or the reintegration of wastelands into the urban fabric. Each of the individual disciplines increasingly recognizes the limitations of addressing these challenges on its own. For instance, in the case of Seattle, engineers realized that they alone could not solve the stormwater problem because the conventional approaches for sewage based solutions were coming to their physical and economic limits. This was the starting point for an innovative exercise where planners, hydrological engineers, road planners, and ecologists worked together with landscape architects, and in particular the local community, to come up with an innovative solution for a green infrastructure with multiple benefits. This solution was informed by ecological thinking to retain and infiltrate stormwater and establish a biodiverse landscape.

This case study, which is just one out of many that could have been reported here (Ahern *et al*. 2006), highlights the importance of projects in green infrastructure planning. Unfortunately, rigorous comparative assessments of pilot projects are rare, and this should be one important area for future research. Pilot projects are important not just because they make things happen on the ground but also because they provide opportunities for learning and monitoring. Uncertainty about whether and how new planning and design approaches, such as sustainable urban drainage systems, work in practice are a barrier for their adoption at larger scales. The paradigm shift from combined sewer systems, where engineers and planners have gathered experience of over 150 years, to local retention and infiltration, is not an easy one. It raises many serious questions of technical, social, and economic nature where there is a lack of experience, for instance concerning complex issues related to the management of stormwater quantity and its quality, long-term viability, and management of the implemented solution. Additional questions include whether the new green infrastructure paradigm really improves the quality of the landscape and whether it is acceptable for the local community. These questions cannot be solved in theory but must be answered in practice. The Seattle case study shows that this was a successful exercise from which to learn for application in other places and thus reduce the resistance to move into hitherto unknown territory. This design was not only a way of solving stormwater problems but also achieved wider ecological and social benefits as the street changed from a monofunctional to a multifunctional space. More importantly, the SEA street project as a pilot or model promoted the development of a green infrastructure strategy at city level.

Moving from site projects to strategies at city level is a big challenge. Sound scientific evidence is required to support development of such strategies. The Manchester case study showed how this evidence can be provided (see also Pauleit and Breuste, Chapter 1.1) by complementary research on the potential of green infrastructure to support biodiversity and adapt cities to climate change. One of the most interesting findings from scenario modelling in the ASCCUE project in Manchester was that an increase of green cover in the densely built inner-city could compensate for most of the predicted temperature increases until 2080. The political relevance of this finding should be evident. Indeed, the results of the project have been widely discussed among planners, stakeholders and politicians in the UK North West Region and were an important reference for development of a regional green infrastructure strategy as part of the Regional Spatial Strategy. The results also fed into the forthcoming green infrastructure strategy of the Association of Greater Manchester Authorities. Yet, however important climate change is, it is not the sole issue to be addressed in green infrastructure planning. Hence, the Ecological Framework built complementary evidence on the role of the green infrastructure for urban biodiversity. This study even went one step further in outlining a clear spatial framework for creation or restoration of habitats and their linkage. The case study showed that important synergies exist between these two objectives, thus reinforcing the multifunctional character of the green infrastructure.

Green infrastructure is a relatively new concept. While it's adoption in the North West Regional Spatial Strategy and in Greater Manchester are encouraging signals, the static nature of the planning policies means that it may be a while before the concept is supported by coherent policies at the national level. On the other hand, the momentum created by a number of research studies and the good will of planning authorities mean that green infrastructure is present at the local level where the actual spatial planning takes place. The example of Greater Manchester shows that there is clear evidence that urban green infrastructure contributes to adaptation to climate change and to maintenance of biodiversity. Furthermore, it indicates how the research carried out by local universities and supported by the subregional authorities has influenced planning practice, and how synergies between findings on benefits of green infrastructure could be utilized to support the planning. It also shows that despite ambiguous support for green infrastructure at the national level, the strong championship of the green infrastructure concept at the subregional level, and inclusion of green infrastructure concept into policies on development and growth, has resulted in some decisive steps being taken towards creation of a green infrastructure strategy for the Greater Manchester conurbation.

Green infrastructure is sometimes accused of diluting the ecological principles among other benefits, but the case studies have shown that it can actually work as a bridge between ecology and spatial planning, supported and not jeopardized by other benefits.

CHAPTER 5.4

Building for Biodiversity: Accommodating People and Wildlife in Cities

Jon Sadler, Adam Bates, Rossa Donovan, and Stefan Bodnar

5.4.1 Introduction

To accommodate a rapidly increasing urban population, governments have created a range of policies on urban living and housing provision that appear to be in conflict. On the one hand, they drive local councils to utilize as much brownfield and open space in cities as possible to meet construction targets for new build houses, underlining the move towards the so-called 'compact city' (Burton 2000), whilst on the other hand, policies call for more open spaces to improve the quality of life for city dwellers (e.g. ODPM 2002; EEA 2009). Notwithstanding strong pleas for sustainable development by many organizations (EEA 2009), economic priorities still drive development strategies in cities, with environmental concerns taking a subsidiary role (see also Box, Chapter 5.6). Currently, the policy agenda for build and more build is in the ascendancy as large cities compete to sell and brand themselves as 'global cities', emphasizing their cultural and economic prowess as a means of generating finance to support further development. This has led to increased urban sprawl at city margins (EEA 2006; Irwin & Bockstael 2007), and, in regions with strong green belt policies to restrict city growth (e.g. Europe), it has caused land-use intensification and a loss of greenspace (EEA 2002; Davies *et al*. 2008; Fuller & Gaston 2009, see also Pauleit and Breuste, Chapter 1.1., Colding, Chapter 4.5). Both processes threaten the quality of the urban environment for both people (RCEP 2007) and biodiversity (Tratalos *et al*. 2007).

Establishing how best to manage cities in a way that is both economically and socially acceptable, whilst retaining ecological linkage and functionality, is a key element of the move towards a sustainable twenty-first century city; especially when viewed against the backdrop of global environmental change (Grimm *et al*. 2008). There are a raft of papers considering this issue; many identify the complexity of interactions between the social, political, and natural elements of what has been termed the urban system (e.g. Pickett *et al*. 2001; Williams *et al*. 2009); whilst others illustrate the need to focus attention on individual decisions in the development process (Yli-Pelkonen & Niemelä 2005) and the pivotal importance of the planning system (Niemelä 1999). Much has been written concerning the pervasive impact of urbanization on the type, structure, and location of habitats (Denoel & Lehmann 2006; Garden *et al*. 2007) and its links to the presence, abundance, species richness, and persistence of plant and animal species (e.g. Sadler *et al*. 2006; McKinney 2008). The importance of the ecosystem services (*sensu* Costanza *et al*. 1997) that city habitats provide to urban residents has also been stressed (Tzoulas *et al*. 2007; James *et al*. 2009). Nonetheless, there is a dearth of research that examines the spatio-temporal scale at which development processes are managed and articulated within cities (Borgström *et al*. 2006), and the type of mitigation techniques employed to compensate for habitats lost to this development. This chapter reviews: 1) the science underpinning the management of urban systems, 2) the planning systems and policies that exist to help manage urban areas, and 3) mitigation techniques and tools used to compensate for habitats lost to urban development.

5.4.2 Managing urban systems

A city can be viewed as a complex spatio-temporal mosaic of different land-use types providing living space for people, other animals, and plants (Grimm & Redman 2004). The ecological components of this mosaic, commonly termed 'green infrastructure' or more recently 'green–blue infrastructure', are typically composed of a spectrum of different ecological habitats ranging from relict habitats (e.g. woodlands and parkland) to newly designed ones (e.g. housing and industrial developments), at a range of scales from small (e.g. gardens) to very large (e.g. parklands) (Pauleit *et al.*, Chapter 5.3). Few cities are planned and built as a whole and most have grown over time, consuming and capturing the surrounding countryside, towns, and villages. This historical legacy, coupled with the cyclical nature of renewal, has created a built form that exhibits considerable structural and temporal heterogeneity, and is characterized by markedly different ecological legacies (Niemelä & Kotze 2009). The structural arrangement of habitats (Forman 2008), their size (McKinney 2008), quantity (Mörtberg & Wallentinus 2000), and quality (Angold *et al*. 2006) not only affects the dispersal, occupancy, and persistence of wildlife in the city, but scales to provide important ecological functions (Whitford *et al*. 2001), and ultimately ecological services (Bolund & Hunhammar 1999, McDonald and Marcotullio, Chapter 4.1, Colding, Chapter 4.5).

In cities, people are the arbiters of landscape structure, and the main vehicle for managing its change is the planning system (Niemelä 1999). Planning systems are scalar, and at broad spatial scales (e.g. the city or region) the best master plans do now specifically consider the environment and biodiversity, even if they remain of subsidiary importance to economic and social elements. However, making these broad environmental and biodiversity plans a reality at smaller scales (e.g. the development plot) is fraught with difficulty, given the multitude of competing factors dealt with during the development control planning process.

5.4.3 Planning tools and approaches to urban planning—a UK perspective

In this section we address the varying scales at which planning policies interface with biodiversity within a city. Our focus is on the UK planning system, which we use as an analytical lens for understanding scalar issues and potential pitfalls. However, we believe that the planning issues discussed below are applicable outside the UK as well.

5.4.3.1 National regulatory protection and guidance

Aspirations of sustainability are clearly visible in many aspects of UK planning policy, with a wide range of tools and guidance aimed to help planners manage developments in cities (Table 5.4.1). Planning guidance is provided via a range of policy documents that are used to shape planning decisions at national, regional, and local levels. The main tools available are Planning Policy Statements (PPS) which supersede Planning Policy Guidance (PPGs) notes and which have statutory support. With regards to biodiversity the most significant document in England is PPS 9: *Biodiversity and Geological Conservation* (ODPM 2005), which is supported by Government Circular 06/2005: *Biodiversity and Geological Conservation – Statutory Obligations and their Impact within the Planning System* which provides an interpretation of the policies contained in PPS 9. Biodiversity is also given some consideration in PPS1: *Sustainable Development*, PPS 3: *Housing*, and PPS 7: *Rural Development*. Wales, Scotland, and Northern Ireland have their own separate Planning Policy Guidance notes, which broadly follow the English guidance.

In the UK planning system ecological sustainability is operationalized using the notion of 'environmental capital' (Cowell 1997), subdivided into 'critical' and 'constant' natural capital. Critical natural capital is the term given to those particularly important ecosystems or rare or vulnerable species that are given statutory protection to prevent their loss. Constant natural capital is the term given to the ecological features not given statutory protection, which, importantly, can be traded-off during the development process using different mitigation

Table 5.4.1 U.K. planning policy guidance documents that relate to environmental management of cities

Unit	Protection	Planning policy guidance and legislation
Wider urban landscape	Legal	NERC Act (2006 and 2009)
		PPS 1: *Delivering Sustainable Development*
		PPS 1 (Supplement): *Planning and Climate Change*
		PPS 9: *Biodiversity and Geological Conservation*
		PPS 12: *Local Spatial Planning*
		PPS 25: *Planning and Flood Risk*
		PPS 22: *Renewable Energy*
		NI 185 & 186: *Reduction of* CO^2 *Emissions Per Capita and via Local Authority Operations*
		NI 188: *Adapting to Climate Change*
		NI 189: *Flood and Coastal Risk Management*
		NI 194: *Level of Air Quality*
		NI 197: *Improved Local Biodiversity*
	Advisory	Supplementary Planning Guidance (SPG)
		Supplementary Planning Documents (SPD)
		Biodiversity Action Plans
		Habitat Action Plans
Individual sites	Legal	PPS 9: *Biodiversity and Geological Conservation*
		PPG 17: *Planning for Open Space, Sport and Recreation*
		NI 197: *Improved Local Biodiversity*
		NERC Act (2006 and 2009)
	Advisory	Supplementary Planning Guidance (SPG)
		Supplementary Planning Documents (SPD)
		Local Biodiversity Action Plans
		Local Habitat Action Plans
Taxa	Legal	Conservation (Natural Habitats &c) Regulations 1994
		Wildlife and Countryside Act (1981)
		Countryside Rights of Way Act 2000 (England and Wales)
		Nature Conservation (Scotland) Act 2004
		Wildlife (Northern Ireland) Order, 1985 (Amended 1995)
		Conservation (Natural Habitats &c) Regulations 1994
	Advisory	Species Action Plans

and compensation measures. This usually manifests itself spatially within the planning system at a plot scale, when Planning Officers are making decisions on individual development sites. It is less frequently used at larger spatial scales.

In addition to planning policy, various pieces of legislation provide differing degrees of protection to certain species, habitats, and important sites (Table 5.4.1). Legislation set at European level (e.g. Habitats Directive 1992; Conservation (Natural Habitats &c) Regulations 1994 as amended; Birds Directive 1979 as amended) which apply within the European Union economic area are transposed into the UK legislation through the Wildlife and Countryside Act 1981 as amended by the Countryside and Rights of Way Act 2000 (see Table 5.4.1 for a more complete list of legislation). Other pieces of UK legislation (e.g. The Protection of Badgers Act 1992) deal with the protection of individual species or groups of species. Together these Acts list species and habitats which are given legal protection from various acts, damage, and destruction. The Countryside and Rights of Way Act 2000 provides identical legislative protection for both England and Wales; while separate Acts apply to Scotland (Nature Conservation (Scotland) Act 2004) and Northern Ireland (Wildlife (Northern Ireland) Order 1985 (Amended 1995)). The Natural Environment Rural Communities Act 2006 places a greater onus on the Local Planning Authority to protect and manage biodiversity resources within their area and lists priority species and habitats in England and Wales.

Recently, the way in which local authorities are required to report to the UK Government has changed to a streamlined model of reporting against a sequence of National Indicators (DCLG 2007). National Indicator NI 197 relates directly to improving biodiversity, while others emphasize sustainability in its widest sense (Table 5.4.1). In practice, local responses to NI 197, based on National Government Guidelines, consist of recording and evaluating changes in the number of local sites designated for nature conservation and geological importance that are under positive management (i.e. the proportion of sites judged to be managed in a manner that sustains the biodiversity reasons for designation). This is the first attempt to compare performance on biodiversity in a simple recording manner across Local Authorities, but it is not without problems. The main issue is that councils only have the ability to influence land management of areas under their direct control. Privately owned land is managed differently and this creates widely varying baselines in measurement.

