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Urbanization, Biodiversity and Ecosystem Services: Challenges and Opportunities

A Global Assessment

Foreword by Pavan Sukhdev

Springer Open

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Urbanization, Biodiversity and Ecosystem Services: Challenges and Opportunities

A Global Assessment

A Part of the Cities and Biodiversity Outlook Project

Springer

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Foreword

We have entered the *Anthropocene* – an era when humans are a dominant geological force – and at the same time we have entered an Urban Age.[1] Over half of humanity now lives in towns and cities, and by 2030 that fraction will have increased to 60 %.[2] In other words, in slightly over two decades, from 2010 to 2030, another one and an half billion people will be added to the population of cities.

Creating healthy, habitable, urban living spaces for so many more people will be one of the defining challenges of our time. And the quality of city environments – both their built and natural components – will determine the quality of life for an estimated total of five billion existing and new urban dwellers by 2030.

Much of what gets written about the challenges of urbanization tends to be about *built* city infrastructure and its organization and governance: about transportation systems, housing, water works, sanitation, slums – the *hardware* of cities. Less is written about the *software* of cities[3] as centers of creativity and lifestyle, of culture and learning institutions that enable the creation of pools of human capital, which gather critical mass and become drivers of innovation and prosperity. And even less is written about the ecological infrastructure of cities: parks, gardens, open spaces, water catchment areas, and generally their ecosystems and biodiversity. This book *Urbanization, Biodiversity and Ecosystem Services: Challenges and Opportunities* and the Cities and Biodiversity Outlook project (CBO) addresses that gap admirably. It brings out clearly the importance of *nature* for cities, making a convincing case for internalizing ecosystem services in urban policy making.

The book not only quantifies but also lays out the complex linkages between ecosystem services and urbanization, giving us detailed case studies of cities that

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[1] London School of Economics program "Urban Age" http://lsecities.net/ua/

[2] Population Reference Bureau see http://www.prb.org/Articles/2007/UrbanPopToBecomeMajority.aspx

[3] A concept popularized by Sanjeev Sanyal & others, see for example http://www.business-standard.com/article/opinion/sanjeev-sanyal-building-bostons-not-kanpurs-110051200048_1.html

have used an ecosystem services approach, either explicitly or implicitly, in urban planning in order to address the many challenges that urbanization poses.

The problems caused by urbanization are enormous and varied. Over the last century, the migration of hundreds of millions of people from rural to urban areas in search of employment and better living conditions has not been a smooth transition. Millions have been left to live for prolonged periods in makeshift urban slums, suffering from poverty of income, health, nutrition, and safety. Constant threats of food and water scarcity have been brought about by climate change, unsustainable resource use, and inadequate planning. Cities are increasingly unsustainable, vulnerable and insecure, and therefore achieving sustainability and resilience for cities has to be high on any government's agenda. To support this necessary and important focus, the book delivers key messages to policy makers and showcases many instances of smart urban planning that have made use of nature and its services to alleviate or solve some of these problems. In the process, this book redefines cities from being centers of economic growth and consumption to places generating human well-being and even creating positive externalities.

Ecosystem services can address many of the challenges that cities increasingly face, and the false dichotomy between environment and development is nowhere as easy to disprove as in cities. Clean air, safe drinking water, and protection from climate change effects are all highly relevant to human development in cities, and many forms of poverty are caused or exacerbated by a lack of access to these ecosystem services. Furthermore, cities consume tremendous amounts of resources and thus generate large amounts of waste and emissions. These negative externalities of urban growth are borne disproportionately by the income poor, who do not have access (or the means) to procure clean drinking water and health services. The role of natural areas in providing catchment for stable and cheap drinking water cannot be overemphasized – almost a third of the 100 largest cities have proximate natural areas that provide this service. Furthermore, green spaces in or near cities also deliver services such as air purification, temperature regulation, groundwater recharge, and cultural services including aesthetics and recreation, all leading to healthier lifestyles.

Urban biodiversity and ecosystems deliver myriad other benefits, from underpinning social and economic development to climate change mitigation and adaptation. Wetlands can treat stormwater runoff and also offer biodiversity and recreational services. Local food production in cities is an exciting and evolving dimension of cities, and it can both decrease the emissions externality of cities and also improve food security. Restoration and management of near shore ecosystems such as mangroves can reduce impacts of storm surges, decrease climate change vulnerability, and increase resilience.

It is recognized that urban consumption patterns not only adversely impact nearby ecosystems but also ecosystems further away: urban teleconnections and the ecological footprint of cities are geographically dispersed and indeed immense. However, cities cannot be viewed as problematic merely because they form a large consumer base. They also hold the key to changing production and resource use – by decreasing waste production, increasing recycling, and moving citizens to more sustainable forms of consumption. Furthermore, energy-efficient and renewable-energy infrastructure development through economies of scale can reduce emissions.

It the context of such a complex web of issues, problems and solutions, it is important to examine and quantify, as this book has done, both the consequences and future trajectories of urbanization. This can lead us to identify both challenges and opportunities that cities must address in order to be sustainable and indeed viable centers of human habitation and progress. The volume also addresses metrics for urban biodiversity, an evolving space in research and practice.

The book delivers a valuable contribution to integrating knowledge about biodiversity and ecosystem services into urban design and planning. This is essential to ensure both the sustainability and resilience of cities for an 'Urban Age' that is human civilization's present as well as its future.

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Preface

While there is growing awareness that cities affect almost every ecosystem on earth, significantly contribute to the loss of biodiversity, and are increasingly vulnerable to environmental change, a global analysis of the environmental impacts of urbanization has been lacking. While previous studies have examined particular cities or a particular facet of the urban environment, few attempts have been made to assess the prospects for supporting ecosystem services on an urbanized planet. On the one hand, the Millennium Ecosystem Assessment (MA), the world's largest assessment of ecosystems, covered almost every ecosystem in the world but made few references to urban areas. On the other, the World Development Report, the world's largest assessment of urbanization published by the World Bank annually, makes few references to ecosystems. It is this knowledge gap we attempt to bridge by this book and the *Cities and Biodiversity Outlook (CBO)* project at large.

The production of the book has been called for through paragraph six of Decision X/22 of the tenth meeting of the Conference of the Parties (COP 10) to the Convention on Biological Diversity (CBD) in Nagoya 2010. The decision initiated two publications. The first publication, *Cities and Biodiversity Outlook – Action and Policy*,[4] intended for policy makers, was launched at the COP11 meeting of the CBD in Hyderabad in October 2012. The *CBO – Action and Policy* showcases best practices and lessons learned from cities across the world, and provides information on how to incorporate the topics of biodiversity and ecosystem services into urban agendas and policies.

The current book – *Urbanization, Biodiversity and Ecosystem Services: Challenges and Opportunities* – is the more detailed scientific portion of CBO and the first assessment ever conducted that addresses global urbanization and the multiple impacts on biodiversity and ecosystem services. It has been written and edited by an international team of scientists and includes several of the authors who previously participated in one or both of global assessments: the *Millennium Ecosystem Assessment* (MA) and *The Economics of Ecosystems and Biodiversity* (TEEB). *Urbanization, Biodiversity*

[4] Secretariat of the Convention on Biological Diversity (2012) *Cities and Biodiversity Outlook*. Montreal, 64 pages. http://www.cbd.int/en/subnational/partners-and-initiatives/cbo

and Ecosystem Services: Challenges and Opportunities describes and analyses multiple dimensions of urbanization, focusing on how the processes affect patterns of biodiversity and ecosystem services within as well as outside city boundaries. It is therefore an assessment of the process of urbanization, rather than an assessment of cities *per se*. Further, it focuses on the biosphere and analyses how the living environment is impacted in a rapidly urbanizing world, and explores connections to human well-being and how an increasing urban population may succeed or fail to develop mechanisms for reconnecting with the biosphere. Thus, this book is not an assessment of all the challenges connected with urban growth, such as e.g., challenges linked to management of waste, energy and transportation.

Our aim has been to make a thorough synthesis of current knowledge and frame this in a policy relevant context with the intention of stimulating a vigorous debate on how urban challenges could be addressed. However, even more importantly, we have aimed to encourage a debate on how the many opportunities created by urbanization could result in innovative policy for more sustainable development on a global scale. This book is about the imperative of reconnecting cities to the biosphere; it explores urban areas as social-ecological systems and the social-ecological foundation of cities and their sustainability. It details how this urban ecological embedding may be facilitated through a new and bold urban praxis.

One challenge when starting the assessment was that the concepts of *urbanization* and *urban biodiversity* are not well defined. There is no general agreement on what is urban, and considerable differences in classification of urban and rural areas exist among countries and continents. We have in the CBO used working definitions and define *urbanization* as a multidimensional process that manifests itself through rapidly changing human populations and changing land cover. Urban growth is due to a combination of four forces: natural growth, rural to urban migration, massive migration due to extreme events, and redefinitions of administrative boundaries. With *urban biodiversity* we refer to the biological variation at all levels from genes to species and habitats found in urban landscapes. Several aspects of biodiversity differs compared with biodiversity in other areas, e.g., there is often an extreme patchiness and large point-to-point variation over short distances, and composition of species is often dominated by non-native species introduced for specific purposes. Urban biodiversity therefore often represents a biodiversity *intentionally designed by humans for humans*. The multiple dimensions of this have been overlooked in both ecology and in social sciences, and contributing to bridge this knowledge gap constitutes another important rationale for the CBO project.

The book has a global scope but it also makes a strong connection to the regional and local scales. In addition to *Regional Assessments* of urbanization in Africa, Asia with special focus on China and India, Latin America, Oceania, North America and Europe, *Local Assessments* come from a number of cities: Bangalore, Cape Town, Chicago, İstanbul, Melbourne, New York, Rio de Janeiro, Shanghai, Stockholm and urban *satoyama* and *satoumi* landscapes in Japan. The regional assessments reflect the broad scope of current and expected future urbanization trends around the world. The cities represented in the local assessments were selected because they represent areas where urbanization processes and social-ecological systems have been established fields of research for some time.

One crucial issue apparent when starting this project was that much relevant information on urban development, biodiversity and ecosystems, particularly at the local scale, tend to occur in non-peer reviewed literature. We have nonetheless excluded references to the bulk of non-peer reviewed literature such as unpublished reports, conference abstracts and other non-peer reviewed literature, but in a few instances included references to technical reports and policy documents when these have been judged to be highly relevant.

The publication represents a collaborative effort among a large number of scholars, the CBD, and Stockholm Resilience Centre (SRC) at Stockholm University, and includes significant input from ICLEI – Local Governments for Sustainability. An Inter-Agency Task Force and an Advisory Committee (see Appendix), as well as the Global Partnership on Local and Sub-National Action for Biodiversity have provided valuable oversight of the entire process. Nearly 200 scientists and practitioners have been involved as authors or reviewers in the entire CBO project and we are very grateful for their contributions. We thank members of the pan-European project URBES (Urban Biodiversity and Ecosystem Services) for contributing with the scientific input as well as perspectives from policy and practice. We thank Oliver Hillel, Andre Mader, Chantal Robichaud, David Ainsworth and Fabiana Spinelli at the Secretariat of the CBD, Elizabeth Pierson the Technical Editor of the *CBO – Action and Policy*, Andrew Rudd from UN-Habitat and Russell Galt, Kobie Brand and Georgina Avlonitis at ICLEI for their enormous contributions during the development of the CBO project. We also want to thank Femke Reitsma at the University of Canterbury, Jerker Lokrantz at Azote, and Félix Pharand-Deschênes at Globaïa for excellent help with the design of figures and illustrations. We extend our gratitude to Audrey Noga, Katie M. Hawkes, Megan Meacham and Laia d'Armengol, for invaluable assistance with the texts in the project's final phase. The project has intellectually benefitted from discussion with numerous members of DIVERSITAS, IHDP and specifically members of the Urbanization and Global Environmental Change Project (UGEC) at IHDP as well as with members of the research network URBIO. The framework on cities representing complex social-ecological systems has, over the years, developed significantly within the urban group in the Resilience Alliance and the Urban theme at SRC, and we would like to specifically thank Carl Folke, Johan Colding, Erik Andersson, Stephan Barthel, Guy Barnett, Sara Borgström, Åsa Gren, Charles Redman, Brian Walker and Maria Tengö. We also want to thank UNESCO and specifically Christine Alfsen for pioneering several ideas and initiatives, including URBIS (the Urban Biosphere Initiative), applying the ecosystem approach to urban landscapes. The CBO project has benefitted much from the kind contribution by the African Center for Cities at the University of Cape Town (UCT) in South Africa. UCT hosted an important workshop in February 2012 with participants from several African countries, which resulted in a significant contribution to the understanding of urbanization processes in Africa. A special thanks to Pippin Anderson for assisting with the organization of the workshop. We also want to thank Stellenbosch Institute of Advanced Studies (STIAS) for providing a

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Chapter 1
A Global Outlook on Urbanization

Karen C. Seto, Susan Parnell, and Thomas Elmqvist

1.1 Introduction

This volume is based on the argument that, just as it is no longer possible to construct sound ecological science without explicit attention to urbanization as a key driver of global ecological change (Chaps. 3, 11, and 26), cities can no longer be uncoupled from a full understanding of their ecological foundations. The populations and economies of urban areas rely on hinterlands for resources, but there is a disconnect between using resources for urban areas and preserving or conserving ecosystem services that are outside of urban areas (Chaps. 2 and 3). While it is recognized that urban areas and urban dwellers will need to begin to take greater responsibility for stewardship of Earth's resources (Seitzinger et al. 2012), urban sustainability efforts often are prone to localism, thus failing to take into account the need to conserve resources elsewhere (Seto et al. 2012a).

A history of disassociation of biodiversity, ecosystems, and urban development alongside a belief in technological solutions gave rise to a logic of urban planning that made it possible to imagine that the governance of urban life could be

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separated from the provision of food, water and other ecosystem services on which all human life depends (Chap. 2). As a result, urban areas today are both more tightly coupled to their ecological hinterlands than ever before and yet it is difficult for urban residents and their representatives to manage urbanization sustainably. This book is about the imperative of reconnecting cities to the biosphere; it explores a fresh understanding of the social-ecological foundation of cities and their sustainability, and details how this urban ecological embedding is being facilitated through a new and bold urban praxis.

1.2 Five Major Urban Trends

Throughout the book we will elaborate on five major trends in the urbanization process, which have implications for biodiversity and ecosystem services:

1. The physical extents of urban areas are expanding faster than urban populations, suggesting that the world will require increasingly more land to build cities and supply urban consumption as urban populations continue to increase. In some urban areas that are shrinking in population or economic activity, new and emerging challenges are associated with vacant or abandoned land and buildings. Aggregated to the city level, the size of these unused areas can present new opportunities for vegetation regrowth and challenges for urban renewal.
2. Urban areas modify their local and regional climate through the urban heat island effect and by altering precipitation patterns, which together will have significant impacts on net primary production, functions of ecosystems, and biodiversity.
3. Expansion of built-up areas will draw heavily on natural resources, in particular water, timber, and energy. The continued outward growth of cities will often consume prime agricultural land, with knock-on effects on habitats, biodiversity and ecosystem services elsewhere.
4. Urban land expansion is occurring fast in areas adjacent to biodiversity hotspots and faster in low-elevation, biodiversity-rich coastal zones than in other areas.
5. Most future urban expansion will occur in areas of limited economic development and institutional capacity, which will constrain abilities to invest in the protection of biodiversity and the conservation and restoration of ecosystem services.

Here, in the introductory chapter, we will expand on trends 1 and 2 as a foundation for the coming chapters to elaborate on trends 3–5.

1.2.1 Trend 1: Urban Areas Are Expanding Faster Than Urban Populations

The global proportion of urban population was a mere 13 % in 1900 (UN 2006). It rose gradually to 29 % in 1950. If current trends continue, by 2050 the global urban population is estimated to be 70 % or 6.3 billion, nearly doubling the

3.5 billion urban dwellers worldwide in 2010 (UN 2010), and most of the growth is expected to occur in small and medium-sized cities, not in mega-cities (Chap. 21). Biodiversity and ecosystem services do not represent the immediate concern for the approximately 900 million people who live in slums with lack of basic services, substandard housing, and unhealthy living conditions (UN-Habitat 2003). At the same time, overall levels of urban residents' consumption are rising, placing greater strain on the resource base and increasing the imperative to allocate natural assets fairly and equitably. While mega-cities are the focus of much attention, it is the medium-sized cities (with populations of 1–5 million) that will experience the fastest rates of urban growth, and in fact most of the world's urban population will live in small cities of less than one million by 2050 (Chap. 21).

Urbanization is a complex and dynamic process playing out over multiple scales of space and time (Grimm et al. 2008a, b). Historically, cities have been compact and have concentrated populations. Today, cities are increasingly expansive. Across the world, urban areas are growing on average twice as fast as urban populations (Seto et al. 2011; Angel et al. 2011). In addition to being increasingly physically expansive, urban land change is also predominantly characterized by peri-urbanization, the process whereby rural areas both close to and distant from city centers become enveloped by, or transformed into, extended metropolitan regions (Simon et al. 2004; Aguilar et al. 2003). This results in a tight mosaic of traditional and agricultural juxtaposed with modern and industrial land-uses and governance systems (Chaps. 8 and 26). As a physical phenomenon, peri-urbanization involves the conversion of agricultural land, pastures, and forests to urban areas. As a social phenomenon, peri-urbanization involves cultural and lifestyle adjustments of agrarian communities as they become absorbed into the sphere of the urban economy. In developing countries, especially in Asia and Africa, peri-urbanization is the most prominent form of urban growth and urbanization, with different characteristics across countries and regions. As a result, emerging urbanizing regions represent probably the most complex mosaic of land cover and multiple land uses of any landscape (cf. Batty 2008a, b).

What Is Urban?

There is no general agreement on a definition of what is urban, and considerable differences in classification of urban and rural areas exist among countries and continents. Most comparative assessments use national definitions, even though these are not comparable measures. In Europe and North America, the urban landscape is often defined as an area with human agglomerations and with >50 % of the surface built, surrounded by other areas with 30–50 % built, and overall a population density of more than ten individuals per hectare. In other contexts, population size, the density of economic activity or the form of governance structure are used to delineate what is a town, city, or city region, but there

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is significant variation in the criteria for defining what is urban. In this book, we use a *working definition of urbanization as a multidimensional process that manifests itself through rapidly changing human population and changing land cover*. The growth of cities is due to a combination of four forces: natural growth, rural to urban migration, massive migration due to extreme events, and redefinitions of administrative boundaries.

Understanding and disaggregating the demographic transitions associated with the future urban world is an essential step in assessing the ecological impact of cities (Chap. 21). For now, in the absence of robust sub-national census information on migration or fertility, all the urban data need to be treated as indicative.

While everyone struggles to define exactly what is meant by a city, nobody negates the shifting patterns of urbanization or the overall growth of cities. In this volume we introduce the framework of cities as complex social-ecological systems (see Chaps. 2, 11, and 33), since they include much more than a particular density of people or area covered by human-made structures.

There is significant variation in urbanization across and within countries and it is important to recognize that there is no single "urban transition." For example, Brazil's urban population reached 36 % in 1950, whereas India's urban population is currently at 31 %. In Russia, central planning led to a high proportion of large cities relative to small ones, and disproportionate urban primacy (Becker et al. 2012). Rates and periods of urbanization, cultural patterns of land use and the biophysical conditions that urban managers face vary tremendously.

Although cities have existed for centuries, the urbanization processes today are different from urban transformations of the past in significant ways, the most important of which are:

(1) *the scale*, (2) *the rate* and (3) *the shifting geography* of urbanization.

1. *The scale of urbanization* is unparalleled in terms of urban population size, urban extent, and the sheer number of large urban areas (Seto et al. 2010). Today's cities are bigger than those at any other time in history in terms of their populations. In 1900, there were no cities with a population of ten million. Today, there are 19 urban agglomerations with populations of ten million or more; Tokyo-Yokohama has a population of nearly 40 million (see Fig. 1.1). Urban areas have also become extraordinarily large in physical size. The urban extent of Tokyo-Yokohama covers 13,500 km^2, an area that is bigger than Jamaica (11,000 km^2). The number of large cities is also unparalleled. At the start of the 1800s when the world population was around one billion, Beijing was the only city with a population of one million. Today, there are nearly 400 cities with populations of over one million.

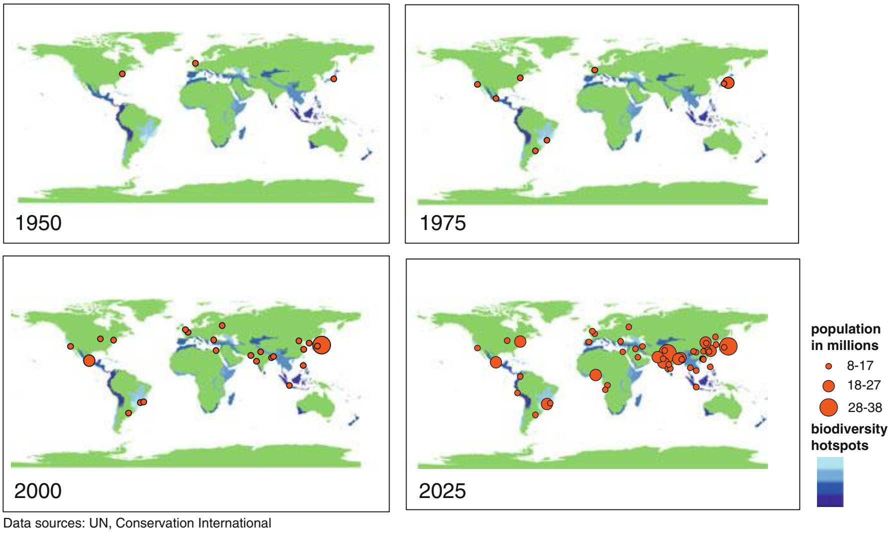

Fig. 1.1 Global urbanization and biodiversity hotspots, 1950–2025. For a definition of hotspots, see Chap. 22 (Reproduced from Secretariat of the Convention on Biological Diversity 2012, p. 8. Prepared by and published with kind permission of © Femke Reitsma 2012. All rights reserved)

2. *The rate of urbanization* is a major characteristic of this century. It took all of history until 1960 for the world urban population to reach one billion, but only 26 additional years to reach two billion. In cities and towns across the developing world, the growth of urban population translates into everyday challenges for city managers and residents as they seek to ensure the physical infrastructure and resource supplies on which new urban residents' livelihoods will depend. While some aspects of household security and economic opportunity can be achieved through individual effort, living in a city inevitably implies some dependence on collective organization. There is considerable debate on how much government or what form of government is most appropriate to a sustainable city and even how government should work with others to enable intergenerational urban opportunity while protecting the environment (Chap. 27). As several of the forthcoming chapters and the city case studies illustrate, significant energy and commitment is being directed toward finding ways to work together to reconfigure the governance of cities, city regions and the network of cities in ways that enhance rather than detract from Earth's biodiversity.
3. *The geography* of urbanization is *shifting*. The world's 20 fastest-growing urban regions are in Asia and Africa, not Europe or North America (Chaps. 4, 13, 14, 15, and 28). The urban transition in Europe and South America occurred in the 1950s through the 1970s (Chaps. 13, 14, 15, and 28). Urban growth in the coming decades will take place primarily in Asia (China and India in particular) (Chaps. 4, 5, and 6) and in Africa (especially Nigeria) (Chap. 23) and expand into farmland, forests, savannas and other ecosystems. Whereas the urbanization

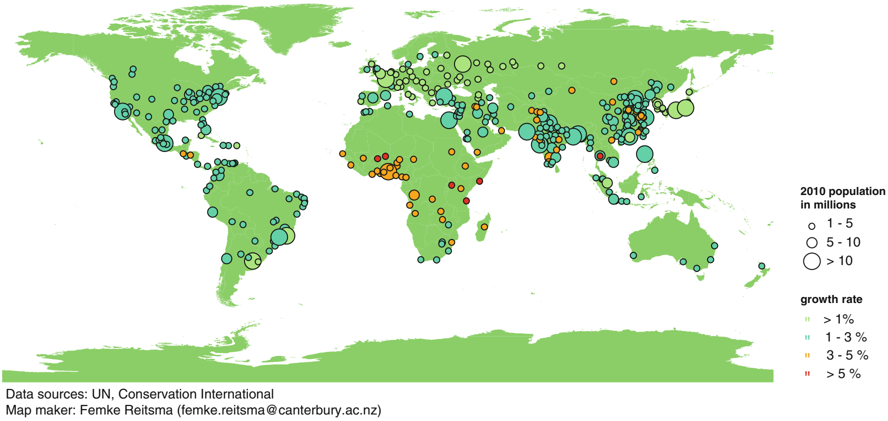

Fig. 1.2 Predicted urban growth from 2010 to 2025 for cities that have a population of greater than 1 million people in 2010 (Reproduced from Secretariat of the Convention on Biological Diversity 2012, p. 12. Prepared by and published with kind permission of © Femke Reitsma 2012. All rights reserved)

levels in the Americas and Europe are already high—80 % in South America and 75–78 % in Europe and North America—the urban populations on the continents of Africa and Asia are less than 40 % of total population. Over the next two decades, while the rural population will also rise, the urban populations of both continents are expected to increase to more than 50 %, and parts of Africa and Asia will have urban growth rates of more than 5 % (Fig. 1.2). The location of urban land change will parallel these changes in population growth. China and India will experience significant expansion of urban built-up area, as will Nigeria (Chap. 21). A majority of these new urban residents will be relatively poor, with estimates that between one-quarter and one-third of all urban households in the world will live in absolute poverty (UNEP 2002).

When analyzing the most vulnerable areas, it is clear that coastal ecozones are important and predominantly urban (McGranahan et al. 2005, 2007), and in particular are home to the largest cities. Globally, approximately 400 million people live within 20 m of sea level and within 20 km of a coast (Small and Nicholls 2003). Many large cities occupy coastal locations that are flood prone and vulnerable to extreme events, although there is a wide range in the distribution of vulnerability across cities and even among different communities within cities (Parnell et al. 2007). Hurricane Sandy in 2012 and the Asian tsunami in 2004 showed that all cities—even those in wealthy countries—are vulnerable to disasters and extreme climate events, and that coping capacity and resilience differ significantly among cities. In cities of the developing world, adapting to increased risk is understandably more difficult, not just due to the limits on resources. In many African and Asian cities and towns, local officials rarely have full knowledge of, or control over, the evolving urban form because planning and enforcement capacity is weak or illegitimate (Chaps. 7, 8, 24, and 29). High levels of informality in urban areas may even make parts of the city impenetrable, compounding vulnerability and precluding the use of ecosystem based adaptation to risk.