Signatories to the Convention on Biodiversity (CBD) are also required to create national Biodiversity Actions Plans (BAPs), which include Species Actions Plans (SAPs) and Habitat Action Plans (HAPs) for priority species or habitats. Following the most comprehensive review ever undertaken the list, which covers the whole of the UK, includes 1,150 priority species and 65 priority habitats (http://www.ukbap.org.uk/NewPriorityList.aspx). A lead partner, which can be a statutory (e.g. Environment Agency) or a non-statutory organization (e.g. a Conservation NGO), formulates each HAP or SAP. While there is no automatic legal protection inferred, the presence of a BAP species or habitat on a development site should be a material consideration within the planning decision-making process. Although, it should be stressed here that the BAP process is currently undergoing a systematic review, so the situation is certainly set to change in the near future.

The UK Joint Nature Conservation Committee (JNCC) select and schedule areas of habitat as Special Sites of Scientific Interest (SSSIs), National Nature Reserves (NNRs), and Special Areas of Conservation (SACs), which are overseen by Natural England, Scottish Natural Heritage, and the Countryside Council for Wales, which are government funded, statutory organizations. These habitats are also afforded some level of protection against development. Very few of these, however, are found in urban areas. For example, Birmingham and the Black Country, the UK's second largest conurbation, has only three SSSIs.

5.4.3.2 Regional and local guidance

At a local council and regional assembly level, Supplementary Planning Guidance (SPG) and Supplementary Planning Documents (SPDs) (Table 5.4.1) can be used to create Sites of Importance for Nature Conservation (SINCs), Sites of Local Importance for Nature Conservation (SLINCs), and Local Nature Reserves (LNRs). In practice, however, status can be misleading and in some cases designation as a Local Nature Reserve may confer a greater level of site protection than national designations such as SSSI. In addition, broad planning policies concerning biodiversity, such as nature conservation strategies (e.g. BCC 1997) and those that have influences on land management, such as the Sustainable Management of Rivers and Floodplains (SMURF) SPD (2006) and the Parks Strategy SPD (2006), can be created and used to help guide the broad direction of future development. All of these are used in the planning system as a means of requiring developers to be mindful of biodiversity on their proposed sites and can, in some cases, be used to set planning conditions on a site. These planning conditions, however, rarely relate to preservation of important habitats, but rather mitigation of some kind as compensation for their destruction. If a strong economic case for development is provided, then site destruction is a likely outcome, although stakeholders in the local communities, such as conservation NGOs, do have a right of appeal against any planning decision.

Three issues arise from this discussion. First, is it clear that the UK planning system, as it interfaces with biodiversity, is hierarchical in nature and also somewhat fragmentary. There is a range of international and national legislation protecting certain species and habitats and a wider number of tools that can be used to 'guide' the planning process. Developers are only strictly bound by the legislation; other planning tools are negotiated at the time application with the planning officers. This frequently leads to inconsistency in terms of their application. Second, planning decisions that lead to landscape changes operate at the scale of an individual development and are, therefore, often site specific. This means cities frequently develop in a piecemeal fashion through the development planning decision-making process, despite the presence of regional planning guidance that provides a coherent vision for biodiversity at a higher spatial scale. Third, in the UK planning applicants have a right of appeal against the local planning authority if the authority refuses permission, so local decisions can be over-ruled even when planning decisions are made which support a coherent biodiversity strategy.

5.4.4 Building and managing biodiversity: mitigation techniques and habitat enhancement

Urban greenspaces can usually be considered constant environmental capital and are, therefore, available for new development, with the developer offsetting the habitat loss with the creation of habitat of similar quality. In reality, an environmentally sustainable (*sensu* Callicott & Mumford 1997), like-for-like replacement of habitat resource is rarely attempted or achieved, instead the available biodiversity mitigation techniques help move a new development towards sustainability. That aside, we believe that the most ecologically sustainable mitigation techniques need to fulfil two key criteria: 1) they have to align with people's sense of scenic beauty and amenity, and 2) they have to be considered throughout the development and management process, not just as an afterthought or bolt-on.

Many of the most important and biodiverse urban habitats are associated with negative social characteristics and do not easily appeal to the public sense of amenity and scenic beauty (Parsons 1995; Harrison & Davies 2002; Özgüner & Kendle 2006; Gobster *et al*. 2007; McDonnell & Hahs 2008). Human perceptions of landscape aesthetic beauty are intuitive and unconscious, being set by cultural identity and evolutionary preferences for Arcadian, Savannah-like environments that afford prospect, refuge, and easy way-finding. This means that they are deeply ingrained, and very difficult to alter (Nassauer 1995; Parsons 1995; Parsons & Daniel 2002; Gobster *et al*. 2007). There is also a tendency for most people to positively associate these landscape preferences with ecological good health and value even when this is not the case (Nassauer 1995; Gobster *et al*. 2007). This is doubly problematic because in order to be successful, and therefore truly sustainable, the public must appreciate and value biodiversity mitigation and enhancement schemes (Harrison & Davies 2002; Hunter & Hunter 2008).

If urbanism and nature are to successfully coexist, it is imperative that schemes that aim to enhance biodiversity counterbalance preference for Arcadian landscapes with: 1) community planning involvement and environmental education that foster a greater appreciation of the importance of natural diversity; and/or 2) designs that try to reach a mutually beneficial compromise between the scenic and ecological aesthetic of developments (Harrison & Davies 2002; Jim & Chen 2006; Gobster *et al*. 2007; Hunter & Hunter 2008). At the very least, even in areas designated as natural habitats, it is always essential to show evidence of care (Özgüner & Kendle 2006; Gobster *et al*. 2007; Özgüner *et al*. 2007; Hunter & Hunter 2008). The development and refinement of these approaches is underway (e.g. Dunnett & Hitchmough 2004; Hunter & Hunter 2008), but it is clear that both the public and professionals remain to be fully convinced about the value of more ecologically sensitive landscape designs (Özgüner & Kendle 2006; Özgüner *et al*. 2007). This is especially the case for less popular species groups such as insects (Hunter & Hunter 2008) and habitat elements that are perceived as particularly unattractive or dangerous, such as brownfields, erosive features, marshlands, and deadwood (Harrison & Davies 2002; Gobster *et al*. 2007).

The other key theme relevant to biodiversity design measures is that the consideration of biodiversity measures is necessary throughout the site development and management process (Harrison & Davies 2002). We suggest five main reasons for this:

(1) It is necessary in the first instance to be aware that there is on-site biodiversity interest and have some understanding of that interest. Developers are often unaware that there is any biodiversity interest on site, and this is especially the case for habitats perceived as aesthetically unattractive, such as brownfield sites. In order to get a complete picture of the biodiversity potential of a site, surveys have to be done during different seasons for different groups of organisms, therefore the lead-in time for adequate biodiversity surveys is long.

(2) Development will need to take place at times of the year when target species do not have vulnerable stages in their life-cycles (e.g. trees in leaf, nesting birds, migratory butterflies). This requires an early awareness of the species on site and a careful integration of species and development lifecycles.

(3) If it proves necessary to preserve some of the existing habitat *in situ* or to put aside topsoil or plants from the development site for later use in on-site habitat recreation, this space will need to be incorporated into the design of the development at an early stage. Furthermore, it is not possible to preserve habitat *in situ* if it has already been destroyed at an early stage of the development process.

(4) If biodiversity conservation has been a consideration throughout the development process, mitigation measures are less likely to be removed at a late stage of the development due to budgetary overspend earlier in the development process.

(5) If the post-development management of the site is inappropriate, however carefully designed the on-site biodiversity conservation measures, their value will be compromised or destroyed.

In the 'business as usual' mechanistic developmental process, issues of timing are of far less importance, when a roof needs to be built, a roof can be built; when foundations need to be laid, foundations can be laid, and this is almost regardless of season and natural disturbances. However, it is, for example, impossible to recreate a species-rich brownfield habitat with several stages of succession supporting several rare species when called for (c.f. Kayes *et al*. 1993; Donovan *et al*. 2005). The timeline of the mechanistic development process is, therefore, asynchronous with that necessary for ecological sustainability, because habitats and ecological communities are driven by cycles of colonization, succession, disturbance, and climate, which are all time-dependent. The necessary level of integration between ecological and developmental timelines, therefore, requires ecologists to be part of the initial development team (Harrison & Davies 2002), with a similar level of authority as the project managers, engineers, architects, planning consultants, surveyors, contractors, and environmental legislators that are more traditionally associated with the development process.

Full inclusion of ecological guidance in the development process is not only more ecologically sustainable, but, contrary to the thinking of most 'business as usual' developers, can also actually benefit the developer. Discovering the presence of legally protected species late in the development process can result in expensive design changes and delays that can significantly increase development costs. Protests from environmentalists and the local public who feel that the biodiversity interest of a development site has been ignored can do likewise (Harrison & Davies 2002). Demonstrating to the public, environmental legislators, and environmentalists that safeguarding the biodiversity interest of a site is a high priority can help forestall these problems and provide positive publicity for the developer.

5.4.5 Building for biodiversity: constructing a more ecologically sustainable built form

A myriad of green technologies now exist that can be used for restoration and habitat creation; ranging from basic nest and roost boxes to broader scale initiatives, such as permeable pavements, living walls, and green roofs. Similarly, in parks and other public spaces (e.g. woodlands, canal corridors, and roundabouts) there is potential for modifying management strategies to enhance habitats. However, there has generally been only limited interchange between ecological research, practical conservation (Sutherland & Hill 1995) and practical ecological design, and biodiversity enhancement measures have rarely been tested systematically (but see Gee *et al*. 1997; Biggs *et al*. 2005; Gaston et al. 2005; Conservation Evidence: www.ConservationEvidence.com). An extensive consideration of all possibilities would run to many chapters. Therefore, we limit our discussion to a contrast of easily achievable generic and site-specific design solutions, before describing green roofs and how the key design issues described above can be incorporated into the development process.

5.4.5.1 Generic versus site-specific biodiversity mitigation techniques

There are a huge range of possible biodiversity mitigation techniques available both for the built fabric and the wider plots of new developments. The techniques vary from off-the-shelf generic solutions that can be installed quickly with little or no interference in the planning and design process, to site-specific tailored solutions that will often require ecological design input and consideration in the survey, planning, and design process (Table 5.4.2). The amount of available information in the popular 'grey literature' varies widely according to the biodiversity measure. There is, for example, a wide literature on broad brush 'gardening for wildlife' and garden pond design (e.g. Hallis 2000; Kelsey-Wood & Barthel 2006; Tait 2006; Nottridge 2009); but relatively little on, for example, creating bat roosts or bee (except the Honey Bee) nesting habitat. The information contained in this literature, although often sound, is usually based on anecdotal observations, and has not usually been systematically tested (Biggs *et al*. 2005; Gaston *et al*. 2005). When design and management recommendations are more rigorously tested they can prove erroneous in some circumstances, for example, the supposed detrimental effect of shade on ponds (Gee *et al*. 1997; Biggs *et al*. 2005), or the effectiveness of different types of artificial bumblebee nests (Gaston *et al*. 2005).

This weakness of the grey literature is not solely due to poor communication between ecological research and practical conservation; it is because very few biodiversity mitigation techniques have actually been systematically tested (Sutherland & Hill 1995). This is especially so for artificial habitat resources which, in contrast to more traditional semi-natural habitats (e.g. woodland, heathland, and fenland), have often been seen as of limited conservation potential, when this is not necessarily the case. For example, recent research has shown that types of artificial pond traditionally considered of limited biodiversity importance because of high pollution levels or small size, such as ponds on golf courses and motorway stormwater retention ponds, can have very similar biodiversity importance to other, more 'pristine', ponds (Colding *et al*. 2009; Le Viol *et al*. (2009)). Owing to this lack of systematic research, it is difficult to make robust detailed predictions about the efficacy of the various available mitigation techniques. Nonetheless, we argue that tailored site-specific solutions will in most cases out-perform generic off-the-shelf solutions in the provision of ecological value.

One of the most common types of mitigation technique in the construction of the built fabric is the artificial provision of nest or roosting sites (Table 5.4.2). Direct evidence is limited, but for both bats and birds, large and rapid changes in temperature in the nest or roost site will likely reduce both occupancy rates and the rate of successful rearing (Du Feu 2002; Forestry Commission 2005). Therefore, where possible, site-specific solutions like the *in situ* preservation of trees with rot holes, concreted sawdust specialist boxes, or in-built nest bricks, recesses, and access holes in buildings, which can all reduce the range of nest temperature, should be favoured. Another adaptation of the built fabric is permeable paving, which although designed for use within more sustainable, integrated urban water management (IUWM) schemes, can be used to enhance biodiversity value above that of more traditional impermeable paving methods. More tailored approaches, such as using carefully chosen robust species mixes of grass for low vehicle traffic; or targeted paving, where paved areas are restricted to only those necessary to support vehicle wheels, which allows the planting of low-growing robust herbs inbetween; will increase the biodiversity value.