Despite the importance of urbanization as a defining trend in the twenty-first century, we lack critical information and data about urban areas and urbanization processes. For example, while the UN *World Urbanization Prospects* publications provide country-level information on the percentage of populations in urban areas, they do not supply intra-country variations of urban population distribution, the location of urban areas, or changes in urban areas (for a detailed discussion on themes that appear in debates over data and methodologies utilized for generating global population projections, see Box 21.1). Furthermore, information about quality of life and basic socioeconomic variables such as education and equity are not available or collected across cities in a systematic fashion. There have been recent efforts to develop comparable city and urban indicators that measure a range of urban services (e.g., Global City Indicators Facility, UN Global Urban Indicators), but these efforts are only now underway and developing time series will take years in the making. Even so, additional challenges are presented in that these undertakings tend to represent larger cities rather than all cities and towns.

1.2.2 Trend 2: Urban Areas Modify Their Local and Regional Climate

Cites are not just subjected to risk, they are also drivers of changes in climate and ecosystems. Land-cover changes associated with urbanization have considerable impacts on temperature and precipitation in and around urban areas (Seto and Shepherd 2009). The most studied manifestation of urban modification of regional climate is the urban heat island (UHI). The conversion of vegetated surfaces to hard-made surfaces modifies the exchange of heat, water, trace gases, and aerosols between the land surface and overlying atmosphere (Crutzen 2004); this leads to the "urban heat island effect," characterized by elevated daytime and nighttime temperatures in and near urban areas (Oke 1974; Arnfield 2003) compared to surrounding regions. The urban heat island effect has been documented for nearly 100 years (Howard 1833) and is affected by the shape, size, and geometry of buildings as well as the differences in urban and rural gradients. The role of the urban heat island in regional climate has been the subject of numerous investigations. However, the impact of urbanization on regional climate extends well beyond the UHI. The concentration of activities (e.g., transport, industrial production) in urban areas produces patterns of aerosols, pollution, and carbon dioxide that are more highly concentrated in urban areas than in non-urban, rural areas (Pataki et al. 2007). Aerosols affect regional climate by scattering, reflecting, or absorbing solar radiation. Whether aerosols produce a cooling or warming effect depends on the aerosols in question: sulfates produce a cooling effect while carbon-based aerosols produce a warming effect. There is mounting evidence that urbanization affects precipitation variability, a phenomenon described as an "urban rainfall effect" (Shem and Shepherd 2009).

In some parts of the world there is an observed increase in regional precipitation due to urbanization, while in other regions there is a measurable decline in precipitation.

In addition to the UHI and urban rainfall effects, urbanization significantly affects terrestrial carbon cycle by reducing net primary productivity (NPP). In China, regional annual primary production decreased by 14 % during the 1991–2001 period (Xu et al. 2007) and in some localized cases in South China, resulted in an average annual reduction of 45.93 Gg of carbon (Deyong et al. 2009). In the United States, NPP losses from urbanization alone are roughly equivalent to about 6% of the annual caloric requirement of the U.S. population (Imhoff et al. 2004).

1.2.3 Trend 3: Urbanization Increases Demands on Natural Resources

Urban expansion affects the demand for natural resources required for the construction and operation of built environments. Studies show that increases in energy and material use efficiencies at the building scale can substantially reduce energy consumption and resource demand (Gustavsson and Sathre 2006; Fernández 2007). These studies emphasize technological and efficiency improvements, but neglect the scale and spatial configuration of urban land use. At the metro region scale, there is emerging consensus that compact urban development can reduce demand for raw materials from buildings and infrastructure (Wheeler 2003; Sovacool and Brown 2010). Studies that examine both improvements in efficiency and scale of urbanization show that gains in efficiency at the building scale are often overshadowed by the sheer magnitude of urban expansion (Güneralp and Seto 2012). Moreover, changes in lifestyles and consumption patterns associated with urbanization, especially increasing demand for residential energy and water, is placing dramatic pressures on ecological services (Hubacek et al. 2009).

1.2.4 Trend 4: Urban Expansion Is Increasing Near Biodiversity Hotspots

By 2030, new urban expansion will take up an additional 1.8 % of all biodiversity hotspot areas (Seto et al. 2012b). Case studies from around the world show that urban expansion in and near critical habitats is ubiquitous both in developing and developed countries (Wang et al. 2007; Pauchard et al. 2006) (Fig. 1.1). Almost 90 % of the protected areas likely to be impacted by future urbanization are in rapidly developing low- and moderate-income countries (McDonald et al. 2008). Five biodiversity hotspots are forecasted to have the largest percentage increase in adjacent population and highest probability of becoming urbanized by 2030: the Guinean forests of West Africa, the Caribbean Islands, Japan, the Philippines, and the Western Ghats and Sri Lanka (Seto et al. 2012b). Worldwide, 32,000 km^2 of protected areas were urbanized circa 2000, representing 5 % of global urban land (Güneralp and Seto 2013). In Europe, where there is an extensive protected area network, more than 19,000 km^2 of protected areas were urbanized circa 2000. That is, 13 % of the total urban land in Europe was located in protected areas. China and South America also had substantial amounts of urban land within their protected areas with 4,500 and 2,800 km^2, respectively (i.e., 6 and 3.5 % of their respective urban lands).

Recent analyses show that there will be substantial growth in urban land across the world near protected areas in the next couple of decades (Fig. 1.1). In general, the largest increases in the amount of urban land near protected areas are forecasted in developing countries and emerging economies. The greatest increases in urban land around the protected areas will take place in China with the amount of urban land increasing as much as three to seven times over 30 years (Güneralp and Seto 2013) (Chap. 5).

1.2.5 Trend 5: Urbanization Influences the (Green) Economy

There is an evident trend towards economic reasoning in the sustainability agenda, although clearly not all dimensions of a city's ecology can (or should) be expressed in monetary terms or in terms of fiscal risk or economic return (Chap. 11). There is some concern that the green agenda, in which biodiversity and climate change are key drivers, has become dominated by an economic rationale (Marvin and Hodgson 2013). Nevertheless, across the cities of the world, governments and private developers have turned increasingly to defining economic value of ecosystem services in hopes to drive a greater integration of ecological principles of urban design, construction and management.

What is meant by 'the green urban economy' is deeply influenced by context. At a very general level it is possible to detect three overlapping tendencies in the green economy agenda that relate directly to how urban biodiversity challenges are understood. First, as the dependent relationship between the quality of the natural environment of cities and the quality of urban life and urban livelihoods becomes more apparent, the economic value of biodiversity becomes more obvious, but the means for raising revenue to protect these ecological amenities for public access often remain opaque. Second, austerity and the social desire to reignite the economy, especially in Europe and North America, has placed expectations of growth on the introduction of green economic innovations (Marvin and Hodgson 2013). More generally, but especially in the growing cities of Asia and Africa, the economic anticipation of a bigger emphasis on urban biodiversity and ecosystem services extends beyond the trend of green construction and incorporates also the growth of green production, distribution and consumption.

Third, for cities in the global south, the urban management of ecological resources, even when there are potential returns on investment, is complicated because both urban management capacity and revenues are limited. But the problem is not only local, raising significant governance problems. In China and regions of India, international capital, especially multi-national corporations, international real estate developers and property management firms have become major actors in shaping local patterns of development (Seto et al. 2010), and the influx of international capital also increasingly affects local urban consumption patterns (Davis 2005).

1.3 Cities and Their Dependence on the Biosphere

After decades of mutual neglect and an artificial divide between nature on the one hand, and cities on the other hand, there is now a shift in ecological science to include urban places as integrated components of long-term resilience (Chaps. 17, 18, and 19). Urban planners are also increasingly acknowledging that cities have an important role as stewards of the ecosystems on which they depend and that functioning natural systems such as watersheds, mangroves, and wetlands are indispensable for supporting health and vibrant livelihoods as well as reducing urban vulnerabilities to natural disasters. This will be the theme of much of the rest of the book, starting with a historical overview of how the urban disconnection from the biosphere gradually emerged and accelerated, followed by a look at more contemporary efforts that have begun to reconnect cities to the ecosystems upon which they depend.

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Chapter 2
History of Urbanization and the Missing Ecology

Thomas Elmqvist, Charles L. Redman, Stephan Barthel, and Robert Costanza

Abstract In this chapter, we explore the historical dimension of urbanization and why the ecology of urbanization has, until recently, been missing. We discuss the consequences of this for our perceptions of urbanization throughout history and also discuss the emerging reintroduction of ecology and the concept of natural capital into the global discourse on urbanization and sustainability. Humans and the institutions they devise for their governance are often successful at self-organizing to promote their survival in the face of virtually any environment challenge. However, from history we learn that there may often be unanticipated costs to many of these solutions with long-term implications on future societies. For example, increased specialization has led to increased surplus of food and made continuing

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urban growth possible. But an increased urban—rural disconnection has also led to an alienation of food production from the carrying capacity of the land. While connections and feedbacks with the hinterland that supported growing urban centres were often apparent in the past, this has increasingly been lost in a globalized world. The neglect of a social-ecological perspective and the current disconnect between the urban and the rural risks mean that important feedback mechanisms remain invisible, misinforming policy and action with large consequences for global sustainability. We argue that through reintroducing the social-ecological perspective and the concept of natural capital it is possible to contribute to a redefinition of urban sustainability through making invisible feedbacks and connections visible.

2.1 Human History and Urbanization

History offers many lessons relevant to sustainability by exhibiting how humans and their societies have recognized and responded to challenges and opportunities of their natural environment (Redman 1999; Diamond 2005; Costanza et al. 2007a; Sinclair et al. 2010). Three of the basic approaches to problem solving in antiquity were: (1) mobility of people to available resources, (2) ecosystem management to secure enhanced local growth of produce, and (3) increasing social complexity encoded in formal institutions that guided an expanding range of activities. These solution pathways were fundamental to the rise of early civilizations and are instrumental for integration in the design of sustainable cities in the future (Redman 2011).

2.1.1 Three Approaches to Human Problem Solving and the Emergence of Cities

The first approach, mobility of people to available resources, has been the dominant way of securing adequate subsistence for the vast majority of the human enterprise. Until 10,000 years ago (and more recently in many regions) virtually all people had to move among several locations each year to take advantage of the seasonality of ripening resources and variation in water availability. The dominance of this pattern was only broken by the introduction of agriculture that allowed the establishment of year-round settlements in many regions of the world. Agriculture is thus an example of the second approach to problem solving, ecosystem management for enhanced productivity. This has proven to be an astonishingly successful solution to feeding an ever-increasing global population and to enabling virtually all people to live in permanent settlements (for an overview of human and agricultural development and links to other events through human history, see Fig. 2.1). In fact, the implementation of agriculture and the infrastructural improvements made to enhance productivity were strong incentives for the spread and growth of sedentary communities. A highly effective human-nature relationship emerged from millennia of experimentation—i.e.

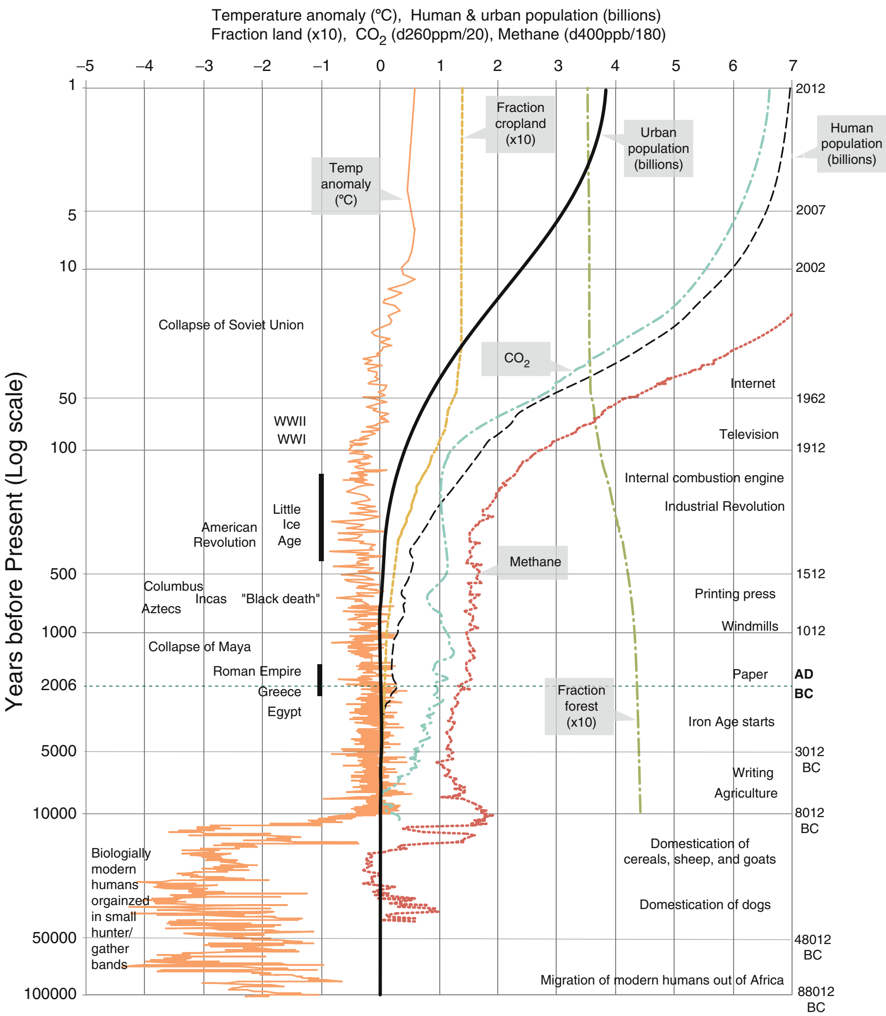

Fig. 2.1 Overview of human history, urban growth, development of agriculture, technology and industry as well as corresponding links to economic growth (GDP), environmental changes and changes in land use (Modified after Costanza et al. 2007a. Published with kind permission of © The Royal Swedish Academy of Sciences. All Rights Reserved)

the village farming community—and became the dominant settlement form across the globe. Small settlement sizes, flexibility in the sources of subsistence, and a balance between extraction from and the regeneration of the local ecosystem made this the most enduring and widespread community type. Although it existed as early as 9 or even 10,000 years ago in the Near East, the concept spread or was reinvented, and similar farming communities housed over half the world's population as recently as the middle of the twentieth century.

The village farming community proved to be a highly resilient socio-economic unit, yet some of these communities expanded on their approach to ecosystem management to the point where larger aggregations of population were necessary to supply the required labor.

A third approach to problem solving emerged, however, when larger populations required a transformation in the social order, which was largely achieved through innovations in social complexity. This is at the heart of what scholars call the Urban Revolution and it appears to have occurred first in Mesopotamia (Childe 1950; Redman 1999). The formation of the first cities and their linking together as one civilization on the Mesopotamian plain was relatively rapid, considering the scope of the social and technological changes involved. In about 5500 BC, only 2,000 years after the earliest known occupation of this region, cities emerged, and writing and other traits of urbanism such as monumental buildings and craft specialization had appeared. The rise of cities is not simply the growth of large collections of people—rather, it involves communities that are far more diverse than their predecessors and more interdependent. Relative independence and self-sufficiency characterized village farming communities, but it also limited their growth. Specialization in the production of various goods and complex exchange networks represented one way in which urban societies were able to grow. Cities were dependent on their hinterlands of surrounding towns and villages and developed ways to extract goods and services from their neighbors (see left panel in Fig. 2.2). It is clear that technological inventions such as effective irrigation agriculture, the manufacture and widespread exchange of goods, and the advance of science and mathematics were fundamental to the growth of cities. In turn, cities became and continue to be centers of innovation. Moreover, new inventions in the social realm, such as class-structured society, formalized systems of laws, and a hierarchical territorially-based government made cities possible and have continued to characterize their operation.

2.1.2 *Early Development of Cities*

The landscape-productivity-human relationship evolved in villages and towns; this enabled the growth of large, diverse populations that would aggregate into what are now called cities. The cities of antiquity in Mesopotamia and other regions responded to the specific opportunities and constraints of their local social and ecological environment, yet general patterns emerged that share commonality with contemporary cities and may provide useful insights (Simon 2008; Smith 2012). The hallmarks of cities are: (a) a large population that (b) aggregates in a central location with (c) buildings and monuments that (d) represent institutions that organize and facilitate productivity. From the earliest times in Mesopotamia and in other regions, aggregations of people and their wealth have been threatened by military hostilities and they have repeatedly sought refuge behind strong defensive fortifications (Redman 1978). This has led to densely packed cities behind defensive walls, but at the same time growing rural to urban migration has led to settlements spreading

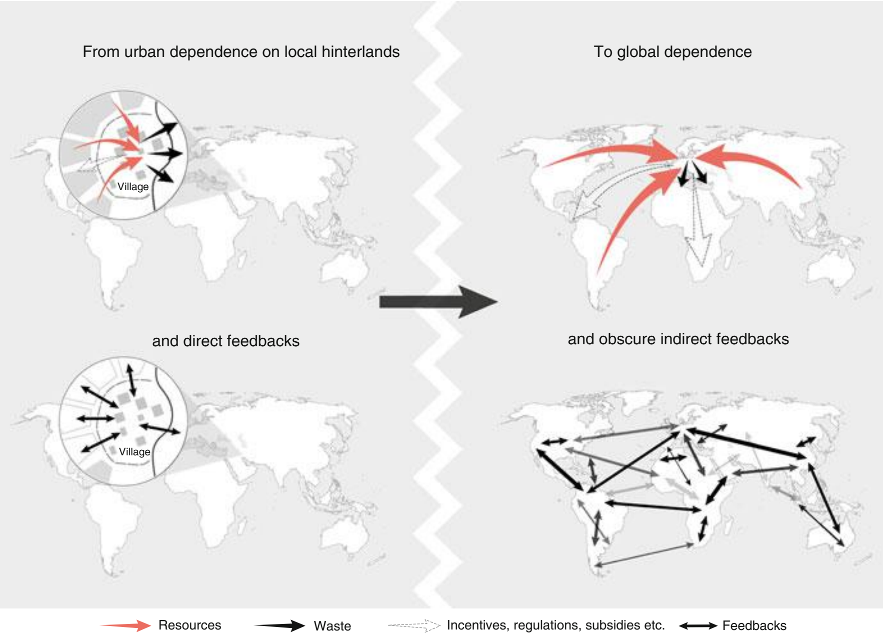

Fig. 2.2 Urban centers have moved from being more directly linked to their hinterlands and resource base to a situation where food and other resources are transported across the globe resulting in complex and often masked feedback mechanisms (Prepared by and published with kind permission of © Jerker Lokrantz/Azote 2013. All Rights Reserved)

outside the walls, a phenomenon that today one might call sprawl. This pattern of densely packed housing and central institutions within the walls, and residential settlement spreading far beyond the walls was frequent in the Near East, Asia, and Medieval Europe (Boone and Modarres 2006). In fact, Marco Polo reported that around the Mongol capital that would eventually become Beijing, "There is a suburb outside each of the gates, which are 12 in number, and these suburbs are so great that they contain more people than the city itself" (reported in Smith 2010).

A different type of sprawl characterized the layout of other ancient cities where residences were interspersed among agricultural plots in an extensive low-density continuum surrounding central institutional buildings and monuments. Scholars have identified this settlement structure among the cities of the Khmer of early medieval Cambodia, the classic Maya of Central America, and some precolonial African societies (Evans et al. 2007; Scarborough et al. 2012; Simon 2008). The capital city of the Khmer, Angkor, is well known for its central temples and massive hydraulic works, but it was supported by a vast sprawl of residences, farm plots, local ponds, and an infrastructure that tied together roughly 1,000 km^2 of low density urbanism (Evans et al. 2007). Low density urbanism also characterized many of the major Mayan cities, such as Tikal in Guatemala and Caracol in Belize, where major constructions of temples, pyramids and palaces in a central location were surrounded by a vast spread of housing complexes, agricultural plots, and an infrastructure of roads, causeways, and reservoirs tying them together (Scarborough et al. 2012). In both of these cases, agriculture within the broadly defined urban boundaries provided a major share of the city's subsistence; this highlights the ancient roots of the modern revival of urban agriculture (Barthel and Isendahl 2012).

Examining events and processes in the past often will provide useful insights into the origin of driving forces that impact cities today. However, the productive relationships that underlie the growth and success of cities may at the same time lead to relationships that are maladaptive, creating increased long term risks. For example, the concept of private property emerged to replace weak sense of ownership, lack of ownership, and/or the concept of community ownership. Farmers were increasingly able both to produce more food than their family required and they found ways to store this surplus for trade or for guarding against future bad harvests. However, one could only eat so much and a variety of factors limited the amount of food that could be effectively stored, including the ability of landlords and elites to appropriate some of the surplus through taxes. Hence the stimulus to produce a surplus remained limited in most farming villages. What changed this relationship, and is key to the growth of urban society, is the ability to transform locally produced surplus food into enduring prestige items associated with elevated status. This could only take place under a new social order that acknowledged classes with differential wealth, access to productive resources, power, and status. The promulgation of such a social order required an ideology (through religion, myth, constructed history, and/or law) that legitimized the existence of elite classes and the precious goods that helped to identify them. Of significant importance was that along with the evolution of private property, surplus production, elite goods, and hierarchical class society, the inheritance for membership in these classes and ownership of precious goods became more often defined by family and clan rather than merit. Strength, agility, and intelligence certainly were important, but which family, clan, and class one was born into set the limits on one's future potential in the age of early cities; to some extent, these constraints continue to operate today (Adams 1966; Prahalad 2005; Scott 1998).

Organizing society into hierarchically stratified classes became widespread as urbanization proceeded; this stratification continues to characterize most regions of the world up to the present day. This administrative framework and the widely accepted ideology that legitimize it became effective means of organizing large groups of people and large-scale productive activities. Territorially-based authority also emerged largely through successful military action and a monopoly on the use of coercive force. This secular authority also needed a source of legitimization, which often manifested in the form of constructed histories, law codes, and institutions of management and enforcement. Not surprisingly, in Western, Middle Eastern and some Asian societies, religious- and secular-based authorities interacted closely and often have been unified into a single entity or a closely cooperating team. Hence, in the newly emergent urban society of Mesopotamia—and later elsewhere across the globe—people could produce more, larger numbers of people could live in a single community and be marshaled as a labor force, sacred orders were established and widely accepted that legitimized the social order and explained appropriate behavior, and security was provided through a monopoly on the use of force and formal systems of laws. This new social and governing order was often reaffirmed through the construction of massive monuments, the performance of complex rituals, and expression through large-sized representational art. The concentration of people, stored supplies, and elite goods led to early cities being targets for raiding and organized military activity; this in turn led to further investment in defense walls and armies to defend cities. This cycle of concentration of wealth leading to military aggression, leading to investment in armies for defense and offense purposes is a cycle that dominates all of human history and can be seen operating today at many levels (Adams 1966; Scott 1998).

2.1.3 Disconnecting the Urban from the Rural: Alienation of Food Production from the Carrying Capacity of Land

Although there is great variation between different urban histories, large numbers of people aggregating into cities generally allowed for specialization of labor and other efficiencies of scale. This often generated the outcome that a large proportion of urban people were no longer self-sufficient in food production and hence, a greater proportion of people elsewhere in rural areas were be responsible for growing food for themselves, for the people in the city, and enough to monetarily offset the cost of transport and distribution. This put a tremendous burden on rural farming communities to produce much more than they would if solely working to supply enough for themselves. As the societal roles of the urban and rural populations grew increasingly different and complex, the objectives and understandings of these populations changed as well. Farmers experienced a shift away from traditional practices of the earlier village-farming era, in which they would have more intimately understood the landscape and productive systems and would have been inclined toward conservation practices wherein they balanced extractive activities with the regenerative capabilities of the land. The urban elite also experienced a shift away from traditional subsistence practices, and began to focus on the net produce they were able to extract from the countryside (or urban industries) and insisted on maximum production with little knowledge of, or concern for, the potential deleterious effects on the rural landscape (Jacobsen and Adams 1958; Redman 1999). However, the disregard for local dynamics of ecological integrity was not simply the product of urban demand; rural land owners, and national and transnational agricultural businesses were also instrumental in the alienation of food production from the carrying capacity of land. The rise of population that the enhanced production of food facilitated was not accompanied by innovation in trans-locational governance or in governance regimes that integrated cities and their hinterlands. In an ideal hierarchical society, even though decision-making authority would be concentrated at the top, one could assume that knowledge would travel up the hierarchy, and that informed decisions and concern would be displayed by decisions that traveled down the hierarchy. This was, however, seldom the case, and rather the dominant pattern was of maximizing short-term returns with little concern for long-term consequences. In many instances, archaeological evidence attests to the intense environmental degradation in the regions around ancient cities, and one can see the impact of urban demand on the rural countryside continuing today (Diamond 2005; Redman 1999) (Fig. 2.2). In Chaps. 22 and 26, we highlight the impact of the rising urban demand for food that is resulting in a competition for agricultural land; this competition is a global trend in land use that is largely unregulated.

Other outcomes of an increased urban efficiency create challenges of their own. Many of the world's devastating contagious diseases were virtually non-existent until the growth of dense urban populations. The spread of the plague, small pox, measles, cholera, and many other diseases can be traced to a combination of humans' close association with domestic animals and living in large, dense populations. Cities were the centers of people, economic activity, and the arts, but until public health innovations of the twentieth century, cities were also the centers of disease, many of them fatal. Urban agglomerations that are now better connected to each other through air transport continue to pose major health and biodiversity risks, necessitating a rethink of the global response to urban plant- and animal-disease outbreaks. The positive aspects of large urban populations described above also created new challenges that were unknown when the largest communities were several hundred people or less. The simple issue of knowing who everyone is and how to act toward them can no longer be easily handled when a community grows beyond 500 people. Similarly, tranquility and security break down as the population aggregation grows larger; this prompts the introduction of formal, less personal solutions to human interactions and security. Similar challenges that grow with scale of the community, such as transport of people and goods, sanitation, and supply of water and food need to be addressed by formal institutions beyond the extended household. While these more public governance regimes may confer social and economic liberties on some urbanites, especially women, the shift away from the (often male) head of household and community leaders and toward appointed or elected city authority does increase individuals' dependency on the central authority for basic needs, including personal health and ecosystem integrity.

2.1.4 Lessons for the Future

Several lessons stand out from this brief review of early urbanism. First, humans are amazingly successful at self-organizing to promote their survival in the face of any environment challenge, but there are unanticipated costs to many of these solutions and continued implications for future societies. People manage their social-ecological systems according to their often-limited perceptions of the opportunities and risks, and how they value the alternatives. However, this valuation process may appear very different to people in different social positions and the true "costs" of some alternatives are not recognized at the time; ultimately, they may even threaten the society's very survival. In general, people respond to problems and opportunities by transforming biota, landscapes, and the built environment so that their immediate net yield is increased and perceived risks are reduced even though native biota and ecological systems may be degraded. Humans also create new values and institutions for collective action to control and optimize the shifting capacities and risks presented by their evolving environments. These collective responses are seen most obviously (but by no means exclusively) in the nineteenth and twentieth century rise of corporations, nation states, and local governments. These structures of power represent the product of struggle, and not all the impacts of individual or collective decisions provide pathways toward a more sustainable and desirable existence.