The wider plot of built developments provides the most opportunity for building for biodiversity, but there is a current trend to favour generic low-maintenance, aesthetically replicable planting schemes that have low biodiversity value (Donovan *et al*. 2005). This is another symptom of the low level of ecological input in the development process, and results in the replication of sterile, overly tidy habitats that mask local cultural and ecological character. Site-specific solutions, like the *in situ* preservation of mature trees, and bee banks, or landscaping with regionally appropriate planting schemes, however, can better preserve biodiversity interest and the visual character of the local vegetation (Table 5.4.2). In many instances, this

Table 5.4.2 Mitigation techniques available to maximize biodiversity during the building process (Sources: Du Feu 2002; Dunnett & Hitchmough 2004; Dunnett & Kingsbury 2004; Forestry Commission 2005; Gregory & Wright 2005; Rose & O'Reilly 2006; Sutherland & Hill 1995)

Biodiversity measure	Generic solutions	Site-specific solutions	E.g. U.K. species/habitat beneficiary
Built fabric			
Bee nests – tube nesters	nest box	nest brick, aesthetic design	*Osmia ruf, O. caerulescens, Megachile centuncularis*
– ground nesting Bumblebees	nest box	—	*Bombus terrestris, B. lapidarius, B. vestalis, B. rupestris*
Bird nests – song birds	(a) nest box	(b) specialist nest box/brick	(a) *Parus major* & *Cyanistes caeruleus* (b) *Motacilla alba, Passer domesticusz Phoenicurus ochruros*
– raptors	(a) nest box	(b) tall building nest box/recess	(a) *Falco tinnunculus,* (b) *Falco peregrinus*
– owls	nest box	—	*Strix aluco, Athene noctua*
– specialist eave nesters	nest box	nest brick, access holes to indoors	*Delichon urbica, Hirundo rustica, Apus apus*
Hedgehog boxes	box	—	*Erinaceus europaeus*
Bat roosts	bat box	(a) bat bricks/tubes or access holes	(a) *Pipistrellus pipistrellus Plecotus auritus Myotis nattereri*
Living walls	—	irrigated vegetation mat, hydroponics	*Sedum, Dianthus Sempervivum, alpinum*, Insect pollinators
Green roofs	(a) *Sedum* mat, meadow mat green roof trays	(b) intensive green roof, eco-roof brown roof	(a) *Sedum Silene, Alium, schoenoprasum, Primula veris* (b) garden flora, brownfield, grassland, flower meadow
Permeable paving	block paving, porous asphalt	grass paving, targetted paving	mosses *Sedum, Thymus polytrichus, Chamaemelum nobile,*
Wider plot			
Bee banks/scrapes	—	*in situ* preservation, creation	*Andrena clarkella, Halictus tumulorum Crossocerus pusillus*
Set-aside areas – mature trees	—	*in situ* mature tree preservation*	Birds *Strix aluco, Certhia familiaris Garrulus glandarius* Bats *Nyctalus noctula, Myotis nattereri Myotis daubentoni*
– scrub, log piles	—	pocket set-asides	*Erinaceus europaeus*, insect pollen/ over-wintering resource
Landscaping – trees	(a) trees – replicable aesthetics	(b) food-bearing, local provenance	(a) *Tilia x europaea, Platanus x hispanica, Sorbus aria* 'Lutescens' (b) *Prunus avium, Quercus robur, Sorbus aucuparia*
– borders	(a) low maintenance shrubs	(b) food-bearing, local provenance	(a) *Hebe albicans, H. franciscana Mahonia, x media, Cornus alba* (b) *Prunus spinosa, Rosa arvensis, Ulex europaeus, Ilex aquifolium*

(*Continued*)

Table 5.4.2 Continued

Biodiversity measure	Generic solutions	Site-specific solutions	E.g. U.K. species/habitat beneficiary
– grassland	amenity grassland	(a) spring/summer meadow strips	(a) *Primula veris, Rhinanthus minor, Leucanthemum vulgare*
	—	(b) wildflower lawn	(b) *Prunella vulgaris, Achillea millefolium, Trifolium pratense*
Ponds & IUWM^ – (retention) ponds	pre-formed plastic ponds	(b) 'soft-edged' ponds & reedbeds	(b) *Triturus cristatus, Ischnura elegans, Anisosticta 19-punctata*
– swales, retention basins	—	pocket wet meadows	*Filipendula ulmaria, Angelica sylvestris, Mentha puleguim*

* valuable for most species of bats including those that often roost in buildings.
^ IUWM—integrated urban water management.

additional ecological input into the development process can reduce expenditure in the construction and management phases of a project. For example, traditional amenity grassland is characterized by high fertility levels and a frequent mowing regime, which have significant associated costs for topsoil and labour. More biodiverse grassland is supported by low fertility levels, with a less frequent mowing regime (Baines 1995). By using existing on-site low nutrient substrates and leaving dedicated strips of spring or summer meadow, and by instigating longer pauses in summer mowing to allow wildflower lawns to flower, potential exists for the development to save money, provide visual interest, demonstrate evidence of care, and still enhance local biodiversity.

5.4.5.2 Vertical displacement of habitats: green roofs as mitigation for habitat loss

Green roof is a broad term for a roof covered in a growth medium with plants growing on it. The uptake of green roof technology in many countries has increased recently as a result of a variety of potential environmental benefits, including: improved building thermal performance, urban cooling, removal of air pollution, and reduced stormwater runoff (Mentens et al. 2006; Oberndorfer *et al*. 2007; Yang et al. 2008). They vary hugely in their character, from intensive green roofs, which are usually heavily landscaped 'gardens' that are characterized by deep (>20 cm) growth substrates and a rigorous maintenance regime; to extensive green roofs that are characterized by shallow (2–20 cm) growth substrates that require infrequent maintenance (Oberndorfer *et al*. 2007). One type of extensive green roof that is potentially of particular value when building for biodiversity in the development process, are roofs that are designed to emulate old demolition and post-industrial brownfield habitat at an early stage of succession (Grant et al. 2003; Kadas 2006; Bates *et al*. 2009), which are termed 'brown roofs' in this chapter.

Brownfields often provide important centres of biodiversity and species rarity not only locally within an urban area but sometimes also across regional and national scales (e.g. Gilbert 1989; Spalding and Haes 1995; Eversham *et al*. 1996; Small *et al*. 2003; Muratet *et al*. 2007). They are increasingly viewed as habitats deserving conservation protection, both to conserve the habitat itself and to safeguard wider urban biodiversity and habitat connectivity (Harrison & Davies 2002; Donovan *et al*. 2005). However, the redevelopment of brownfield habitat is considered by many as a key element of sustainable regeneration and is often encouraged using funding initiatives and tax breaks (e.g. Pediaditi *et al*. 2005; Thornton & Nathanail 2005). Therefore, brownfield communities are under considerable development pressure in many cities (Harrison & Davies 2002), and striking a balance between the ecologically sustainable goal of conserving urban biodiversity and the social benefits of redeveloping 'wasting' assets is difficult. One potential solution is to use brown roofs to mitigate for brownfield habitat loss on the ground.

Brown roofs are a relatively new concept and their true value as mitigation for the loss of brownfield habitat is still unclear (Grant *et al*. 2003; Bates *et al*. 2009). Nonetheless, knowledge gathered from research on the ecology of brownfields (Small *et al*. 2003; Donovan *et al*. 2005) and brown roofs (Brenneisen 2006; Kadas 2006) suggest important elements that a brown roof designed to mitigate for the loss of on-site brownfield habitat should include:

(1) The growth substrate should closely reproduce the character of the original brownfield soil in its size distribution, nutrient content, and diversity. This will usually require the growth substrate to have low nutrient levels and a largely bimodal size distribution, which will help prevent the dominance of a few highly competitive plant species, and provide cover for invertebrates due to the presence of coarse sediment. It will also require the use of several types and grades of substrate so that a range of the original brownfield microhabitats can be recreated. It will often not be possible to use soil from the existing brownfield site because of contamination issues and a too high a proportion of silt and clay in the original brownfield soil. Silt and clay can block the filter layer on green roofs causing waterlogging, which is unfavourable for plant growth and will increase loading (FLL 2002; Dunnett & Hitchmough 2004).

(2) The brown roof plant assemblage should as closely as possible replicate that found at the brownfield site. If topsoil which contains the original seed bank supplemented by seeds collected on site can be used, this is ideal (Brenneisen 2006), as the plants will be of local provenance, which will increase survival probability and the conservation of genetic diversity (Wilkinson 2001; Bischoff *et al*. 2006), and will more closely reproduce the original plant assemblage. If topsoil cannot be used due to contamination issues, seeds should be collected on site and supplemented with seeds or plug plants from a reputable domestic supplier.

(3) The brown roof should provide disturbance refugia that can provide protection during extended dry periods and the winter. These refugia can include pieces of dead wood and large stones, but should most importantly include deep areas of substrate (10–20 cm) that will retain moisture for long periods and support more luxuriant plant growth that will provide grass tussocks and dead stems for invertebrate over-wintering habitat.

This 'best practice' brown roof development process will require careful design input at several stages in the development timeline that are illustrated in Fig. 5.4.1. After the acquisition of the site, the survey phase should carefully evaluate the conservation value of the existing brownfield site. The value of this habitat survey will vary seasonally and so will require a long lead-in time to properly instigate. This information should feed into planning and visioning, and if the brownfield habitat has high conservation value a decision may be made to include a brown roof on the development. Once decided, an assessment of the most valuable and critical habitat elements (with an understanding of the limits of green roof technology, for example, it is very difficult to reproduce early succession woodland on a green roof) can be made, and this information used to produce brown roof design criteria for the development. It is essential that the development team is aware that a brown roof will be part of the development before the design phase, because extra roof support may be needed to allow sufficient substrate depth to provide refuge during drought disturbances. In addition, green roofs alter the runoff pattern from roofs, moderating the extreme flows (reducing the 'spikyness' of the hydrograph), and thereby reducing the necessary diameter of downpipes and drains and saving on material costs (Mentens *et al*. 2006).

Displaying strong evidence of having fully incorporated conservation objectives into the planning and design process will help the development gain planning approval by helping to convince planning officials and public consultees that the developer is taking ecological sustainability seriously. If approved, the collection of seed and topsoil, before and during site clearance, can be implemented, and the brown roof can be installed during the construction phase. In the management phase the brown roof will need limited maintenance to remove species with strong roots and rhizomes (e.g. *Buddleja davidii*, *Betula* spp.), ensure drains are clear, and ensure the 'fines free' firebreaks remain unvegetated (FLL, 2002). Therefore, a best practice brown roof installation includes consideration of the brown roof at all stages of the development timeline (Fig. 5.4.1).

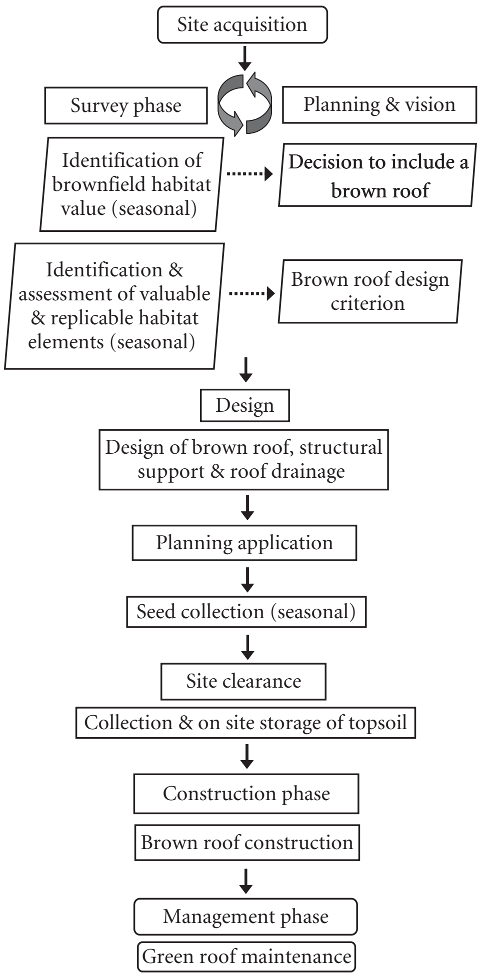

Figure 5.4.1 Best practice development timeline for the installation of a brown roof

A well-designed brown roof should be characterized by many of the habitat elements of a brownfield site, and, therefore, should have areas of bare ground, areas of 'rubble', dead stems, and a predominantly 'weedy' flora. Reconciling ecological and aesthetic value is, therefore, particularly difficult in this instance. To some extent, by moving the habitat to the roof, the brownfield type habitat has been moved out of sight of many. However, occupants of the new build and other nearby buildings will often overlook the site, so it is important to make the roof as aesthetically pleasing as possible without compromising the roof's ecological value, and there are several ways of doing this. First, expectation management is important. Terming the new roof a green roof, or rooftop garden, or showing example photographs of similar roofs at their aesthetic peak in early summer, will give a tenant unrealistic aesthetic expectations of how the roof will look. A tenant will often be dissatisfied with the roof's appearance and believe that it has been badly managed, or 'gone wrong'. By terming the new roof a brown roof, and explaining that the roof will go through cycles of growth and dieback, and is deliberately designed for conservation goals, the tenants' future acceptance of the roof aesthetic will be made more likely. Secondly, design elements can be incorporated that improve the roof aesthetic at key periods that are deemed unattractive. For example, bright annuals can be used to provide colour and visual interest in the first year before most perennials have been able to flower, and different coloured aggregates can be used to create interesting designs that will provide visual interest in areas of bare ground and during winter dieback. Finally, without compromising the ecological value of the brown roof, areas of more traditional green roof, that are designed to be aesthetically pleasing all year round, can be installed on the roof. In this way, carpets of *Sedum* sp., Chives and House Leeks can provide visual interest, despite their lower biodiversity potential. So, even in the most unlikely situation, it is possible to go some way to marrying ecological and aesthetic goals.

5.4.6 Conclusions

There are multiple social and institutional barriers to building for biodiversity in cities because they were constructed without biodiversity in mind. The whole concept of building for biodiversity is itself a 'bolt-on' to city planning and legislative structures that were largely created at a time when nature was ignored or seen as an adversary to be subdued. Recent understanding of the value of green infrastructure has led to an increasing realization that greenspaces are not only valuable for plants and animals, but also for people (James *et al*. 2009). To fully benefit from linked habitat networks and ecosystem services cities must be managed at large spatial scales (Borgstrom *et al*. 2006), however, this is made very difficult because development decisions are usually made at the local scale.

The development of truly ecologically sustainable cities requires the wide-scale alteration of physical habitat, legislature, and cultural perceptions, and is, therefore, an enormously challenging problem that seems a distant and ambitious goal. However, small-scale improvements can be made now using appropriate planning and mitigation techniques and cityscapes can be managed in a more ecologically sensible manner using careful design frameworks (e.g. Pickett & Cadenasso 2008). The first steps towards the sustainable city are clearly visible, even if the whole road bends out of sight.

Acknowledgements

We thank the book and section editors for their valuable comments on the text and their patience and support while it was written. The work and research in this paper has been funded by EPSRC grants EP/E021603 and EP/007426/1, an award from The Big Lottery Fund via the Open Air Laboratories (OPAL) network and the European Union through the UNESCO SWITCH sustainable urban water project. We are grateful also for comments and discussions with colleagues within our respective institutions.

CHAPTER 5.5

Linking Social and Ecological Systems

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5.5.1 Introduction

On 16 November 2005 a water sample was taken from an urban stream in a metropolitan area in the southern United States and tested for the presence of *E. coli.* Although water samples from this and other streams in the metropolitan area frequently registered over 15,000 colonies/100 ml, this particular sample is unique in that it registered a reading of 70,000 colonies/100 ml, 350 per cent greater than the 200 colonies/100 ml—the Environment Protection Agency's standard for streams. The fetid floodwaters in New Orleans from Hurricane Katrina, which had a contamination level of 10,000 colonies/100 ml and attracted considerable public attention, were cleaner than this stream at the time of sampling.

Although a number of factors can contribute to this high reading, the stream consistently failed to meet water quality standards throughout the year. In addition, children from the local neighbourhood often played in the stream. Yet, presentations on water quality issues and potential health hazards did not raise any concerns among the citizenry, news outlets, and policy-makers.