2.2 Urbanization, Ecosystems and Ecosystem Services

2.2.1 Urban Food Production

Even though ecosystems have been overlooked in urban scholarship (Sinclair et al. 2010), it is evident how significant urban green and blue spaces have been historically in producing a range of provisioning ecosystem services, such as agricultural produce, fish, game, water and fuel (Fraser and Rimas 2010; Redman 1999). In contemporary cities, approximately 200 million urban residents produce food for the urban market, and provide 15–20 % of the world's food (Armar-Klemesu 2000). For example, in Dar es Salaam, 90 % of all vegetables consumed originate from urban and peri-urban agriculture; the same is true of 60 % of all vegetables in Dakar, and in Hanoi, 58 % of the rice consumed is produced within the jurisdiction of the city (Moustier 2007; Lee-Smith 2010; Lerner and Eakin 2011). Such figures are much lower in Southern African cities (Simon 2013; Battersby 2007), and low but on the increase in some European and North American cities (Simon 2008) (for three historical examples of urban food production see Box 2.1).

2.2.2 Urban Green Spaces

Not all of the green space in pre-industrial urban landscapes, however, was used to produce food. For example, open spaces have often been used as religious sites and as cemeteries. In many cities, particularly European, pleasure parks and pleasure gardens for purely recreational uses have also been present in cities since millennia, but these have mainly been the privilege of emperors, kings and other urban elites. In Stockholm, for instance, ordinary citizens were not allowed to enter such parks and gardens until the mid-1700s (Barthel et al. 2005). The main social drivers that led to a shift toward public use of such green spaces were the rapid urbanization during the industrial revolution, in combination with emerging social values inspired by the Romantic Movement and the French Revolution (Barthel et al. 2005). However, clear delineations between urban and rural areas and use of urban green spaces for purely recreational purposes did not emerge until the nineteenth and twentieth centuries, and were reinforced by the development of a globalized economy, the fossil fuel energy regime, and technological innovations such as the steam engine and the railway (McNeill 2000; Barthel and Isendahl 2012; Barthel et al. 2013). Across Swedish cities, urban food production was ubiquitous until the development of the railway network, and the towns were in fact producing 50 % of their food consumption within their boundaries, and some were producing much more. For instance, in the mid-1700s, Uppsala produced more food than the city dwellers themselves consumed and the surplus was exported outside the city (Björklund 2009).

However, the mental models that developed among urban theorists in the beginning of the 1900s soon excluded the rural aspects of life in the city. One example is the Chicago School of urban sociology. Based in ecological theory (cf. Clements 1916) and using Chicago as a case study, the Chicago School of urban sociology emerged in the 1920s and 1930s to establish a modernist understanding of urban life as separate from rural life (McDonnell 2011). The idea of cities as separate entities essentially detached from their broader life-support systems (Wirth 1938) was strongly linked to major innovations in transportation technology as Chicago became an important hub in the U.S. railroad network in the 1850s, and food transportation over great distances became possible. Establishment as a railroad hub enabled Chicago to grow rapidly from a few thousand inhabitants in the 1850s to over two million in the early 1920s. Industrial-era technological innovation, cheap and efficient travel, and economic growth (opening new markets, speeding up production cycles, and reducing the turnover time of capital) catered for the first wave of space-time compression[1] (Harvey 1990). Hence, the modernist ideology underpinning the emergence of urban planning during the early decades of the 1900s distinctly separated local agricultures and other rural dimensions as obsolete in futuristic and normative understandings of the city as an autonomous social system (Barthel and Isendahl 2012).

2.2.3 Historical and Cultural Dimensions of Urban Biodiversity

Urban green infrastructures, often rich in species, are, in most parts of the world, remnants of domesticated landscapes with a long-term history of land use. There are exceptions to this in regions that do not have a long-term history of agriculture, for example in parts of Oceania, South Africa and North America. It is in the cultural landscapes that biodiversity and ecosystem services are produced, and over which growing cities expand (James et al. 2009). Habitat legacies include long-lived species, meadows, gardens, ponds, agroforestry areas, *satoyama* systems, hedges, and

[1] Socio-economical processes that accelerate the pace of time and reduce the significance of distance.

Box 2.1 Three Historic Examples of Urban Food Production and Emergence of Biodiversity-Rich Urban Landscapes

Ancient Mayan Cities. Cities in Meso-America traded a variety of food commodities both short- and long-distance (Dunning 2004; Isendahl 2006), but seasonally impassable rivers and energetically costly overland transports put a relatively high cost on trade and inhibited bulk-staple exchange (Isendahl 2006, 2012). Hence, much of the food consumed by the urban Maya Indians came from proximate lands (Isendahl 2006, 2012). For instance, large sectors of fertile soils inside the urban landscape were devoid of settlement constructions, but were used as city infields (Isendahl 2012). The management of these infields in Mayan cities was markedly different from the larger and state-owned farmstead gardens (Barthel and Isendahl 2012; Isendahl 2012), which were put under tremendous pressure when competition between city-states intensified, a condition which at least partly contributed to the collapse of Mayan cities in the tenth century AD (Tainter 2011). The infields were used as household farmstead gardens, which concentrated agricultural knowledge and stewardship of the agricultural biodiversity that was the ultimate survival strategy for the populace (Ford and Emery 2008). Owing to residential proximity it was most carefully tended, and most carefully fertilized by the organic waste concentrated by city dwellers, and was used for plant breeding, experimentation, and for seed storage (Ford and Nigh 2009). The household farmstead garden held the key to a resilient flow of urban ecosystem services and provided food security for the population (Barthel and Isendahl 2012). Remnant urban ecosystems and the rich levels of biodiversity found in the urban Yucatan today are hence viewed to be the products of a millennia-long co-evolution in cultural landscapes (Ford and Emery 2008; Ford and Nigh 2009).

Constantinople. Different in many respects from Mayan cities, Constantinople, the capital of the Roman cum Byzantine Empire from the fourth century AD until 1453, got its main source of staples of grain from the Nile Valley and was brought in by trading vessels averaging 40–50 tons each in capacity (Balicka-Witakowska 2010). Although these supply lines were subjected to the difficult winds of the eastern Mediterranean and the fluctuations of Nile river dynamics, the most severe threats to food security were the sieges and blockades that distinctly cut food- and water-supply lines; these disruptions occurred on average every 65 years during the last 1,000 years (Barthel et al. 2010b; Barthel and Isendahl 2012). The most difficult blockade on the food supply lines, at the end of the fourteenth century AD, lasted an astonishing 8 years, but it did not succeed in starving out the urban population (Ljungqvist et al. 2010). To accommodate growth and respond to food and water insecurities during such sieges, an additional wall (the Theodosian Wall) was erected 1.5 km westwards of and about a century after the first

(continued)

Box 2.1 (continued)

(the Constantine Wall). Major water cisterns and a 3 km^2 green common used for cultivation and pasture area were allocated between the old and new walls. This area, in addition to the 2-km-wide buffer zone of farm fields immediately outside the Theodosian wall, resulted in a total of 15 km^2 of agricultural lands in direct proximity to the urban core; these lands were used as main sources of food production during periods of siege. Even in a city exhibiting a relatively compact urban spatial form, food production was a pertinent feature until the beginning of the fossil fuel energy-regime (Björklund 2009; Barthel and Isendahl 2012). The rich levels of biodiversity found in remnant seminatural patches of the contemporary Istanbul region (see Chap. 16, Local Assessment of Istanbul) is hence a product of co-evolution between cultural practices and the bio-physical environment.

Stockholm. The newly protected and biodiversity-rich National Urban Park of Stockholm (protected by law in 1995) has a millennia-long history of food production (Barthel et al. 2005). The ecosystems here are relatively rich in terms of biodiversity; they are remnants of land used for production of food, fiber, fuel, feed and building material. More than 1,000 Lepidopteran species, 1,200 Coleopteran species, and 250 bird species have been observed here. Furthermore, there are more than 60 IUCN Red-Listed insect species, of which 29 are threatened and 27 are vulnerable. In addition, more than 20 species of Red-Listed vascular plants, mammals, amphibians, reptiles, and fish can be found in a landscape that was, until the 1700s, used for agriculture and later as hunting ground, and the legacies of which can be seen in the present-day mosaic in the landscape (Barthel et al. 2005) (see further Chap. 17, Local Assessment of Stockholm).

orchards (Ford and Nigh 2009; Duraiappah et al. 2012) (see Chap. 10). The combination of such legacies in cultural landscapes can be powerful generators of biodiversity if environmentally benign and historically informed management practices are applied (Andersson et al. 2007; Galuzzi et al. 2010) (Chap. 10). Stewardship of ecosystem services in metropolitan landscapes is thus dependent on the continuation of historically informed management practices. Current biodiversity and ecosystem services are conditioned by history, regional context and continuity (Foster et al. 2003). Continuity is carried by memory, as in memory of past environmental responses carried in the genes of organisms, in community compositions and in habitat legacies, as well as in people carrying social memory such as oral tradition, rituals, institutions and tools that guide management practices (Barthel et al. 2010a; Barthel and Isendahl 2012). Much of this memory has been lost, and there is a need to regain and produce new and relevant knowledge for management of urban social-ecological systems (see Chaps. 27 and 30).

2.3 Natural Capital: Reintroducing Ecology into Urban Economy and Governance

During the previously described long stretch of history, societies and economies were not growing very quickly (Fig. 2.1). However, since the beginning of the industrial revolution, and especially after the start of the "great acceleration" following the end of WWII, there has been rapid economic expansion coupled with rapid urban growth—all driven by rapid expansion of fossil fuel use, especially oil (Costanza et al. 2007b). Indeed, one of the hallmarks of contemporary urbanization is that urban areas are growing *faster* and *larger* than they did in the past as well in new geographic locations (Seto et al. 2012b) (Chap. 21). Current mainstream concepts and models of the economy were developed in this period of rapid expansion as if the world we lived in had unlimited capacity for growth in the material economy. In this "empty world" context, *built capital*—the houses, roads, and factories–things that are concentrated in cities—was the limiting factor to improving human well-being. *Natural capital*—our ecological life support system—and *social capital*—our myriad relationships with each other—were viewed to be abundant (Costanza et al. 1997a). It made sense in this context not to worry too much about environmental and social "externalities" – effects that occurred outside the market—since they could be assumed to be relatively small and ultimately solvable. Instead, the focus was on the growth of the market economy, as measured by Gross Domestic Product (GDP), as a primary means to improve human welfare. The dominant thinking categorized the economy as only marketed goods and services and the goal of society was simply increasing the amount of these goods and services produced and consumed (Costanza et al. 1997a).

We now live in an interconnected global system that is relatively full of humans and their artifacts (Fig. 2.1) in what some are even calling a new geologic era—the "Anthropocene" (Crutzen 2002; Steffen et al. 2011)—and have shifted into a human-dominated planet and into a new full-world context (Daly 2005). Some have also argued that we have already moved beyond the "Anthropocene" into the new urban era (Seto et al. 2010; Ljungqvist et al. 2010). Now we have to think differently about the relationship between humans and the rest of nature. If we seek "improved human well-being and social equity, while significantly reducing environmental risks and ecological scarcities," as the UN has recently proclaimed as the primary global goal (UNEP 2011), we will need a new vision of the economy and of cities and their relationship to the rest of the world that is better adapted to the new conditions we face. We will require a vision of the economy and urbanization that reintroduces the ecology of the urban. Material consumption and GDP are merely means to that end, not ends in themselves, and we need to better understand what really does contribute to sustainable human well-being (SHW), and recognize the substantial contributions of natural and social capital, which are now the limiting factors to improving SHW in many countries. We must be able to distinguish between real poverty in terms of low SHW and merely low monetary income.

To achieve sustainability, we must incorporate natural capital (and the ecosystem goods and services that it provides) into our economic and social accounting and our systems of social choice. Ecosystem services are defined as, "the direct and indirect benefits people obtain from ecosystems" (Costanza et al. 1997b; Millennium Ecosystem Assessment 2005) (Chap. 11). These include provisioning services such as food, water and medicinal plants; regulating services such as air quality regulation, water purification, regulation of floods, drought, and disease; supporting services such as soil formation and nutrient cycling; and cultural services such as recreational, scientific and spiritual benefits (Costanza et al. 1997b; Daily 1997; de Groot et al. 2002, 2010). People in cities benefit from ecosystem services at a number of spatial and temporal scales (Chap. 11). Urban residents could not survive without these life support services and it is therefore necessary to take a comprehensive, integrated, multi-scale approach to what constitutes urban infrastructure and assets. It is not just the built capital of cities that we need to consider. It is the full spectrum of assets including social and natural capital at local, regional, national, and global scales.

We can expect many ecosystem services to go almost unnoticed by the vast majority of people, especially when they are public, non-excludable services that never enter the private, excludable market. Conventional economic valuation presumes that people have well-formed preferences and enough information about trade-offs that they can adequately judge their "willingness-to-pay." Since these assumptions do not hold for many ecosystem services (Norton et al. 1998) we must either:

1. inform people's preferences by demonstrating the underlying dynamics of the ecosystems in question and their connection to human well-being;
2. allow groups to discuss the issues and "construct" their preferences within a framework that conveys information about the connections; or
3. reject current models of macro-economy in urban governance and use other techniques that do not rely directly on preferences to estimate the contribution of ecosystem services to human well-being, for example, through the use of scientific studies and computer models that can trace the complex linkages between ecosystem functioning and human well-being.