Obviously, there appears to be a disconnection between social and ecological systems as reflected by the lack of concern by residents, natural resource managers, and decision-makers to the degraded stream conditions. This disconnection suggests the following question: How are social and ecological systems linked in urban landscapes and how does one begin to examine that linkage? In this chapter, we explore the linkage between the ecological and social systems of urban landscapes. First, we examine two metrics—sense of place and land cover—that have been used to integrate social and ecological systems. Second, we examine how system models have been used to link ecological systems with social systems. Third, we introduce the concept of complex adaptive systems (Gunderson & Holling 2002), as it may apply to urban landscapes, and finally we present a socio-ecological model (Morse 2007), based on complex adaptive systems and structuration theory (Stones 2005), as a means to link social systems with ecological systems.

5.5.2 Socio-ecological integrators

Westley *et al*. (2002) eloquently discuss how ecological and social systems are quite different and that the systems may not be as congruent as ecologists would like them to be. There are several reasons for this difference. Ecological systems are characterized by time and space. Social systems are too characterized by time and space, but there is also a third dimension—'structure of significance' (Westley *et al*. 2002). Structure of significance refers to the ability of humans to construct and manipulate symbols, principally words, thus collectively inventing a reality that may or may not reflect true conditions. Human actions and decisions are influenced by this structure of significance. In our water example, conditions may not be perceived by individuals as badly as the actual condition of the stream, thus no action. Although there is ecological change, as reflected by water quality, there is no social response.

This does not mean that social and ecological systems cannot be linked in an urban landscape. For example, the Baltimore Long-Term Ecological

Research program (LTER) has taken a patch approach to characterize social and ecological systems (Pickett *et al*. 1997). Pickett *et al*. (1997) proposed that by defining the urban landscape through social and ecological patches one can overlay the different patch types and examine how social and ecological systems are related. To accomplish this approach, Grove *et al*. (2006) used PRIZM, a marketing classification system, to define social patches and vegetation cover to characterize ecological conditions of riparian habitat, private lands, and right-of-ways. PRIZM categorizes people into lifestyle clusters based on household education, income, occupation, race/ancestry, family composition, and housing (Claritas 1999).

Grove *et al*. (2006) report that standard variables, such as income and education, did not explain variations in vegetation cover of the selected habitats. Likewise, they observed that population density was not a good predictor of vegetation cover, a social metric often used to characterize social conditions. Grove *et al*. (2006) did observe that lifestyle behaviour was the best predictor for vegetation cover of private lands, and housing age was significantly associated with vegetation cover for each of the selected habitats. They also reported that social stratification was a better predictor of potential vegetation cover, whereas lifestyle behaviour was a better predictor of present vegetation cover.

The aspect of scale is especially problematic in socio-ecological research. For example, at the fine-scale level, individual decisions affect the context in which ecological structure and function occur. Yet, many of the policies regulating management decisions are implemented at the broader scale. Grove *et al*. (2006) illustrate this interplay of scale of fine-scale decisions and broad-scale management with respect to social systems. Lifestyle of landowners influenced not only the vegetation on their property but also on right-of-ways (managed lands), which are governed by a broad-scale management plan.

The use of PRIZM information to define social patch types may be effective as a site specific analysis; however, cross-site analyses may be limited without further characterization of environmental attitudes, perceptions, and behaviours of marketing classes regionally, nationally, and internationally. Do the environmental attitudes, perceptions, and behaviours of a marketing class vary regionally? Are they the same across a nation? Do they differ among nations? With further research, databases like PRIZM may provide insights into how social and ecological systems are integrated in urban and urbanizing landscapes.

Cross-site analyses provide an opportunity to compare how social and ecological systems are similar or dissimilar across urban areas. Both sense of place and land cover types have been used in cross-site analyses to evaluate how socio-ecological systems vary among urban areas, and will be examined in greater detail below.

5.5.2.1 Sense of place

The Millennium Ecosystem Assessment (2003) defines 'sense of place' as one of the non-material, cultural services provided by ecosystems. It follows then that when ecosystems or landscapes are altered to a measurable degree, the net gain or loss to cultural services should be also altered, as would other provisioning or regulating services like food, water, or climate. Arguably, a construct such as sense of place is more difficult to gauge than other services because the former depends to a greater extent on human perception. Still, these ecosystem services are articulated strongly in instances of ecosystem and landscape change. Bengston *et al*. (2005) remark: '[a]t the local level,... the core of the debate about sprawl... is the emotional impact people experience when they lose places in their own communities they feel deeply attached to'.

Sense of place is both a conceptual and an empirical approach to assess humans' emotive and cognitive, non-tangible, cultural connection to place (Relpf 1997). Fundamentally, sense of place refers to people's interpretation of a place and their resulting identification with the same. The domains of sense of place include place attachment (self-identity related to place), place satisfaction (attitudes toward place), place meanings (descriptive of why the place is important), and place characteristics (environmental attributes) (Stedman 2002, 2003). We assume that sense of places varies by perceiver and that attachments are imparted to a place based on people's experiences with places. Meanings are not necessarily inherent in a place but are assigned and may vary accordingly among individuals or groups, much like Westley's *et al*.'s (2002) significance of structure. Although we regard sense of place as socially constructed, we also assume that there are more generally held interpretations of place that can be discerned by socio-demographic groups or other meaningful subgroups. Because of the subjectivity of sense of place, it is also assumed to be dynamic, continually changing and evolving based on structural drivers such as changing demographics, political influence, or natural change. Place definitions, even at a given point in time, are open to multiple interpretations, but once a standard has been established it is repeatable; hence, its inherit application to cross-site analyses (Jorgensen & Stedman 2006). Sense of place, however, has its own set of problems with respect to standardization due to the very definition of 'sense of place,' that its complexity resists exact definition, and attempts to quantify it may miss the point.

Sense of place measurement

Measurement of sense of place and related constructs (place attachment) use both quantitative and qualitative methodologies, although qualitative or phenomenological approaches are common. Entrikin (1991) discusses the fundamental problems of accounting for human perceptions as variables in place analyses. The subjective meanings, feelings, and symbols which comprise sense of place are difficult to adequately quantify with standard positivistic measures such as Likert scales. Entrikin (1991) proposes the use of open-ended narrative as a method of assessing place perceptions and the use of conjoint analysis.

Still, a rich literature exists on quantitative means of assessing place-related constructs (primarily place attachment) dating back to the early 1990s—for instance, Williams *et al*. (1992) seminal work on the construction of a place attachment scale, and more recently Williams and Vaske's (2003) use of confirmatory factor analysis to examine the generalizability and validity of a two-dimensional scale of place attachment; and Jorgensen and Stedman's (2001) attitudinal scale representing three commonly accepted dimensions of sense of place and place attachment: place identity, attachment, and dependence.

Ecological beliefs and environmental values are potential modifiers of individual land-use choices and of an individuals' sense of place (Jorgensen & Stedman 2006). One way of quantifying these values is to map them spatially through place-based mapping (Brown 2005). Mapped landscape values will provide the link from social understanding to ecological analysis of specific places on the landscape. The mapped landscape values will be supplemented by a more nuanced understanding of ecological attributes that are being developed for the satisfaction domain of sense of place and the detailed information on ecological beliefs and values and behaviours. Through use of the ecological data, we can provide realistic scenarios for potential ecological change and directly link them to potential changes in landscape values and individuals' sense of place. Realistic 'what if' scenarios can be developed to understand trade-offs between sense of place, landscape values, and ecological change.

A number of efforts have been made towards mapping landscape values (Brown 2005). The most developed of these focuses on landscape values and activities and relates them to sense of place. Specific ecological values attributed to the landscape are rather limited in this literature and it is expected that the area needs to be expanded to facilitate integration with ecological data. By mapping landscape values there is the potential for suitability analysis, conservation planning, identification of local knowledge, and hot spot identification (Brown 2005). An additional benefit is the inclusion of landscape values in development scenarios to assess effects on social systems through the measurement of place attachment by validity and generalizability of a psychometric approach (Brown 2006). When combining mapping with a method for calculating potential conflict (Manfredo *et al*. 2003), the approach could provide a very useful guide for regional planning and conservation of natural areas and future development, as described by Yli-Pelkonen and Niemelä (2005).

5.5.2.2 Land cover

Humans transform a landscape for their habitat, a process known as urbanization. These transformations have been studied extensively to assess their effects on ecosystem patterns and processes—ecology in the system—and have been illustrated in this text, chapters in this book, and elsewhere (e.g. McDonnell *et al*. 2009). The use of that landscape by humans is called 'land-use' and is generally classified using a classification system devised by Anderson *et al*. (1976). How humans manage that land-use creates a variety of complex land covers such as yards, gardens, vacant lots, forest remnants, and agricultural plots (see Pauleit and Breuste, Chapter 1.1; Ellis *et al*. 2006). Although additional research is needed to link land cover to social structures and patterns, we hypothesize that these fine-scale attributes, both ecological and social, can be used to characterize a 'signature' of a landscape. These signatures would reflect specific ecological structure associated with specific social structure along the urban–rural gradient.

Fine-scale mapping to link social and ecological structure is not new. Biotoping has been used extensively for fine-scale mapping of habitats in urban landscapes, but is very labour intensive (see Pauleit and Breuste, Chapter 1.1; Sukopp & Weiler 1988). More recently, biotoping has been linked to social context to evaluate the effect of context on biodiversity (Cilliers and Sibert, Chapter 3.2; Cilliers *et al*. in review). Again, this approach is very labour intensive and may not be suitable for cross-site comparisons because of differences in social context. Cadenasso *et al*. (2007) have developed a fine-scale mapping protocol, HERCULES, that links infrastructure with vegetation cover, and have demonstrated its usefulness in predicting water quality as compared to an Anderson classification. Although Anderson land-use classification was not developed for predicting water quality (the use of per cent of impervious surfaces may be an easier method than the spatial mapping required by HERCULES), HERCULES begins to address the issue of infrastructure and its effect on ecosystem processes. Like biotoping, HERCULES is labour intensive and is rather unwieldy because of the numerous types of patches generated. Nonetheless, for small catchments, the protocol may be useful in linking ecological structure and function with social patterns and processes when supplemented with social characterizations. Quantifying error, a necessity for cross-site comparisons, may be problematic (see Ellis 2000). Similarly, Grove *et al*. (2006) used fine-scale analysis to identify what social attributes affect public land management, but their approach may not have worldwide applicability because of the use of PRIZM, a marketing classification system for the United States, and the costs associated with purchasing PRIZM data.

To couple human and natural systems, attention must be given to both land-use (human use) and land cover (biophysical condition) and their spatial and temporal dynamics and autocorrelation (Rindfuss *et al*. 2004). Obviously, multidisciplinary teams composed of natural, social, and spatial disciplines are necessary. Of these disciplines, social characterizations of a land parcel may be the most fluid. Rindfuss *et al*. (2004) observed that a land parcel may change ownership, be borrowed or rented, have multiusers with different purposes, and may have multiple jurisdictions affecting it. Yet, from a spatial perspective, the patch and its boundaries may not change with changing social context, and if change does occur, there may be a time lag before it is recorded by the observer. Rindfuss *et al*. (2004) further report four issues that occur with linking natural, social, and spatial attributes over time: 1) aggregation and inference problems, 2) land-use pixel links, 3) data and measurements, and 4) remote sensing analysis. Aggregation and inference problems are scale issues. Patterns observed at the aggregate level of county or district may not exist at the household level (see Robinson 1950) and similarly patterns at the household level may not be the same operating at an aggregate level. Rindfuss *et al*. (2004) state that the solution is rather simple '. . . the level of aggregation in the measurement needs to match the level of aggregation in the hypothesis being examined'. In other words, link your methodology to your research question.

The challenge of linking land-use to pixels has three issues: 1) the fundamental differences between the way data are collected on people and pixel, 2) the spatio-temporal implications of the data collections, and 3) analytical problems associated with combining issues 1 and 2 (Rindfuss *et al*. 2004). As previously mentioned, the parcel and its boundaries may not change spatially over time but ownership and use may. Obviously, longitudinal studies are needed to ascertain how sociological attributes and their organization change within a parcel. The challenge of linking land-use to pixels echoes also in the type of data and measurements taken. Intuitively, this issue is driven by the research question. What sociological measurements are needed and how are they expressed on the ground? Ground truthing for both sociological attributes (e.g. through surveys and interviews) and biophysical features of the parcel, needs to be done congruently. An important aspect of this component is data quality, not only with respect to the individual disciplines but also interactions between disciplines (Rindfuss *et al*. 2004). Unfortunately, such a data error structure has not been developed.

Finally, remote sensing analysis issues are of particular concern for temporal analyses when using multiple scenes from various time points. 'False change' errors may occur with mis-registration of maps and textural differences (Ellis *et al*. 2006). These problems can be corrected through estimating changes in the ecological map across the sample cell (Wang & Ellis 2005). Another error is disagreement between interpreters, which can be minimized through training and testing quality assurance at different intervals of study. This testing will yield a measure of interpreter error which can be used to give a conservative estimate of prediction error for the reported spatial changes.

Even with the issues presented, fine-scale analyses (e.g. land cover) may be the best opportunities to link social and ecological data and enable cross-site comparability. For example, Ellis *et al*. (2006) used land-use and land-cover data, interpreted from high spatial resolution (<1 m) imagery, for six 1 km^2 sites, two in the United States (urban) and four in China (agricultural villages), to compare long-term ecological changes within densely populated landscapes. Using their protocol, one could modify land-uses and land covers for urban landscapes only. By appropriately overlaying sociological data, one can be begin to evaluate fine-scale changes sociologically and ecologically. For instance, a wealth of sociological data existed for New Orleans neighbourhoods before Hurricane Katrina impacted the city, and a considerable amount of data has been collected after the storm. By coupling that data with high resolution imagery and on the ground sampling, one can begin to examine the resiliency of social and ecological structure in response to a trauma. Similarly, many cities in Europe are losing their population because of low birth rates and emigration. This dynamic change provides the opportunity to study stability of social and ecological systems in a changing economic environment.

5.5.3 Modelling social-ecological systems

In 1998, the US National Science Foundation established two long-term ecological research sites—Baltimore, Maryland, and Phoenix, Arizona—to study urban landscapes. One of the desired outcomes from these programmes was to link social-economic systems with ecological systems (Redman *et al*. 2004). For ecological systems, the LTER network in the United States focuses on five core research areas: pattern and control of primary production, spatial and temporal distribution of populations selected to represent trophic structure, pattern and control of organic matter, pattern and movement of inorganic material, and patterns and frequency of disturbances (http://www.lternet.edu/coreareas). By having similar core research areas, LTER sites can conduct cross-site comparative studies of ecosystem structure and function across a vast array of bioregions. With the advent of the urban LTERs, similar core research areas have been proposed for sociological patterns and processes within the LTER network. Core areas are demography, technological change, economic growth, political and social institutions, culture, and knowledge of information exchange (Redman *et al*. 2004) (Table 5.5.1). Redman *et al*. (2004) propose that the human and ecological components of urban landscapes interact across multiple spatial and temporal scales and these interactions are mediated by land-use, land cover, production, consumption, and disposal.

An initial approach to link social and ecological systems was to adapt the human ecological model proposed by Machlis *et al*. (1997) to urban landscapes (Pickett *et al*. 1997). In their model, Machlis *et al*. (1997) provide a detailed list of socio-economic attributes that define social systems and link those attributes to natural resources. The model, itself, does not identify how each attribute interacts, but

Table 5.5.1 Redman *et al.* (2004) proposed the following social patterns and processes to serve as core research areas for social analyses within the National Science Foundation Long-Term Ecological Research Network

Demography: the growth, size, composition, distribution, and movement of human populations.
Technology change: the accumulated store of cultural knowledge about how to adapt to, use, and act on the biophysical environment and its material resources to satisfy human needs and wants.
Economic growth: the sets of institutional arrangements through which goods and services are produced and distributed.
Political and social institutions: enduring sets of ideas about how to accomplish goals recognized as important in a society. For instance, most societies have some form of family, religious, economic, educational, health, and political institutions that characterize its way of life.
Culture: culturally determined attitudes, beliefs, and values that purport to characterize aspects of collective reality, sentiments, and preferences of various groups at different scales, times, and places.
Knowledge and information exchange: the genetic and cultural communication of instructions, data, ideas, and so on.

rather leaves the identification of linkages to the user. For example, Pickett *et al*. (1997) modified Machlis original model by enhancing the biophysical resources to represent ecosystem patterns and processes, thus creating the framework to evaluate the effect of social structure on ecosystem structure and function (Fig. 5.5.1).

Another modelling approach has been to link social drivers with ecological systems. Alberti (2008) identifies important socio-ecological drivers and recognizes the importance of spatio-temporal elements (Fig 5.5.2). In her conceptual model, ecosystem patterns and processes are linked directly to social patterns and processes through relevant interactions and feedbacks. Other models also have been proposed. For instance, Grimm *et al*. (2000) developed a conceptual scheme that integrates ecological and social systems (Fig. 5.5.3). What is particularly interesting about Grimm *et al*.'s (2000) approach is that the feedback loop within the social system is directly linked to land-use and ecosystem patterns and processes. In other words, they propose that in order to study social systems within an urban landscape, knowledge and information about the ecological and social systems is a prerequisite, thus directly linking social patterns and processes to ecological patterns and processes. Yli-Pelkonen and Niemelä (2005) have applied Grimm *et al*.'s (2000) model to guide urban planning in making recommendations for biological conservation of natural areas within an urban landscape. A more detailed description on how each of these models differ is described by Alberti (2008).

These models tend to be structural in their orientation, focusing on system level patterns and processes such as institutions, demographics, policies, and land-use patterns, and are known as system models. They often require detailed enumeration of causes and functional representations, and assume a linearity across a spatial scale (Parker *et al*. 2003). In general, these models are primarily biocentric—focusing principally on how ecological components are affected, this is to say the ecology *in* urban landscapes. In general, they do not account directly for the effect of the decision-making process or social drives on social patterns and processes. The research focuses principally on how urbanization, through different socio-economic drivers, affects ecosystem patterns and processes. That is not to say that socio-economic components are not identified and recognized as being affected too (see Grimm *et al*. 2000), but rather that the feedback mechanisms of how ecological systems are changing socio-economic systems are generally not quantified. For instance, in our water quality example, we can identify and quantify those anthropogenic factors creating the condition leading to degraded water quality, but we have not identified changes in social responses to this condition. This is understandable considering ecologists and biologists are using the models to study ecological patterns and processes under different land management conditions and social contexts—the ecology *in* urban landscapes.

Another set of models defining socio-ecological systems are called agent-based models. This set of models focuses on individuals or agents and their decision-making processes and actions (Parker *et al*. 2003). Agents are perceived to have control over

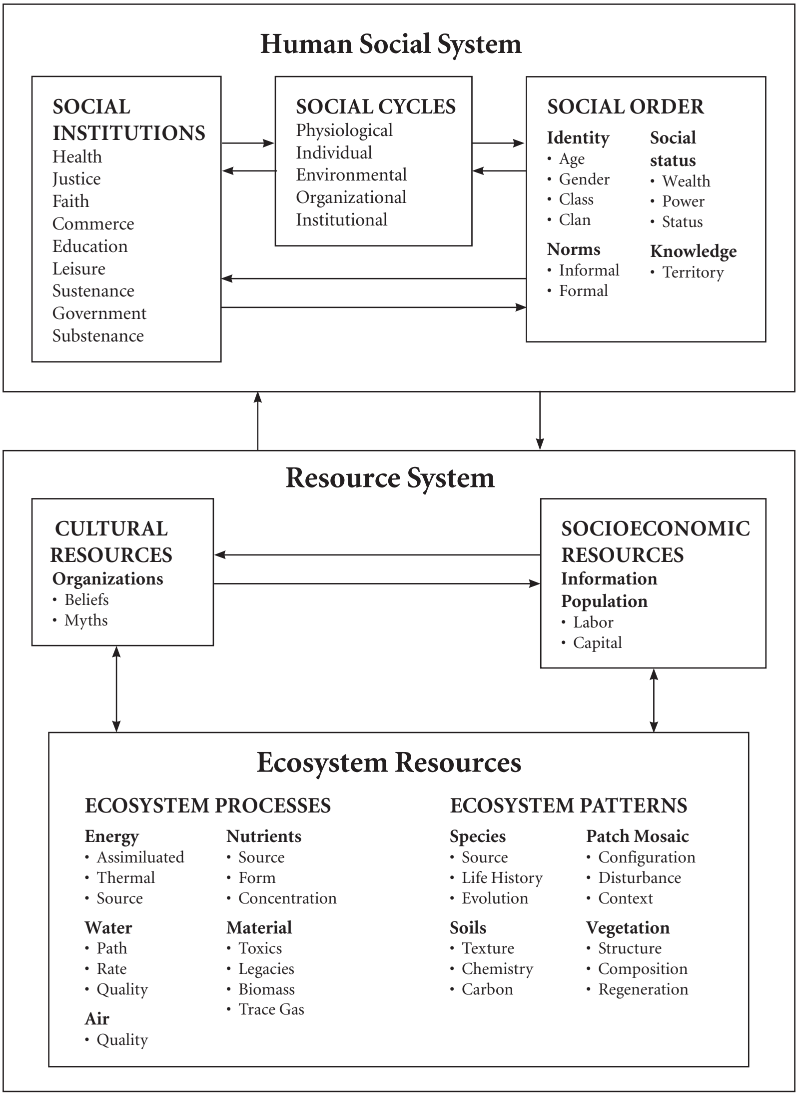

Figure 5.5.1 A conceptual model illustrating components within a human ecosystem and general linkages between those components, as proposed by Pickett *et al*. (1997). With kind permission from springer Science+Business Media

their actions and behave according to some model of decision-making, often some form of rational choice theory from economics. These models can be used to explore emergent patterns from individual human interactions between themselves and with the environment (Alberti 2008).

A third approach has begun to integrate system- and agent-based models, thus bringing together autonomous human agents and structural drivers, creating multiagent system models (Parker *et al*. 2003). Institutions, policies, and other social systems and ecosystem resources are seen to both enable and constrain the agent's actions, on the one hand, while also being the products of previous human actions and intentions (Morse 2007). For instance, Morse (2007) used an agent-based model to identify how policy was used to influence individual decisions on land-use and land cover and how the actions they

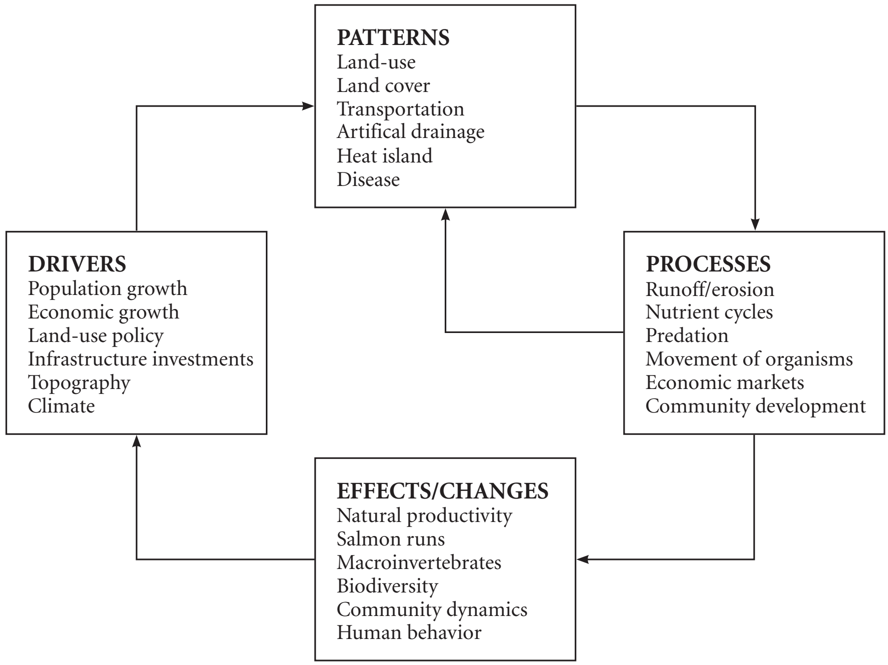

Figure 5.5.2 A conceptual model representing the relevant interactions and feedback of a socio-ecological system for urban ecosystems, as proposed by Alberti (2008). With permissin from The University of California Press

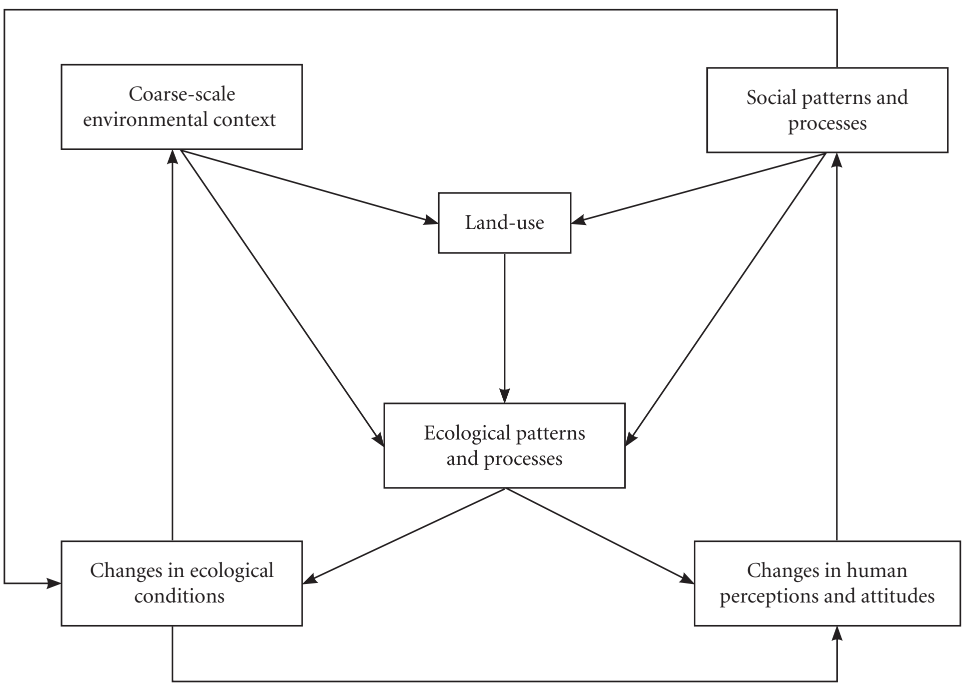

Figure 5.5.3 A conceptual scheme for integrating social and ecological systems in urban landscapes, as proposed by Grimm *et al.* (2000). With permissin from The University of California Press

took affected landscape patterns in an agrarian landscape in Costa Rica. A critical feature of this kind of model is to link the decision-making process across multiple levels in a dynamic manner that allows for understanding of decision-making within a changing structural (social and ecological) context. Even more importantly, this type of agent–structure–agent model allows for understanding of feedback that links changing ecological conditions to decision making. How do actors learn from past decisions? What information is taken into account when making decisions? Is ecological information taken into account? If it is, where did this information come from? How accurate is the information? How do we change the structures that influence future decisions? In essence, how do we respond to environmental feedbacks?

5.5.3.1 Complex adaptive systems

The processes of structural change to both social and ecological systems are examined by focusing on the cyclical process of human actions and interactions with the environment. The approach is based on the premise that urban landscapes are complex adaptive systems (see Gunderson & Holling 2002). In complex adaptive systems (CAS), the interactions of lower level components result in emergent patterns at higher levels that, in turn, feedback to influence future lower level interactions (Levin 1998). It is through this cyclic process of interactions and feedback that CAS self-organizes, often into nested systems (Levin 1998). CAS are also characterized by high levels of uncertainty, cross-scale interactions, threshold effects, and the possibility of multiple equilibria (Gunderson & Holling 2002). With a foundation in CAS, it is clear that understanding the feedback between and within social and ecological systems is critical if we are to manage for the resilience of these systems and their sustainability. Adaptive environmental assessment and management was developed as a flexible approach to address uncertainty and to provide a framework for active learning (Gunderson & Holling 2002). One of the central tenets of adaptive management is learning, and the learning is dependent upon processing of information in a formalized manner through experimentation, monitoring, and assessment (Gunderson 1999). But is all the information available? With the inevitability of surprise in CAS, is it even achievable? Is the information applicable or useful to human actors managing the system? How do different actors or decision-makers translate the information into actions? Before we continue the discussion on CAS, we need to expand our understanding of CAS by linking social and ecological systems in a Structuration of Complex Adaptive Systems (SoCAS) framework (see Morse 2007).