However, one must not confuse expressing values in monetary units with treating ecosystem services as tradable private commodities. Most ecosystem services are public goods that should not be privatized or traded (cf. Daniel et al. 2012). This does not mean they should not be valued (see Chap. 11). But because natural capital is a public good, it is not handled well by existing markets, and special methods must be used to estimate its value and new institutions are needed to manage it (Chaps. 11 and 27).

2.4 Conclusion

As we have argued in this chapter, a social-ecological dimension of urbanization has been neglected, resulting in a conceptual separation of the urban and the rural, and thus shaping our perceptions of the urbanization process itself and our policies and actions (cf. McGranahan et al. 2005; Grimm et al. 2008; Pickett et al. 2011; McDonald and Marcotullio 2011; Folke et al. 2011; Anderson and Elmqvist 2012; Wu 2013). Urbanization affects ecosystems both within and outside of urban areas, and as stated in Chaps. 1 and 21, on a global scale urban land expansion will be much more rapid than urban population growth—in some places resulting in large, complex, urbanizing regions comprised of aggregations of interconnected cities and interspersed rural landscapes with multiple impacts, dependence and feedbacks (Seto et al. 2012a; Seitzinger et al. 2012). Recently, new and promising conceptual frameworks based on analyses of urban land teleconnections have been proposed to further explore the multiple dependence and impacts of cities on distant places well beyond the urban hinterland (Seto et al. 2012a); this holds promise to make many invisible social-ecological feedbacks and connections visible (Chap. 33). Many of the following chapters, including Chaps. 3, 10, 11, 22, 26, and 27 will further explore this missing link—the urban social-ecological connections and their governance implications.

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Chapter 3
Urbanization and Global Trends in Biodiversity and Ecosystem Services

Robert I. McDonald, Peter J. Marcotullio, and Burak Güneralp

Abstract This chapter introduces patterns of urbanization, biodiversity, and ecosystem services at the global scale. Underpinning the goals of the chapter is the notion that cities are inextricably linked to the biophysical world, although these linkages are increasingly difficult to clearly identify. The chapter starts by introducing the idea that cities both impact and depend upon the biophysical environment. We go on to discuss how urbanization is both the cause of societal or environmental problems and the solution to many problems, depending on the time-scale and scope of the analysis. Finally, we provide a global overview of cities' relationships with two key facets of the environment: biodiversity and freshwater ecosystem services.

3.1 Cities Both Impact and Depend on the Environment

As highlighted in Chap. 1, city growth and the urbanization process are linked with biophysical and ecological processes. The totality of these linkages are often too daunting to track down; therefore, researchers tend to adopt one of two primary

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modes of analysis to dissect the interaction between cities and the environment as exemplified in the chapters of this volume.

One mode of analysis of urban/environment interactions is to focus on the impact of urban areas upon biodiversity or ecosystem services. These impacts can occur over a range of spatial scales (McDonald et al. 2009). At a very local scale, the pattern of urban development determines how natural habitat is fragmented, which affects how native biodiversity is impacted and where invasive species become established, as discussed in Chap. 10. Chapters 11 and 12 discuss specific factors affecting urban form and their implications for biodiversity and ecosystem services. For a more complete discussion of policymakers' attitudes toward urbanization and policies that can decrease environmental impact, see Chap. 27.

A second mode of analysis of urban/environment interactions is to study the dependence of urbanites on biodiversity and ecosystem services. Dependencies can occur over a range of scales, just like impacts. To be a true ecosystem service, a desirable ecosystem process has to occur near consumers of that service (McDonald 2009). The degree to which proximity is essential—the transportability of an ecosystem service—varies from service to service. Urban street trees, for instance, provide shade to urbanites over a scale of tens of meters. At a watershed scale, many cities depend on natural habitat to provide an adequate supply of clean water. At a global level, urbanites depend on the climate regulation services supplied by ecosystems. Chapter 11 discusses many kinds of urban dependencies in detail. Chapter 25 looks at how cities depend on a stable climate and how climate change may affect them. Chapter 31 discusses how to restore ecosystem services and biodiversity when ecosystems are degraded.

In this chapter, we first focus on how global patterns of urban growth intersect with global patterns of biodiversity, which is often seen as the foundation for ecosystem service provision. We then illustrate the dependence and impact of cities on ecosystem services at the global level in the context of one of the most vital: freshwater ecosystem services.

3.2 Urbanization as a Problem and a Solution

Global urbanization has been an uneven process, both temporally and geographically (Satterthwaite 2007). The increase in the global urban population began slowly. In 1800, around 3 % of humanity lived in cities, with an estimated 1.7 % of global population in cities of 100,000 or more and 2.4 % of global population in cities of 20,000 or more. As late as 1900, the share of the world's population living within cities of these sizes remained less than 10 % (Davis 1955). By 1950, however, estimates suggest that approximately 729 million people worldwide lived in all cities; this number corresponded to 29 % of the global population (United Nations 2010b). Subsequently global urbanization increased rapidly. By 1960 there were approximately 998 million in the world's cities, by 1985 there were 1.98 billion, and by 2010 there were 3.49 billion. The period of the most rapid annual increases globally were experienced between 1950 and 1965, when rates exceeded 3.0 %. By 2010, the annual growth rate for global urban population had fallen to 1.85 %. This amounts to adding 67.5 million people to the urban population each year. The UN (2010a) suggests that the numbers of people moving to cities annually will continue to increase until around 2030, when more than 72 million people are predicted to be added to cities annually. Thereafter the annual additions are expected to decline (for further discussion on population projections, see Box 21.1 in Chap. 21).

In terms of geographical variability, urbanization has reached high levels in the developed world, both of which largely manifest in the temperate zone. Generally, cities in these Northern areas are now growing more slowly than those of the South and some are even contracting in terms of population (Chap. 12). At the same time, urbanization is increasing in the developing world, much of which is located in the tropics and sub-tropics. In these locations, cities are absorbing large numbers of people.

The advance of urbanization, particularly after the 1950s, has coincided with global environmental degradation, increasing consumption of natural resources, habitat loss and ecosystem change (McNeill 2000). It is therefore not surprising that analysts often depict cities as the source of many problems. Lester Brown (2001, pp. 188–190), for example, 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." The ecological footprint of a city, the area required to supply its citizens with resources and services from the environment, is much larger than the area of the city itself (Wackernagel and Rees 1996).

This viewpoint of cities as a source of environmental problems, however, often rests on a relatively simple scope of analysis. A simple equation for calculating such an impact is the so-called I = PAT equation (cf., Dietz and Rosa 1997), where Impact (e.g., tons of greenhouse gases emitted) equals the number of People times the Affluence (e.g., energy consumption per capita) times the Technology (e.g., tons of greenhouse gases emitted per unit energy). If total impact from an urban area is the scope of analysis, then in most cases larger cities will cause a larger impact on the environment, for the simple reason that the population is larger. By this logic, a city of zero population size would have zero environmental impact.

However, the process of urbanization also influences both the Affluence and Technology terms in the I = PAT equation, in sometimes complex ways. Incomes tend to be greater in cities than in rural areas, and greater in bigger cities than in smaller cities (Bettencourt et al. 2007), which can sometimes increase resource consumption. However, there are often efficiencies that are gained with dense settlement. Studies in the United States, for example, have pointed out that residents of cities consume less energy per-capita and therefore generate less greenhouse gas emissions per-capita (Brown et al. 2008). Similarly, urban residents in the United States eat less beef and pork (Davis and Lin 2005a, b) than their rural counterparts. In the developing world, in contrast, those in cities consume more meat than their rural counterparts (cf., Dhakal 2009), which appears to be primarily due to the increase in income in urban households rather than changes in dietary preferences associated with living in a city (Stage 2009).

It is also, arguably, inappropriate to simply talk about the environmental impact of a city relative to some hypothetical case where the city simply disappeared. A more sophisticated analysis might specify a counterfactual scenario: what would have happened to the environment without the urbanization (McDonald and Marcotullio 2011)? These counterfactual scenarios are very difficult to construct. Without migration to cities, there might be less environmental impact from cities, but perhaps more impacts in the countryside. Economists have long suggested that urbanization has a strong positive correlation with economic activity (Williamson 1965; Annez and Buckley 2009), although rapidly growing urban areas can have offsetting negative effects through crowding, environmental degradation and by overwhelming city administrations' capacities (cf., Bloom et al. 2007; Bai et al. 2012). Certainly, without urbanization, economic development will potentially be limited, and since rural fertility rates are generally higher, a larger total population may result than in the urbanization scenario.

As discussed in Chap. 2, urbanization is a multifaceted process, and it is very difficult to specify what would have happened to the environment in a society if urbanization did not occur. Urbanization is promoted by numerous factors, including: increased ease of communications and transport, economies of scale and agglomeration economies (Bai et al. 2012), increased personal contact among workers and entrepreneurs, and efficiency gains from the high population density in cities (for a review, see Montgomery et al. 2003). As people move to cities they leave the agricultural sector for employment in industry and services, thus substantially changing the economies of nations as they urbanize. Urbanization is also associated 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). Thus, from the perspective of the economic development and human well-being of a nation, urbanization is often an integral part of the solution.

The pragmatic truth is that a counterfactual scenario without urbanization is unlikely to ever occur. All developing economies urbanize and there are no examples of nations with high economic development that have not experienced urbanization (Fig. 3.1). Moreover, policy attempts to limit urbanization have not only had limited effects on rates of urban growth and they have had disproportionately negative impacts on large portions of societies; typically the poor. 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 % in 1996 to 73 % in 2005 (cf., Bai 2008). While they have had significant negative impact on the lives of rural-to-urban migrants, these policies have had little long-term effect on urbanization and an arguable negative impact on economic growth (Bloom et al. 2007; Bai et al. 2012).

In short, if demographic forecasts are correct, a large amount of urban growth is coming as poorer countries urbanize (for forecasts, see Chap. 21). Therefore, it is necessary to examine the types of biophysical environmental impacts expected from urbanization without forgetting that the process of economic development and urbanization can also help the world find solutions to poverty and environmental degradation.

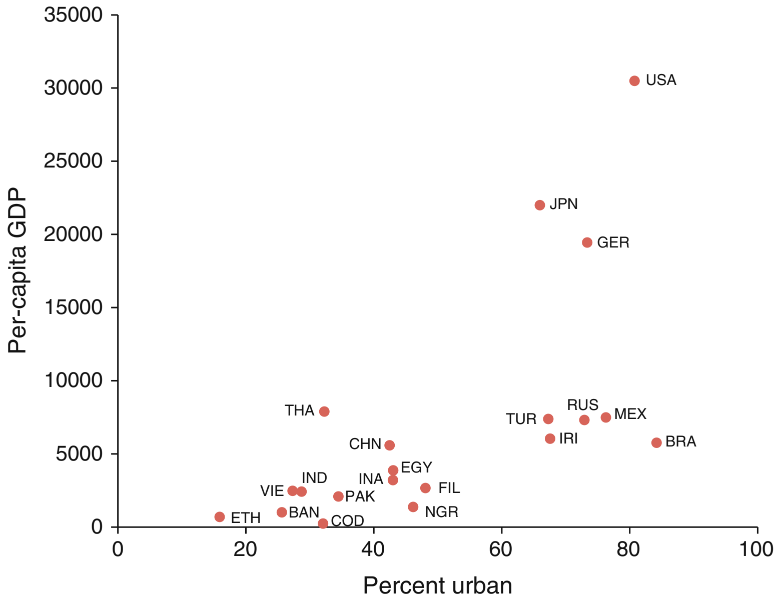

Fig. 3.1 Correlation between percent urban and per-capita GDP for the 20 countries with the greatest population in 2005. Per-capita GDP is taken from (Maddison 2001), and is shown in 1990 International Geary-Khamis dollars, a method of correction for different purchasing powers over time and space. Percent urban is taken from the World Urbanization Prospects database (UNPD 2009). Countries are abbreviated as: *BAN* (Bangladesh), *BRA* (Brazil), *CHN* (China), *COD* (Congo Kinshasa), *EGY* (Egypt), *ETH* (Ethiopia), *FIL* (Philippines), *GER* (Germany), *INA* (Indonesia), *IND* (India), *IRI* (Iran), *JPN* (Japan), *MEX* (Mexico), *NGR* (Nigeria), *PAK* (Pakistan), *RUS* (Russian Federation), *THA* (Thailand), *TUR* (Turkey), *USA* (United States), *VIE* (Vietnam)

3.3 Global Urbanization and Biodiversity

Biological diversity is an essential component of many invaluable ecosystem services for human material welfare and livelihoods. For example, many components of people's homes are provided, regulated or supported by biodiversity, including food, the wood in the building, fresh water from taps and fuel in stoves. Nitrogen fixation is important for biological productivity, and only a few plants such as legumes can perform this service. Preserved forests close to coffee-plant flowers, provide reliable sources of pollinators, which have been estimated to improve coffee yields by 20 % (Melillo and Sala 2008). Biodiversity contributes to human security, resiliency, health and freedom of choices and actions (Millennium Ecosystem Assessment 2005). Moreover, biodiversity preservation is a goal in itself, as articulated in the Convention on Biological Diversity and many national-level laws (e.g., the Endangered Species Act in the United States).