Structuration of Complex Adaptive Systems

Central to a SoCAS framework are elements of structuration theory from the social sciences (Stones 2005), those of complex adaptive systems outlined above (Levin 1998; Gunderson & Holling 2002), and the theory of hierarchical patch dynamics (Wu and Loucks 1995) from the ecological sciences (Morse 2007). Both social and ecological theories are used to provide guidance because the drivers of these systems operate differently across spatial and temporal scales. Human actors act with foresight, reflexivity, and can communicate those ideas into the future, while ecosystems do not (Westley *et al.* 2002). For the framework, the social and ecological CAS mirror each other and are linked at the point where human actions and interactions with the environment occur (Fig. 5.5.4).

Structuration theory frames 'the interaction of structure and agency across scales [that] must be the centerpiece of a dynamic understanding of people–environment interaction' (Scoones 1999). Structuration theory avoids both an overly objective structural approach and an exaggerated emphasis of subjectivist, agent-based approaches by focusing on their interaction (Stones 2005). Human action is viewed as a continuous flow of conduct (Giddens 1984). Structure is seen as both 'the medium and outcome of the conduct it recursively organizes' (Giddens 1984). Structure enters into the constitution of the agent as a medium (internal structure) and from there into the practices that the agent produces as an outcome (external structure) (Stones 2005). Structures that are the outcome of one period of conduct (actions, activities) become the medium for the next round of agents' conduct (Stones 2005). Through recursive social conduct,

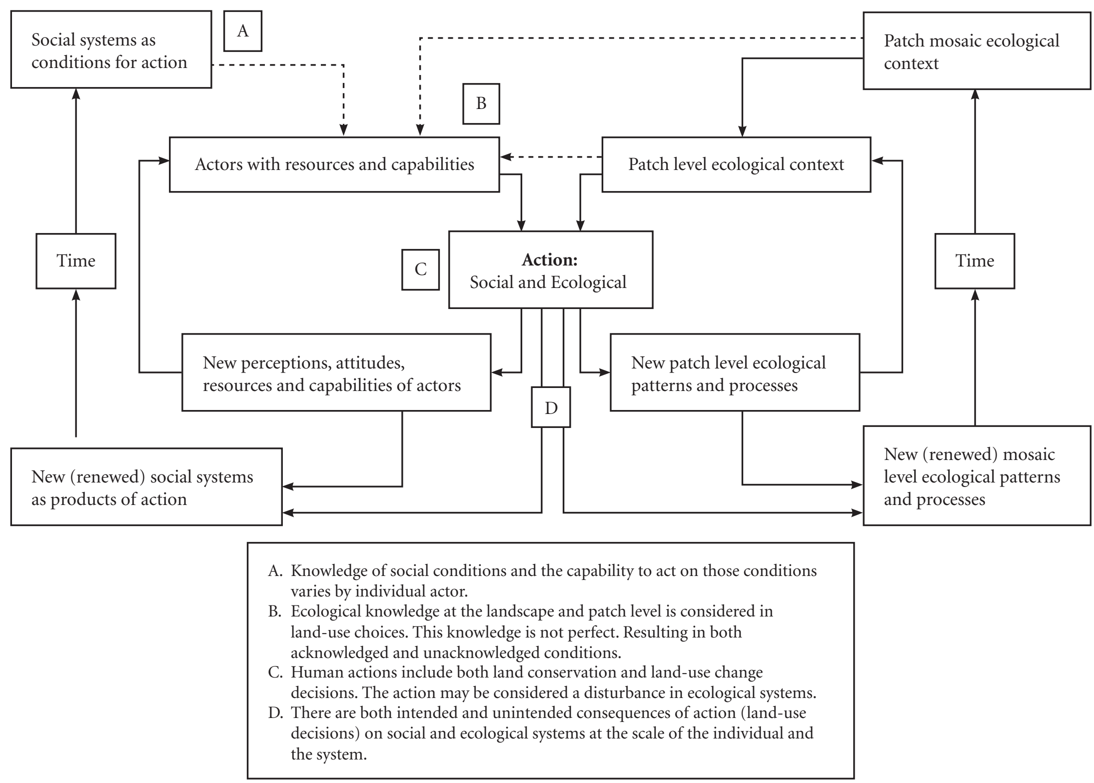

Figure 5.5.4 A conceptual model linking social decision-making processes with landscape dynamics to characterize the effect of land-use on social and ecological systems. This model is nested hierarchically within larger systems. With permission from The University of California Press

structures influence the activity of individuals, who in turn, transform or reaffirm those same structures constantly producing and reproducing society (Kondrat 2002).

Social systems (e.g. markets, governments) can be thought of as the patterns of social relations, or regularized social practices, that stretch across time and space produced by the process of structuration (Giddens 1984). They are the complex, entrenched, and powerful networks of relationships, behaviors, beliefs, interactions, rules, and resources, and are both temporally and spatially contingent (Kondrat 2002). Furthermore, they are integrated with other social systems hierarchically, across space and over time. Structures are considered to be both enabling and constraining of agents actions (Giddens 1984).

The model represents several areas where information is an input for the decision process (Fig. 5.5.4). Some of this information is ecological (B) and is not always perfect (dashed lines). Other information reflects the extent to which the decision-maker knows about external social structures and how to 'make things happen' within the system (A). The decision-maker takes action based on this knowledge (A, B) and incorporates their capabilities (resources at their disposal) to take action.

The black box of actor decision-making in Structuration includes, 1) their motivation to action, 2) the rationalization of action and knowledge, and 3) their ability to reflexively monitor their action (Giddens 1984). Motivations are the wants and desires of an individual and are their overall plans for conduct. Rationalization of action includes the agent's knowledge and how to proceed to obtain their intended outcomes. This knowledge is not perfect, however, and there is always the possibility of unacknowledged conditions and unintended consequences. Reflexive monitoring of action includes the agents' continual monitoring of actions and the consequences of their actions, and of the actions of others. This monitoring can be for both social and environmental consequences. This understanding of the individual's decision-making process matches nicely with that of adaptive management. The focus on knowledge provides a framework to emphasize conservation learning, the use of information, and the development of an ecological aesthetic. 'As knowledge about both social and ecological systems is often imperfect and the cumulative and cascading consequences of our actions within and across both systems are extremely complex and difficult to predict, explicit identification of these knowledge systems (or lack thereof) is a critical feature for a framework of linked social ecological CAS' (Morse 2007). Our model, and the others using this integrated approach, are well-adapted to capturing the interdependencies, heterogeneity, and nested hierarchies among agents and their environments characterizing urban landscapes (Parker *et al*. 2003). Parker *et al*. (2003) provides an excellent review of the application of multiagent system models to simulate land-use change.

5.5.4 Summary

We began this chapter by presenting a scenario involving a highly contaminated stream and the lack of political or environmental response to it. We ended the chapter by discussing ways in which to model social and ecological data. During this course, we presented different techniques and models currently being used to evaluate the integration of social and ecological systems. Throughout the discussion, the underlying assumption is that the ecology *of* the system can be derived for urban settlement along an urban to rural gradient only through long-term monitoring of both social and ecological components across multiple scales. Through the long-term monitoring of social and ecological systems, we will be able to build into these models the nuances of the complexity of our urban systems, thus potentially linking changes to social patterns and process to ecological patterns and processes. In doing so, we begin to take the necessary steps to identify factors affecting the sustainability of urban landscapes.

CHAPTER 5.6

Building Urban Biodiversity through Financial Incentives, Regulation, and Targets

John Box

5.6.1 Introduction

Maintaining, enhancing, and creating biodiversity is a necessity for humans living and working in urban areas. People derive considerable benefits from contact with nature in terms of physical and mental health and well-being (Rohde & Kendle 1997; Douglas 2008; Maller *et al*. 2008; Tzoulas and Greening, Chapter 5.2). However, the benefits are hard to quantify in terms of community health and are usually perceived as being part of a general 'quality of life' factor by individuals living and working in urban areas. Nevertheless, hard evidence of the benefits of contact with nature for personal health is becoming available (Fuller *et al*. 2007; Pretty *et al*. 2007; Mitchell & Popham 2008). Furthermore, there are real economic benefits, albeit usually indirect, from the services provided by ecosystems and habitats—for example, flood regulation, noise reduction, and air quality improvements (Costanza *et al*. 1997; Millenium Ecosystem Assessment 2005; Defra 2007b; Alfsen *et al*., Chapter 4.3., Bridgewater, Chapter 4.4).

Such benefits from biodiversity are relevant at a community level but do not easily translate into direct financial benefits to individual urban residents or businesses. If they did, there would undoubtedly be a powerful incentive for actions to increase biodiversity in urban areas. Environmental taxation (such as taxes on carbon emissions or on waste to going to landfill) and financial savings (through energy efficiency or reduced water consumption) can offer financial incentives for urban residents and businesses in relation to environmental improvements. But neither of these provide an obvious model for retaining biodiversity in our towns and cities, let alone enhancing or even creating ecosystems and habitats.

Changes in the biodiversity of urban areas can only occur through changes in human behaviour whether as individuals, organizations, or businesses. Drivers for change include 1) economic benefits directly affecting businesses and people; 2) legislation, regulation and official guidance; and 3) targets to encourage individuals and organizations to pioneer change and overcome cultural norms by providing evidence of tangible achievements. The application of such a package of measures to retaining biodiversity in urban areas, let alone increasing biodiversity in our towns and cities, will require a real change in human behaviour that must be driven by economics and financial incentives in combination with legislation, regulation, and targets. Voluntary changes in the behaviour of individuals or organizations will also occur—there will always be pioneers who will make a reality of new ideas. But where there are likely to be economic costs which cannot be set against quantified financial benefits, substantial and lasting changes will not be achieved by voluntary changes in behaviour alone. Individuals and organizations are not generally altruistic and require an environment where all are subject to a comparable financial and legal framework so that an extra economic cost does not result in a financial disadvantage.

5.6.2 Economic drivers to increase urban biodiversity

Economic systems can derive good measures of manufactured capital (machines, buildings) and human capital that are used in sustaining human populations. Biodiversity, natural capital, and the associated ecosystem services (such as climate regulation, water regulation, soil formation, nutrient cycling, and waste treatment) are not fully accounted for in economic systems and commercial markets. Natural capital in this context is the ecosystems themselves together with the atmosphere, water, and minerals required to sustain them.

Ecosystem services can range from simple services such as crop pollination, to more complex services such as soil formation, sinks for waste, and climate regulation (see McDonald and Marcotullio, Chapter 4.1). Humans depend on these services for food, clean water, and clean air. Land-use changes often result in a simplification of the ecosystem, for example to create agriculture, thus reducing the ability of the ecosystem to provide a range of services. Urban areas contain very greatly modified ecosystems and curtailed ecosystem services.

The economic benefits from the environmental services provided by ecosystems and habitats are very significant (Costanza *et al*. 1997; Balmford 2002; Millenium Ecosystem Assessment 2005a). The overview of the global value of ecosystem services provided by Costanza *et al*. (1997) estimated that their average value was US$33 trillion ($10^{12}$) per year (1994 prices) within a range of US$16–54 trillion, which compared to global gross national product of US$18 trillion per year. This was acknowledged as being a first approximation of the relative magnitude of global ecosystem services. A detailed account of characterizing, measuring, and valuing ecosystem services is given by Daily (1997).

Commercial markets do not recognize biodiversity, natural capital, and ecosystem services because they cannot be easily accounted for. The benefits to society of the ecosystem services are not reflected in the prices set by the markets for land and land-use changes. Therefore, the value of this natural capital and the ecosystem services it supports is inevitably downgraded in decisions about policy, plans, and projects. But markets and economic models can price in risk and do respond to scarcity in resources or to 'natural sinks' starting to fill up with pollution. Ecosystem services contribute to human welfare through contributing to the generation of wealth in the broadest sense and through preventing damage that inflicts costs on society. Clean air and water, soil formation, climate regulation, waste treatment, good health, and aesthetic benefits are all dependent on these ecosystem services. Crucially, the contribution of natural capital and ecosystem services to both wealth generation and damage prevention needs to be taken into account in policies, plans, and projects.

Because biodiversity, natural capital, and ecosystem services have not been assigned economic values, they are outside the market and are ignored and treated as externalities, which means their costs are born by society whereas their benefits are accrued privately. Market failures occur through prices not giving the real cost to society of land-use changes or producing goods and services. Markets with failures lead to inefficiency and waste which can be corrected by price changes, taxes, or regulation.

Assigning economic values to ecosystem services is a new concept for both environmentalists and economists alike. Awareness of the environment as one of the three fundamental parts of sustainable development (the other two being economic and social) has required proper engagement with environmental issues. There is an argument that we cannot, or should not, put a value on human lives and biodiversity and ecosystems. But society does place value on human lives through mechanisms such as compensation payments for death and injury; through paying for health and safety measures and construction standards (e.g. for buildings, roads, transport systems); and through cost/benefit models, for example for new roads where notional costs are placed on reducing accidents and saving individual lives.

The effective way forward can only be to assign real economic value in decision-making, whether at the policy, plan, or project stage, to the biodiversity and natural capital that produces these ecosystem services. In practice, it is the project stage where the real economic costs of a project are crystallized as part of the pricing, cost-saving, and value-engineering processes. Notional costs for loss of ecosystem services must be able to be assigned to an individual project during project appraisal—and indeed the costed benefits of any ecosystem enhancements that may be derived from a project. Such an approach requires appropriate data to be added into the cost/benefit models.

Balmford *et al*. (2002) took the work by Costanza *et al*. (1997) further by looking at the economic benefits provided by natural ecosystems and by commercially exploited versions (using case-studies of logging, aquaculture, drainage for agriculture, and blast fishing on reefs). They found that the loss of non-marketed services accruing to society outweighed the marketed marginal benefits of conversion. Put simply, commercial exploitation yields private benefits because social benefits from unexploited ecosystems are not costed. Balmford *et al*. (2002) concluded that the development of market instruments would enable the social and global values of natural ecosystems to be captured privately through mechanisms such as carbon or biodiversity credits, or through premium pricing for ecosystems goods such as fish or timber. Whilst these authors were looking at natural ecosystems in a global context and trying to find market mechanisms to conserve natural ecosystems, their fundamental argument is relevant to urban areas—how are community benefits factored into private cost/benefit models?

The UK Government has determined that the value of services provided by the natural environment needs to be reflected in decision-making. It has established a strategic agreement for the delivery of public services known as a Public Service Agreement (PSA) across the whole of the UK Government (PSA 28: *Secure a Healthy Natural Environment for Today and the Future*) (UK Government 2007). The five key indicators for measuring progress on PSA 28 are water quality, biodiversity (wild breeding bird populations), air quality, marine health, and agricultural land management which are supported by a broader set of indicators. Evidence that such work on complex issues is being taken seriously is set out by the UK Department for Environment, Food and Rural Affairs in the action plan to deliver PSA 28 (Defra 2007a) and in the clearly expressed arguments in the accompanying guide to valuing ecosystem services (Defra 2007b). The environmental cost of incremental development of large natural resources is well illustrated in Defra (2007b) by examples based on the residential development of urban parks.