Despite these important contributions to society, biodiversity is declining. Researchers have identified a sixth great extinction event promoted by anthropogenic activities (Wilson 2005). Human actions are fundamentally and irreversibly changing the diversity of life on the planet (Millennium Ecosystem Assessment 2005). Rates of extinction continue to increase and the number of species threatened continue to grow (Pimm et al. 1995).

In this section we examine the global impact of urbanization on biodiversity. We examine this relationship through a review of the direct impact of urban growth as well as through an examination of the indirect impacts of urbanization.

3.3.1 The Global Distribution of Biological Diversity

Biodiversity can be examined a number of different ways. In this overview we review the literature on urbanization's impact on species richness and endemism. While species richness and endemism vary unevenly across the Earth's surface, a number of broad trends have been observed.

Species richness is generally higher in high productivity sites like tropical rain forests and lower in low productivity sites like arctic tundra, for unclear reasons (Willig et al. 2003). The pattern of distribution is called the latitudinal geographic gradient because the highest levels of biodiversity are found near the equator and they drop off as one moves towards the poles (Turner and Hawkins 2004). This pattern holds true for major taxa (classes, orders and families) for microbes, plants and animals in both terrestrial and aquatic systems. The latitudinal gradient is superimposed on a number of other gradients including distance to coast, position within a peninsula, and topographic position (Lomolino et al. 2010).

Species endemism is the number of species unique to one location and is a major concern to conservationists. Examples of endemic species include the Devil's Hole pupfish (*Cyprinodon diabolis*) from the United States, Australia's koala (*Phascolarctos cinereus*) and many different species of cichlid fish found in Lakes Victoria, Tanganyika and Malawi. Endemism is distributed very differently from species richness. While species richness is low on isolated islands, endemism is high in proportional terms, as the geographic isolation of biota leads to speciation that fills empty niches. Coastal areas are also places with a high degree of marine and terrestrial endemism because of the high habitat diversity (Dirzo and Raven 2003).

3.3.2 Direct Impact of Urbanization on Biodiversity

Cities are concentrated along coastlines and some islands as well as major river systems, which also happen to be areas of high species richness and endemism. Ecologists have explained this pattern by examining the correlation between human population density and productivity (Luck 2007), while urban historians have focused on the importance of freshwater and marine trade routes for city formation.

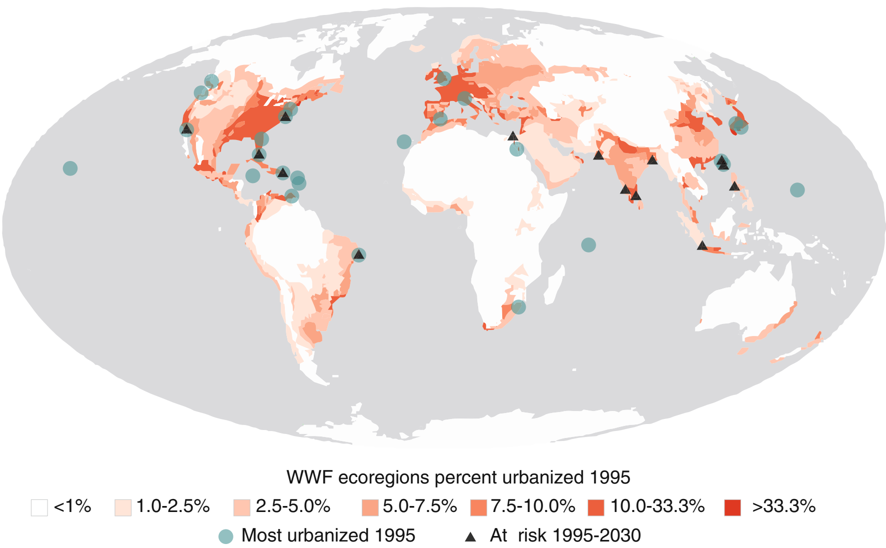

Fig. 3.2 Percentage of an ecoregion's area that was urban circa 1995; ecoregion boundaries follow those of the World Wildlife Fund (Olson et al. 2001). Ecoregions with more than one-third of their area urban in 1995 are marked. At-risk ecoregions, which will lose more than 5 % of their remaining undeveloped area by 2030, are also marked (Modified from McDonald et al. 2008, p. 1698. Published with kind permission of © Biological Conservation 2008. All Rights Reserved)

The most direct impact of cities on biodiversity is the change in land cover associated with urban growth. Urban growth is clearly a significant global driver of land-use conversion and deforestation. Urban areas occupy approximately 3 % of the Earth's land surface (McGranahan et al. 2006), although the actual number varies significantly depending on the definition of urban and the spatial grain of analysis (Schneider et al. 2009; Seto et al. 2010). For a discussion of the various definitions of urban, see Chap. 1.

The spatial correlation between urban growth and endemism means urban growth has already impacted biodiversity significantly (McDonald et al. 2008) analyzed the implications of urban areas *circa* 1995 for ecoregions (Olson et al. 2001), protected areas across the world (www.wdpa.org), and rare species (Ricketts et al. 2005). They found the effect of urban areas to be concentrated in certain localities (Fig. 3.2). The majority of terrestrial ecoregions (comprising 62 % of the Earth's land surface) are currently less than 1 % urbanized and will experience little change through 2030. However, around 10 % of terrestrial vertebrates are in ecoregions that are heavily impacted by urbanization, even though these ecoregions only represent 0.3 % of the Earth's land surface (Fig. 3.2). These ecoregions are concentrated along coasts and on islands, which are generally areas of high endemism (Ricketts et al. 2005). In addition, urban areas seem to have increased the threat to survival of certain vertebrate species, especially those having smaller ranges. Most of this threat is in middle and low-income countries, which raises questions about the institutional capacity to act against potential adverse effects of urban expansion on biodiversity.

Less than 1 % of all biodiversity hotspot areas (Myers et al. 2000; Mittermeier et al. 2004) were urbanized *circa* 2000 (Seto et al. 2012a). However, similar to the ecoregions there is large variation in urban land cover across the biodiversity hotspots with concentration of urban lands in certain hotspots. In particular, the Mediterranean Basin and the Atlantic Forest biodiversity hotspots had the most urban area *circa* 2000 (over 30,000 and 25,000 km^2, respectively). On the other hand, the California Floristic Province and Japan hotspots had the largest percentage of their total land urbanized (about 5 % each).

Around the year 2000, South America had the most urban land in biodiversity hotspots (about 46,000 km^2, nearly 60 % of all urban land in the region) among all regions (Güneralp and Seto 2013). Nearly all the urban land in Southeastern Asia (27,000 km^2) was located in biodiversity hotspots. Most of this urban land was distributed across two biodiversity hotspots: about 10,000 km^2 in the Indo-Burma hotspot that covers most of the mainland portion of the region, and about 13,000 km^2 in the Sundaland hotspot that includes most of the Malay Peninsula and the island of Java. Northern Africa had almost half of its total urban land in the Mediterranean hotspot, the only hotspot in the region. These patterns collectively reflect that biodiversity hotspots predominantly occupy coastal areas that are also places of concentration of urban land.

Globally, 32,000 km^2 of protected areas (PAs) were already urbanized *circa* 2000, corresponding to 5 % of global urban land (Fig. 3.3). In particular, in Europe, which has already largely urbanized and has an extensive PA network, almost 20,000 km^2 of PAs were already under urban land cover (about 10,000 and 9,500 km^2 in Eastern and Western Europe, respectively). This corresponds to 13 % of total urban extent in the continent *circa* 2000, 14 and 12 % in Eastern and Western Europe, respectively (Fig. 3.3). China and South America also had substantial amounts of urban land within their PAs with 4,500 and 2,800 km^2 in each country, respectively (i.e., 6 and 3.5 % of their respective urban lands).

Different impacts will materialize at varying distances from urban areas and ecological mechanisms often link protected areas to surrounding lands (Hansen and DeFries 2007). It is worth noting that some of these effects are positive such as recreational activities and logistical advantages provided by close proximity to ecosystem services provision areas within protected areas.

A great proportion of the world's terrestrial protected areas are also within 50 km of a city. Almost half of the case studies (47 %) in a meta-analysis on global urban expansion are found within 10 km of a terrestrial protected area (Seto et al. 2011). Moreover, the same study found that the average annual rate of urban land expansion of these cities from 1970 to 2000 is greater than 4.7 % and not statistically significantly different from growth rates of urban areas elsewhere. Thus, urban land expansion is as likely to take place near protected land as elsewhere, and proximity of an urban area to a protected area does not necessarily slow the rate of urban land conversion.

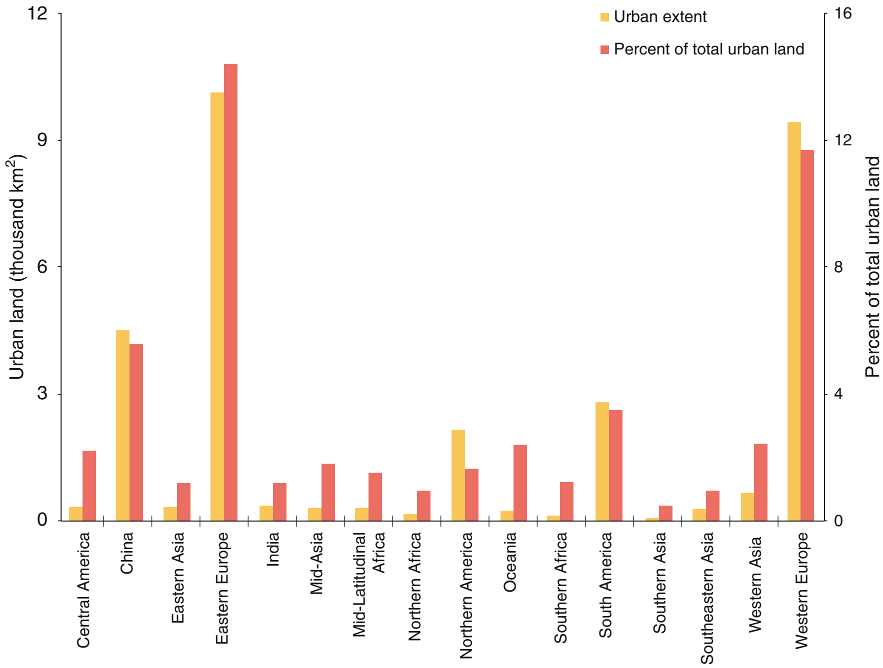

Fig. 3.3 Urban extent and percentage of total urban extent that fall in the IUCN-designated protected areas (PAs) by geographic region circa 2000 (Modified from Güneralp and Seto 2013, Figure S1, p. 3 of supplementary data. Published with kind permission of © Environmental Research Letters 2013. All Rights Reserved)

More than 100,00 km^2 of urban land (15 % of the global total) was within 10 km of a PA *circa* 2000 (Güneralp and Seto 2013). In North America, while there is little urban land located in PAs, the amount of urban land in close proximity to PAs is the largest among all regions. The other two regions that have a high percentage of their populations that are urban, Western Europe and Eastern Europe, also had large amounts of urban land within close proximity of their respective PAs (Fig. 3.4a). Overall, 4 and 11 out of the 16 regions had 50 % or more of their urban land within 25 and 50 km of PAs, respectively (Fig. 3.4b). On the other hand, in almost all regions except Eastern Asia and Western Europe, the percentage of lands that were urban within the 10, 25, and 50 km-wide zones around the PAs was well below 2 % *circa* 2000.

Information on land-use change due to urbanization is not available over long periods of time. However, it is instructive to look at how urban population in different habitat types has changed over time (Fig. 3.5). In 1950, the habitat type with the most urban dwellers was temperate broadleaf forests, followed by tropical moist forests and Mediterranean habitat. However, a more useful proxy measure of

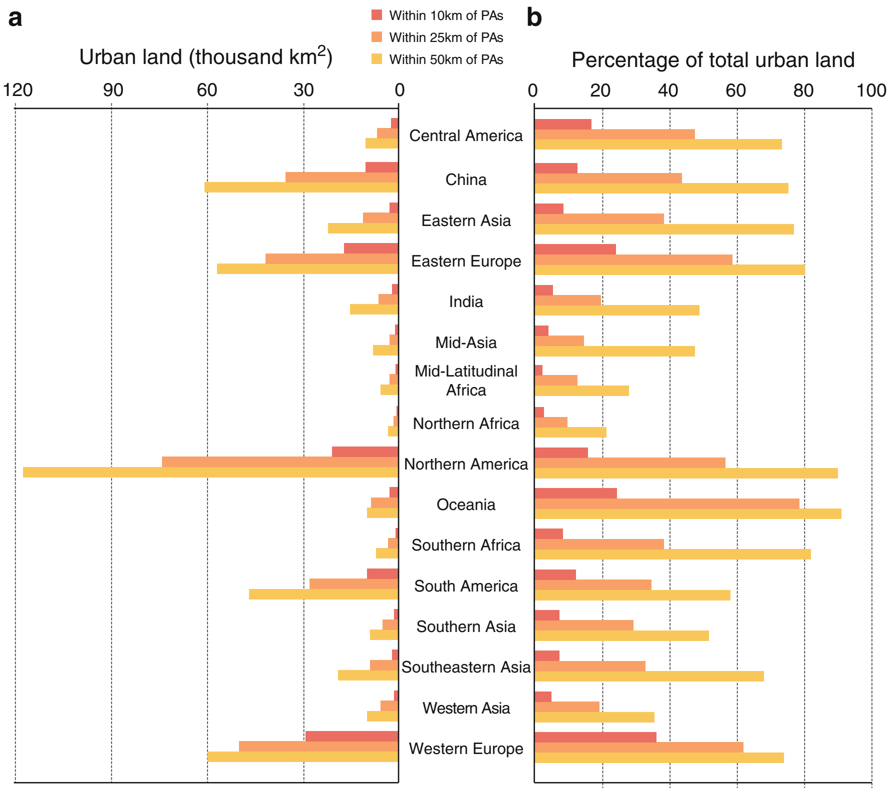

Fig. 3.4 (**a**) Urban extent and (**b**) percentage of total urban extent within a distance of, from *top* to *bottom*, 10, 25, and 50 km of PAs by geographic region circa 2000 (Modified from Güneralp and Seto 2013, Figure S2, p. 4 of supplementary data. Published with kind permission of © Environmental Research Letters 2013. All Rights Reserved)

biodiversity impact is the urban population density in a habitat type (i.e., urban population divided by the total area in a habitat type). Note that this proxy measure is much lower than the population density at which urban settlements occur, but it gives a rough sense of how many urban people are crowded into this habitat type. By this proxy measure, the Mediterranean, mangrove, and temperate broadleaf forest habitat types all have high urban population density per habitat area and hence likely have had significant impacts on biodiversity. By 2000, the number of urban dwellers increased significantly in almost all habitat types. However, the rank ordering of both urban population and urban population density per habitat area stayed similar to patterns in 1950.