An excellent example of the results of costing the environmental goods and services provided by open spaces in an urban area has been set out in the pioneering open space and environmental services plans for Durban, in South Africa, which is managed and administered by the eThekwini Municipality (Durban Metropolitan Open Space System (1999); eThekwini Municipality (2001, 2003)). The open space system covers some 63,000 ha and the estimated value of the environmental goods and services supplied by this open space system in 2003 was R3.1 billion (US$300 million), which could be compared to the operating budget for the Municipality in 2001/02 of R6.5 billion) and the capital budget of R2.8 billion. These plans are being translated into land-use policies and a climate change strategy. The process of establishing the open space system and refining it, together with the costing exercise which was undertaken as a contribution to the global exercise of valuing ecosystem services (Costanza *et al*. 1997), is a case study which merits very careful examination (an overview is given by eThekwini Municipality & Local Action for Biodiversity (2007)).

Another and different example is provided by the economic value that the city of Philadelphia (USA) derives from its 10,000 acres (4,000 ha) of park and recreation system including woodlands, rivers and streams, trails, golf courses, picnic areas, and playgrounds (The Trust for Public Land & Philadelphia Parks Alliance 2008). Seven major aspects were included in the valuation – clean air, clean water, tourism, direct use, health, property value, and community cohesion (through being involved in neighbourhood parks). The direct income received annually by the city in tax receipts derived from increased property values and tourism was estimated at US$23.3 million. Cost savings of US$16.1 million were due to the open spaces providing stormwater management and air pollution mitigation, together with reduced anti-social problems through improved community cohesion. In addition, there are estimated savings to the citizens of Philadelphia of US$1.1 billion annually from free use of the parks and recreation system, combined with savings in medical costs. Finally, the collective wealth of the citizens was estimated to increase by US$729.1 million annually due to increased property values from proximity to parks and profits from tourism. These economic values need to be translated into the planning policies that Philadelphia will require to maintain these ecosystem services.

Much can be achieved for biodiversity and people through the promotion of multi-functional urban greenspace where multiple land-uses are recognized (e.g. for the UK, see Barker 1997; CABE 2004; see also Fig. 5.6.1).

The value of such urban greenspaces can therefore be costed in terms of ecosystem services (e.g. flood regulation, air quality amelioration), thus increasing the notional land value of a given urban greenspace. Planning of urban areas with multi-functional urban greenspace or green infrastructure (e.g. TCPA 2004) is fundamental to making cities work to the benefit of those living in them.

Fiscal incentives are required for the inclusion and, crucially, the maintenance of measures such as accessible natural greenspace, biodiversity-friendly sustainable urban drainage systems (SUDS) (e.g. Woods-Ballard *et al*. 2007), green roofs (e.g. Dunnett & Kingsbury 2004), and new habitats—both in new housing and development projects and retro-fitted into existing developments. The review of housing supply undertaken for the UK Treasury (Barker 2004) included a proposal for a planning-gain supplement imposed on development gains accruing to a landowner who receives planning permission. Such a tax would extract some of the windfall gains and recycle them back to local communities—a concept which is consistent with the potential for transference between economic, social, and environmental assets required by sustainable development. Natural England (the statutory nature conservation agency in England) and the Royal Society for the Protection of Birds (RSPB) assessed ways in which such a tax might benefit nature conservation, including discounted tax rates for developments incorporating biodiversity measures (English Nature & RSPB 2006).

5.6.3 Legislation, regulation, and targets to increase biodiversity

Legislation and regulation is usually a last resort when change is required by government but, in relation to biodiversity, official guidance and standards have been found to be inadequate, with individuals

Figure 5.6.1 An example of multifunctional urban greenspace—a surface water balancing lake used by anglers and overlooked by houses whose residents can appreciate wildlife both on the water and in the surrounding wetlands and woodlands. Copyright: John Box (author)

and businesses insufficiently motivated to undertake the changes required to achieve the desired outcome. Whereas legislation legitimizes the changes for all those affected by a new law or set of regulations, guidance will only work if everyone does what is required.

Biodiversity legislation is generally introduced to give protection to rare, threatened, or notable habitats and species—or those species with economic benefits (e.g. fish and fisheries, deer, wildfowl, and game birds). The breadth of biodiversity legislation in terms of the habitats—and particularly the species involved—is increasing in the UK, as is its depth in relation to the severity of the penalties. This is also the situation in a global context as countries industrialize, and competition for land and resources results in the inevitable decline in ecosystems and species to the point where their rarity requires protection through legislation as a result of public pressure. Wildlife legislation in the UK focuses on the protection of rare habitats and species. This legislation and the planning laws in the UK do not yet set limits for permitted changes in the local or even national populations of particular species, or in the extent of certain habitat types in relation to new developments—due either to the initial land-take or to subsequent disturbance from the operation of the permitted development. The use of the word 'yet' is deliberate, because the introduction of such planning legislation will undoubtedly come as a justified response to increasing scarcity in notable species and habitats.

Sustainability has been proposed as a more appropriate mechanism to achieve net gains in biodiversity for individual projects rather than further losses. It is true that any project—industrial, residential, commercial, agricultural, mineral extraction, fisheries—can be assessed in terms of biodiversity and sustainability. The real tests are set by transferring admirable but theoretical concepts, such as sustainability, from the drawing board to the economic realities of the commercial boardroom or the practical realities of the construction site. If there are neither significant economic benefits, nor legislative constraints, admirable concepts rarely survive initial cost/benefit appraisal, profit/loss financial scrutiny, or construction deadlines.

The regulatory framework for urban development needs to move away from *mitigating* biodiversity losses. Instead, it should demand demonstrable biodiversity *gains* over and above requirements for mitigation or compensation, that are formally agreed by an informed regulator whose standards are based on real evidence and good science. Such an approach is consistent with the need to maintain ecosystem services that are dependent on a stock of natural capital.

Loss of ecosystem services needs to be quantified, costed, and regulated. A comparison with climate change and atmospheric CO_2 levels is instructive. The need for global action has been agreed through the Kyoto Protocol and in the UK the economic case has been comprehensively made by the (then) Head of the UK Government Economic Service (Stern 2007). The UK now has the Climate Change Act 2008 which introduces legally-binding commitments for cutting CO_2 emissions, the 2007 Planning Policy Statement on climate change for England, and the radical overhaul in 2006 of Part L of the Building Regulations for England & Wales, which deals with energy conservation and which now requires all new buildings to be modelled to assess their likely CO_2 emissions per unit area (it is envisaged that future revisions of Part L will seek to achieve zero carbon emissions by 2016). The additional costs of such legislation and regulatory mechanisms is accepted as necessary by the business community because the new carbon regime is added onto existing mechanisms, such as building regulations. The quantification of carbon outputs to the atmosphere and its implications provide a sound justification for such new costs to businesses, which can respond through initiatives such as carbon calculators for planning and designing projects (Atkins 2008). The effects of losses and gains in relation to biodiversity, natural capital, and ecosystem services now requires just such a technical and quantified approach if it is to win the hearts and minds of both individuals and businesses.

Voluntary approaches using standards and targets are usually introduced before a government resorts to legislation and regulation. Standards and targets for urban greenspace are essential for the initial stages of planning large-scale development at a regional or subregional scale in order to incorporate green networks and green infrastructure (TCPA 2004), for example, the Green Network of

Telford in the West Midlands of England (Box, Cossons, & McKelvey 2001), the East London Green Grid Framework (Greater London Authority 2008), and the Green Space Plan 2000 for Tokyo (CABE 2004, p. 16). The concept of green networks and green infrastructure needs to be updated to accommodate the ecosystem infrastructure required to support ecosystem services and natural capital.

Urban greenspace provision is usually seen in terms of quantitative standards (unit area of greenspace per resident or household), or accessibility standards (defined areas of greenspace within defined distances from every resident). An example of a quantitative standard is the 31 largest towns and cities in the Netherlands that have agreed on a guideline of 75 m^2 green space per dwelling (van Egmond & Vonk 2007). An example of an accessibility standard is Aarhus, the second largest city in Denmark, where there are standards defined in the Green Structure Plan that no dwelling should be more than 500 m from a green area of at least 6,000 m^2 (reported in CABE 2004, p. 25).

Standards for accessible natural greenspaces—such as no person living more than 300 m from a natural greenspace of at least two hectares in size—have been adopted by the statutory nature conservation agency for England (English Nature 1996; Natural England 2010). Technical and institutional barriers for the implementation of such an urban greenspace model have been identified and a toolkit produced for local authorities, who are envisaged as being the key agencies for applying the targets at a local level through local planning policies and local development frameworks (Handley *et al*. 2003). A broadly similar process is being undertaken in Wales where comparable accessible natural greenspace standards have been established by the Countryside Council for Wales (Countryside Council for Wales 2006) and are being promoted by the Welsh Assembly Government through the environment strategy for Wales (Welsh Assembly Government 2006, p. 42 & 43) and through planning advice for open spaces (Welsh Assembly Government 2009, p. 9).

Some may argue that there is no room for more urban greenspace in crowded urban areas. But why not create these areas? One very public example of this attitude is shown in Fig. 5.6.2, which is the 'green wall' at the CaixaForum culture forum on the Paseo del Prado in Madrid.

Figure 5.6.2 The 'green wall' at the CaixaForum culture forum on the Paseo del Prado in Madrid, which covers some 460 m^2 and supports 15,000 plants from 250 different species. Copyright: John Box (author)

The nature conservation strategy for the metropolitan county of the West Midlands, which included Wolverhampton and the Black Country, Birmingham, and Coventry, pioneered the aim that all residents should have reasonable access to wildlife habitats (West Midlands County Council 1984). The strategy identified 'urban deserts' based on areas where residents were more than 1 km away from accessible wildlife habitats, and a policy of habitat creation was proposed for these Wildlife Action Areas. Such a methodology has been subsequently used in other nature conservation strategies for major urban areas, for example London (Greater London Authority 2002) and Birmingham (Birmingham City Council 1997).

The challenge is for local authorities and public bodies to turn areas that they own, such as mown amenity grassland, into more interesting and stimulating natural greenspace and to ensure that accessible natural greenspace is incorporated into new developments through planning policies and through working with those involved in the new developments. Creative management of biodiversity at the local level is demonstrated on an international scale by the ICLEI (Local Governments for Sustainability) initiative known as Local Action for Biodiversity, which is supported by the UNEP Urban Environment Unit and IUCN.

High urban land values mean that a significant commitment by the landowner is required for land to be designated such that the primary function is nature conservation (in such cases, the landowner is usually a public body, such as a local authority or local council). However, recognized legal mechanisms are available in the UK to link the granting of planning permissions to the provision by the developer of monies for the provision of new roads, new surface water sewers, new schools, and new open spaces. Such provisions should be extended to the creation of new wildlife habitats and ecosystems, and also to the funding of such habitat creation programmes, from the full range of built developments that require planning consent.

Targets and standards can drive such a process along if there is an overall strategy to increase open spaces. For example, some 16,000 ha of public greenspace are proposed to be created by 2013 in the Randstad, the major urban area in the west of the Netherlands that includes Amsterdam, Rotterdam, The Hague, and Utrecht (van Egmond & Vonk 2007). Tokyo has significantly less green open space per person (6.1 m^2/person) than London (26.9 m^2/person) and the Green Space Plan 2000 for Tokyo aims to develop 400 ha of green space by 2015 (CABE 2004, p. 16). Paris has a goal that all citizens can live within 500 m of a green space, which has resulted in a programme to create new green spaces within identified areas of deficiency, including buying derelict houses to create small green spaces (CABE 2004, p. 82–83).

Climate change creates additional factors to be taken into consideration and will require special solutions. The Stern review on the economics of climate change recognized the need for flexible policies whose aim is to reduce fragmentation and encourage movement and migration of species by making use of wildlife corridors (Stern 2007, p. 481). The construction of 'green bridges' across roads and railway lines at key locations would provide a means of reducing habitat fragmentation and would make it easier for species to move in response to climate change. Moreover, Article 10 of the European Community Habitats & Species Directive (Council Directive 92/43/EEC) encourages the management of features of major importance for wildlife, such as those which have a linear and continuous structure or a function as stepping stones and are essential for the migration, dispersal, and genetic exchange of wild species.

5.6.4 Conclusions

Urban design, planning, and land management need to recognize the economic benefits and critical importance of the ecosystem services model together with the natural capital that sustains the ecosystem functions from which the goods and services are derived. More widespread use is required of methodologies for quantifying and costing ecosystem services—and applying them to urban areas where the context relates directly to human needs for services such as flood regulation, noise reduction, or air quality improvement.

Ecosystem services, natural capital, and biodiversity need to be accounted for in balance sheets and commercial markets which deal with economic services and manufactured capital. Losses and gains from policies, plans, and projects can then be costed. Realistic cost/benefit analyses should inform the introduction of new planning policies and fiscal regimes, which could have beneficial or adverse effects on the biodiversity and ecosystem services of urban areas. Ecosystem services are starting to be costed (e.g. Durban, Philadelphia) and the initial results indicate the order of magnitude benefits of these services; developers need to begin to take these into account. Designers and developers are taking notice of costing carbon as part of new projects, and costing ecosystem services and biodiversity losses and gains will start once the values begin to be quantified.

Land-use planning and all projects, from individual developments to major infrastructure projects, should not compromise ecosystem services. Local planning authorities and other regulators giving land-use consents should be able to require developers to cost for loss of ecosystem services without appropriate mitigation and/or compensation. Compensation could take the form of enhancing ecosystem services elsewhere to compensate for a loss of specific services in a location resulting from a particular project.

Ecological resources and biodiversity need to be conserved for their ecosystem functions. This will require new and radical actions during construction and development projects to counteract the inevitable losses of habitats and ecological features which contribute to ecosystem services. There should be a presumption in favour of moving habitats and ecological features, such as hedges, trees, and ponds, to new locations (habitat translocation) in preference to their loss and subsequent mitigation by habitat creation. Habitat translocation retains structural components of ecological resources which can regenerate more quickly than newly planted habitats (Box & Stanhope 2010). Habitats which cannot be retained nor translocated should be recreated within new urban developments, and this should include innovative techniques such as creating green bridges to link habitats and creating green roofs and green walls on buildings (Dunnett & Kingsbury 2004).

Vibrant, innovative national programmes are required in relation to increasing urban biodiversity, with the health, enjoyment, and well-being of all the urban population at their core. Such national programmes should draw on international knowledge and practical experience of biodiversity and urban greenspaces. The ecology of urban areas, the study of the effects of their surroundings on human well-being, and the economics of ecosystem services are relatively new fields of scientific enquiry, and our state of knowledge is incomplete. Beautiful butterflies, dashing dragonflies, and ornate orchids will enthuse local residents and those who work in urban areas. But it is the fiscal incentives, regulation, and targets that have the capacity to really change behaviour amongst the public, developers, local authorities, and statutory bodies in order to provide urban biodiversity and maintain ecosystem services in urban areas.

Acknowledgements

To all those that I have worked with on urban ecology over the years, my thanks for so much good advice and many rewarding discussions about urban issues. The views in this article are my own and are not derived from any project in which Atkins is involved.

Summary

Philip James

A story that I often tell my students relates to two incidents that occurred in a major chemical factory. An operative was detailed to oversee the filling of a storage vessel. This was a large vessel and flow into it was controlled by a valve which the operative had to open and close manually. Having opened the valve, and knowing that it would be some time before the vessel was full, the operative set about other tasks. On his return he found the vessel overflowing. The incident was reported to the Production Manager responsible for that plant who arranged for the spillage to be cleared, and investigated the cause of the incident and its effects. The costs of the clean-up, making the site safe, the loss of production due to the lack of raw material etc. were born by the company. The possibility of fitting an automatic cut-off valve was discussed, it was expensive and would have to be offset against the Production Manager's budget, so he decided not to install one. The operative was then called to the Manager's office and given a good telling off and that was that. Some weeks later there was a second spillage. The Production Manager reacted differently this second time. Why? What had changed? Between these two incidents the company had introduced a new accounting system. At the time of the second incident the full costs associated with the leakage were to be taken from the Production Manager's budget. The result? An automatic valve was fitted at the next opportunity and the incident did not reoccur.

I recount this story at this point to remind us that if we look at the same problem in different ways we come up with different solutions. If we reframe a question, see it in a different context, or apply different values then we often embark on different paths. Sometimes it is just a matter of when we look at the problem that directs how we see it. In the story above, a change in accounting procedure, putting responsibility in a different place in the management structure, brought about changed behaviour. We can see this now in the drive towards proper recognition of the costs associated with Ecosystem Services. What changes might that reframing of the management of our natural and built environment bring in the future?

One of the key messages emerging from an understanding of Ecology is that change is inevitable. The concepts of balance, which dominated the textbooks from which many of us learnt our Ecology, has been replaced by the adaptive cycles put forward by C. S. Hollings in 1986. This concept is now used to link social and ecological systems and to address one of those other tricky ecological issues; that of scale. What these cycles tell us is that change is inevitable. The best we can hope to do is to steer the direction in which some of that change goes for a certain period of time. That change is inevitable is the insight we gain from thermodynamics, and in particular the second law (this is one of my favourite laws which I refer to and console myself with when I am being admonished for having let my office get untidy). The second law of thermodynamics states that entropy (disorder) in a closed system increases until equilibrium is reached and explains why change takes place. But nature is not a closed system—it receives energy from the sun and is capable of self-organization—so we have entropy driving in one direction and nature driving in another. However, set against this is the prevailing economic paradigm; one that emerged after the Second World War to kick-start the economy of Europe; one that is uni-directional, predicated on economic growth; one that runs contrary to laws of the natural environment. Behavioural economics is an emerging discipline which seeks to address the perceived shortcomings of traditional economics. Behavioural economics challenges the conventional paradigm that people act out of self-interest, are rational in the decisions they make, and act independently of others. Through a series of experiments, and you can test this yourself. Behavioural economists have demonstrated that we are not always self-interested—we are concerned with the effects of our actions on other people, that we are not always rational in our decision-making, and that we have a tendency to follow the actions of other people. In short, behavioural economists see people as humans. One of the key tenets of behavioural economics is that the framing of the issue is important, as it is that framing that influences decision-makers. The way that an issue is presented is important. From this it follows that if a problem is presented in a different way, if it is framed differently, then decision-makers respond in different ways. Behavioural economists are developing new paradigms that challenge conventional economics as they are predicated on the way that people behave. Are we about to see a revolution in the sense described by T.S. Kuhn, who wrote how science advances through step changes, not by gradual incremental change? Perhaps we shall see over the next 30 to 40 years if the traditional approach is set aside in favour of the more human approach.

Here, then, is the space within which environmental interventions take place: the space between the socio (which includes the economic) and the ecological. This space is defined by contemporary context: the challenges of population growth, climate change, peak oil, and failed economic paradigms. It is here that the chapters in this section are placed.

What we learn above all from these chapters is that if the major issues facing us at the start of the twenty-first century are seen as ecological, then we do have the ecological knowledge to address them. We have the knowledge to create environments which address the health issues we face, we have the ecological knowledge to address the imminent food shortage, we can mitigate the effects of climate change, and we can reduce our dependence on carbon based technologies: in short, we can enter the ecological age.

Concluding Remarks: The Way Forward for Urban Ecology

Jari Niemelä, Jürgen Breuste, Thomas Elmqvist, Glenn Guntenspergen, Philip James, and Nancy McIntyre

Urban ecology is making considerable progress as a scientific discipline, as has been illustrated by the chapters in this book. The book illustrated current research on ways that humans affect the urban landscape. Furthermore, the contributions discussed how an ecological understanding of urban landscapes can be used for sustainable urban planning and decision-making.

Following a chapter on the history and development of urban ecology, chapters in Section 1 presented an introduction into the basic physical elements of the urban ecosystem: climate, water, and soil, as they are modified by human activities. It also described the factors that regulate these elements, that is, the constant changes of urban land-use activities and variations in frequency, intensity, spatial extension, etc. Soils, as important elements of ecosystems, are especially intensively modified and damaged in cities. Soil sealing is the key factor for changes in the water cycle, climate, and vegetation cover, but this has not been adequately considered in urban ecosystem management. The effects of soil sealing are visible as high rates of stormwater runoff, flooding, and low infiltration rates in cities. Mitigating the effects of climatic extremes and reducing their epidemiological effects in cities will be one of the most important drivers for changing our urban environment.

The chapters in Section 2 underlined that the biota is an important defining characteristic of the urban environment. These chapters emphasized patterns in biodiversity due to changes in the underlying physical drivers resulting from urbanization and the influence of humans on resource availability, disturbance regimes, and patch structure and distribution across the landscape. Much of the total species richness of urban ecosystems results from non-native or invasive species adapted to particular environments found in cities. A conspicuous feature of urban biodiversity is that a considerable portion is both planned and introduced, occurring within remnants of the pre-urban landscape. Sudden and dramatic changes in urban species' population size and persistence are a consequence of the built environment and decisions made by humans. Although cities are a challenging environment, many species—including native ones—are able to adapt to the urban environment (and in some cases form unique ecological communities), though many others do not.

The chapters in Section 3 illustrated that urban ecosystems pose both familiar and novel ecological and evolutionary challenges to the plants and animals that live within them, with the urban biota responding in terms of diversity, abundance, trophic structure, and behaviour. Although some general responses to urbanization exist across taxa (e.g. replacement of native with exotic species, loss of large-bodied species), many responses are site- or species-specific. With an increasingly urban human population, most of our future interactions with nature will take place with urban flora and fauna, a biotic community that differs qualitatively and quantitatively from indigenous assemblages. The chapters in this section illustrated that we still have much to learn about the ecology 'of' cities, particularly for conservation and management.

As was illustrated by chapters in Section 4, there is a rapidly growing literature and awareness of the values of urban ecosystem services and strategies for improving management and governance, and linking the local to the global. Historical analyses of cities offer essential insights into the nature of socio-ecological interactions under a large number of diverse conditions and the implications of decisions that have already been made. Such historical insights are valuable but also limited, and today urban areas are in continuous and rapid change, facing enormous challenges where continuous learning and experimentation will be crucial. Everyone involved in urban development, policymakers, urban designers, scientists, and planners, need to facilitate far more experimental designs and learning about the interactions between humans and ecosystems. The ecosystem approach, a UN strategy for integrated ecosystem management, addressing equity and continuous learning, is now starting to be implemented in urban landscapes as elsewhere, providing an exciting arena for experimentation and learning about urban social-ecological systems.

In Section 5 the authors examined how urban ecological knowledge and understanding is being and will be used in our cities and towns. Collectively, the chapters highlighted the advances that are being made: new attitudes to construction, new attitudes to planning, the growth of green infrastructures and ecological frameworks within cities, and the legislative and economical changes that are driving these approaches forward. The authors of this section also identified where further work is required, where the evidence base requires strengthening, and where more integration between disciplines is called for.

The rich variety of chapters in this book improve our knowledge of urban ecosystems, though they also demonstrate that there are gaps in our understanding of the functioning of urban ecosystems and how humans interact with their biotic and abiotic environment. As Hahs *et al*. (2009) noted, we need a better understanding of the various aspects of the biotic consequences of land conversion caused by urbanization, that is, ecology 'in' cities. Furthermore, we need a more comprehensive understanding of how urban ecosystems are structured and how they function (McDonnell *et al*. 2009). In addition, we also need to better understand the interactions between ecology and society.

There are undoubtedly gaps in our knowledge, but simply more research is not sufficient to take urban ecology further as a discipline and to enhance its credibility among end-users of research results. In order to achieve these goals we need a 'holistic ecology of urban areas' which would integrate the physical, bio-ecological, and social components of urban landscapes (McDonnell *et al*. 2009). Urban ecosystems are complex, dynamic biological–physical–social entities, in which spatial heterogeneity and spatially localized feedback play a large role (Pickett *et al*. 2008). Therefore, urban ecology should be based on the plurality of concepts from multiple disciplines to address questions about mechanisms that govern urban ecosystems (Alberti 2008). Such an integrated approach would have to represent a synthesis across ecology and social sciences, and would have to include a variety of concepts and approaches from the natural and social sciences (Wu 2008a). For example, Swan *et al*. (Chapter 3.5) proposed the metacommunity concept as a way to explore how local versus regional processes shape the composition of biotic communities in urban settings. Socio-economic factors can be incorporated into the metacommunity concept to understand why species occur together, how they arrived, and what habitat conditions support their coexistence.

Integration of ecological and social aspects of urban life into a research framework is challenging and is only now taking its first steps, although there are 'heartening signs of progress' (Pickett *et al*. 2009). Such encouraging signs have recently appeared in the form of models integrating ecological and social systems in cities (e.g. Grimm *et al*. 2000, 2008; Yli-Pelkonen & Niemelä 2005; Wu 2008a; Pickett *et al*. 2009, Zipperer *et al*., Chapter 5.5), though there are several challenges in building such an integrated approach. For instance, McDonald and Marcotullio (Chapter 4.1) suggested that there is a need for a concerted effort to identify and analyse the specific costs and benefits of dense settlement organization, in other words, cities. The goal is to seek opportunities for both ecological and social solutions which may emerge through design, such as providing for natural biodiversity in cities, increasing the diversity of types of open spaces in and around cities, recycling water and increasing the energy efficiency of buildings, and improving transit infrastructures. In pursuing these goals, society must also reduce poverty and address unequal access to ecosystem services (Alfsen *et al*., Chapter 4.3).

Douglas and Ravetz (Chapter 5.1) identified a 'hierarchy of urban ecologies', with a broad view of the relationships between urban systems and ecosystems. In addition to ecology 'in' and 'of' cities, they proposed ecology 'throughout' the human–environment systems as a way to analyse interdependent processes in both human systems and ecosystems. The overarching message was that urban ecology is embedded in the life and structure of the city and the wider urban region. Echoing this approach, Zipperer *et al*. (Chapter 5.5) proposed a conceptual model linking social decision-making processes with landscape dynamics to characterize the effect of land-use on social and ecological systems.

A research agenda could assist in pinpointing where more understanding is needed for supporting the development of such models. James *et al*. (2009) presented an integrated framework for urban greenspace studies including broad research areas, such as ecosystem services, drivers of change, pressures on urban greenspace, human processes, and goals for the provision of urban greenspace. These research areas change over time and geographical area, while emergent research themes within them deal with specific issues (such as physicality, experience, valuation, management and governance) and are more constant in time and space. The proposed research agenda included 35 research questions, some of which are already being addressed. Importantly, such an integrated research agenda demonstrates that the outcomes from different research themes of urban greenspace are inextricably linked, and include physical and social systems and processes.

An ecosystem services approach could form a central element in future urban ecology by integrating ecology and social sciences in addressing urbanization issues (Alfsen *et al*., Chapter 4.3). Urban greenspaces and their associated biodiversity provide city dwellers with various kinds of ecosystem services, such as noise abatement, maintenance of groundwater, and improvement of air quality (McDonald and Marcotullio, Chapter 4.1). Furthermore, urban greenspaces produce valuable cultural services by providing residents with sites for recreation, experiencing nature, and for recuperating from physical and psychological stress (Tzoulas *et al*. 2007; Tzoulas and Greening, Chapter 5.2). These services contribute to the quality of the urban environment and also enhance the value of urban properties (Luttik 2000). Integration of approaches from ecological and social sciences—including economics—to the study of the services that urban greenspaces provide, could form a unifying theme for urban ecological research. Furthermore, making this understanding available for planners, decision-makers, and citizens would enhance urban ecology as a predictive discipline.

To conclude, the challenges for urban ecology lie in improving our understanding of urban human-ecological systems so that urban ecologists are able to better predict future changes in these systems and thereby provide guidance and support to urban decision-makers, designers, planners, and managers. In particular, urban ecologists need to produce guidance that responds to the challenges presented by climate change, by human population expansion, and by technological changes that affect people's relationships with each other and with their environment. To be able to provide meaningful predictions and guidance, urban ecologists need integrative approaches, such as those presented in this book. Addressing issues of relevance to urban planners is of the utmost importance for the credibility of urban ecology (Pauleit and Breuste, Chapter 1.1, Colding, Chapter 4.5). For these purposes, urban ecology must adopt a language that is understandable to other professions and a language that deals with the uncertainty inherent in ecology. The challenge is in producing a comprehensive framework or set of principles that can deal with the complex urban socio-ecological systems and can be used to guide the planning of sustainable cities. The chapters of this book demonstrate ways in which such principles could be developed.

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