The majority of the global urban population is currently located in the temperate zone (Fig. 3.6). At the turn of the twenty-first century, urban populations were largely located temperate zone between 25 and 55° North latitude. The percent of the urban population trails off approaching the equator with another small peak in

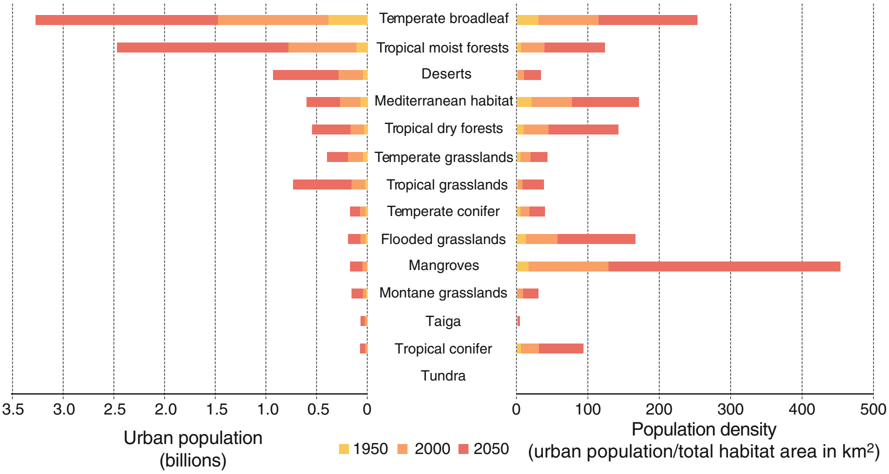

Fig. 3.5 Urban population by major habitat type (*left panel*) and urban population per total habitat area by major habitat type (*right panel*). Major habitat types and boundaries are taken from the World Wildlife Fund ecoregional dataset. Urban population information for 2000 taken from the Global Urban/Rural Mapping Program. Urban population information for 1950 and 2050 were interpolated from the GRUMP data based on rates of urban population growth taken from the United Nations Population Division (2011)

the South Temperate Zone. One might argue that this pattern has actually limited urbanization's direct impact on biodiversity to date as the tropical zones are the areas of highest concentration of different species.

In the future however, urban growth patterns will change. With urban growth, urban land use will likely double (McDonald 2008), although there is significant uncertainty in predicting how much urban population and urban area will increase (Seto et al. 2010). See Chap. 21 for detailed discussion of future urbanization scenarios and Chap. 22 for discussion of the biodiversity implication of these future urbanization scenarios.

This trend is visible in predictions of urban population by major habitat in 2050 (Fig. 3.5). Urban population will increase in essentially all habitat types. There will be particularly noticeable increases in urban population in tropical moist forests, deserts and tropical grasslands. Note that in terms of urban population per habitat area, there will be significant increases in impact in mangroves, flooded grasslands, and temperate broadleaf forests. Also worth noting are impacts to tropical conifer forests, a unique habitat type found only in a relatively small area globally.

Expansion of cities also fragments the remaining blocks of natural habitat. This increases the isolation of natural habitat patches, as the average distance between them increases. Increased isolation tends to reduce population and gene flow among patches, and may break a large regional population into several discrete subpopulations. Seasonal and intergenerational migration is also restricted. Highly mobile taxa like birds are generally less affected by isolation than less mobile taxa like

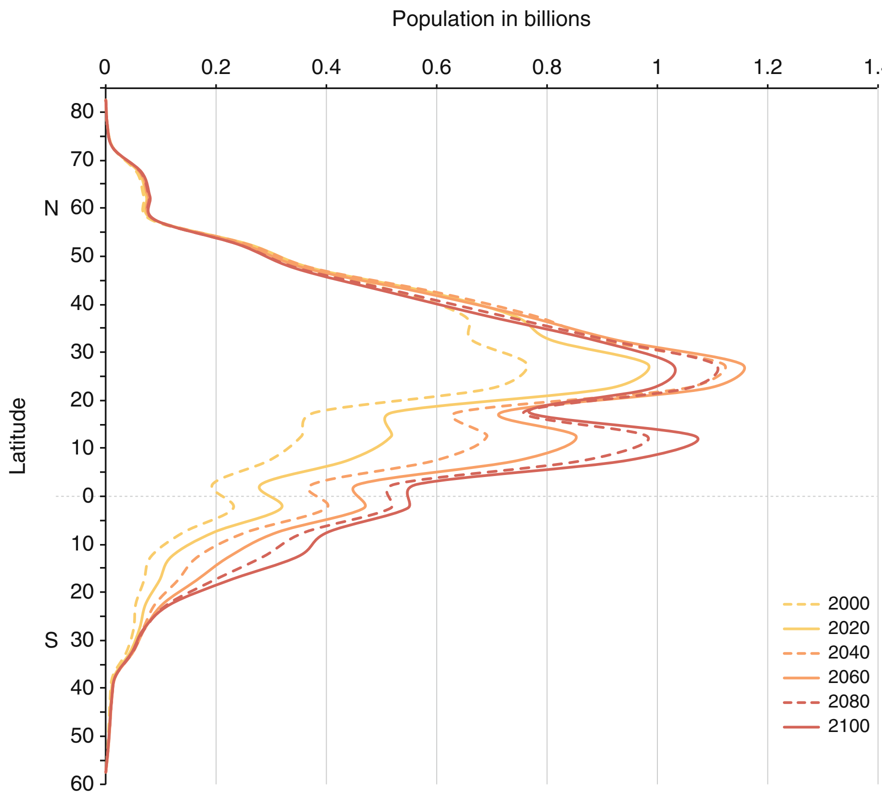

Fig. 3.6 Estimates of growth of global population by latitude, 2000–2100 (Data from United Nations Department of Economic and Social Affairs, Population Division, 2010)

amphibians, although some apparently mobile species avoid moving across urban land cover (Saunders et al. 1991).

Fragmentation necessarily increases the amount of habitat that is near a habitat/non-habitat edge (Murcia 1995). This systematically alters conditions near the edge, affecting the species and processes found there (Fagan et al. 1999). For example, at forest/non-forest edges, temperature is significantly increased during the growing season due to greater solar insolation. This increases average temperatures for tens of meters into the forest interior, the equivalent change in climate to a movement of hundreds of kilometers in latitude (Smithwick et al. 2003). Roads create a particular type of edge, with particular ecological effects (Forman 2000). Road noise is a commonly studied edge effect, and has been shown to significantly alter when and how bird species sing (Rheindt 2003). Finally, biotic interactions may change near edges. Birds' nests, for instance, are more likely to be parasitized by cowbirds when they are near an edge (Lloyd et al. 2005).

Urbanization increases the number and extent of non-native invasive species by increasing the rate of introduction events and creating areas of disturbed habitat for non-native species to become established (e.g., McDonald and Urban 2006). There is a suite of "cosmopolitan" species, skilled generalists, that are present in most cities around the world (McKinney 2006; Kuhn and Klotz 2006). Meanwhile, urbanization often leads to the loss of "sensitive" species dependent on larger, more natural blocks of habitat. The net result is sometimes termed "biotic homogenization." Species richness in cities may actually be higher than that of rural areas, depending on the richness of the suite of cosmopolitan species relative to that in natural habitat, but global species richness declines. The flora and fauna of the world's cities have become more similar and homogeneous over time, at least relative to the diversity of species composition prior to urbanization (Hobbs et al. 2006; Pysek et al. 2004; Grimm et al. 2008). Chapter 10 discusses this complex process in more detail.

3.3.3 Indirect Effects of Urbanization on Biodiversity

Cities may occupy a small percent of the global land area, but they contain the majority of the world's population and are concentrated centers of activity. These activities end up shaping land-use over a far larger land area, and influence the decisions of landowners and the policy decisions of governments in ever widening geographic extents. Chapter 26 examines arguably the most important indirect effect in terms of its areal impact, the impact of cities demand for food on global land-use.

The questions remain, however, how dense settlements interact with other human activities and what would happen if cities were removed from the equation. As mentioned previously, more specific policies focused on the process associated with urbanization may provide more valuable conservation tools than a general attack on cities. Three recent research findings that demonstrate our lack of knowledge on the exact role of urbanization and how examining interactions closely may help conservation efforts.

First, a recent article argues that international trade accounts for 30 % of all global species threats (Lenzen et al. 2012). While the demand for the goods traded probably originated in many of the world cities, this study emphasized better regulation, sustainable supply-chain certification and consumer product labeling as solutions. At the same time, however, there have been all too few studies that have examined the role of urbanization, trade and the environment. Obviously what is traded matters to the outcome of these relationships. How does, for example, the growing trade in electric bicycles to specific cities in the U.S. and Europe impact the environment? Has urbanization influenced production processes to lower environmental impact? Does the concentration of population and subsequent generation of "green" ideology have any impact on individual merchandise choice? In order to understand the role of urbanization in trade's impact on biodiversity, more study is needed to identify not only the distances of materials travel, but also where are they coming from before arriving at urban centers (Seto et al. 2012b).

A second study examined global material consumption over the past century. Researchers estimated that during this period, global materials use increased eight-fold to reach almost 60 billion tons (Gt) of materials per year (Krausmann et al. 2009). At the same time, the total population increased by four-fold. What is interesting is that is that over this century, materials use increased at a slower pace than the global economy, but faster than world population. Consequently, this research suggests that while material intensity (i.e., the amount of materials required per unit of GDP) declined, the materials use per capita doubled from 4.6 to 10.3 tons/cap/year. The role of technology and increasing wealth in these increases is clear. What is much less clear is the role of the growth of cities. During the past century the urban population increased approximately 18-fold. What was the urban impact on materials consumption? On one hand, cities may have helped to increase the rate of consumption through infrastructure development. Certainly, studies have demonstrated the large flows of material into cities as they grow (Decker et al. 2000; Kennedy et al. 2007). On the other hand, given that this infrastructure is shared by large numbers of people, urbanization could have slowed overall material consumption growth. That is, if populations were not densely organized, the levels of materials consumed may have been much larger. These questions suggest that cities and the urbanization process may have beneficial aspects that lower overall consumption levels.

Finally, a third research project examined the role of households rather than population in resource consumption and biodiversity loss. In this case analysts examined the decreasing size of households around the world and the impact of this trend on biodiversity (Liu et al. 2003). This research suggests that even when population size decreased in some locations, the number of households increased with subsequent increases in impacts. This work places the burden of responsibilities on the decreasing size of households (which increases demands for housing), rather than on urban population. The process of urbanization is often associated with economic development, which is in turn associated with smaller household size, but teasing out causality here is difficult.

These examples demonstrate that the indirect processes by which urbanization affects biodiversity loss are unclear, but potentially quite significant. Moreover, in many analyses it is difficult to separate the effect of urbanization *per se* from other confounding processes, like economic development and changes in demographics.

3.4 Global Urbanization and Freshwater Ecosystem Services

There are many different types of freshwater ecosystem services that cities depend on. Land cover in watersheds (including natural habitats) affects rates of evapotranspiration and hence the quantity of surface or groundwater available. In some cases, natural habitats have lower rates of evapotranspiration than anthropogenic land cover, while in other cases the converse is true. In certain climates, trees can also play an important role in increase precipitation, as fog settles out of the air on to foliage.

Land cover also affects many factors that impact water quality, including erosion, nutrient loading, and biogeochemical cycling. In many cases, natural habitats have lower rates of erosion and a greater capacity to absorb excess nutrients and other pollutants than anthropogenic habitats.

Thus, urbanization affects land cover, which in turn affects the quantity and quality of water available. But urbanization also requires water. Water is directly needed for human use, and supports a variety of other secondary ecosystem services (e.g., recreation, biodiversity, transportation). Globally, water consumption is greatest from the agricultural sector. The energy sector, however, withdraws a large amount of water for use in extracting and processing natural resources (e.g., coal, and cooling thermoelectric power plants). Urban consumption of food and energy contributes to increased water use in agriculture and energy, so in a certain sense a true accounting of cities' water use requires consideration of these linkages. For instance, the main water use of Chinese cities comes from the water needed to mine coal and burn it in thermoelectric power plants.

Urban residents need water for their daily activities (drinking, cooking, cleaning) as well as disposal of human wastes through sanitation systems. Per-capita water use substantially varies among cities. Within the United States for instance, residents in San Diego, CA use 700 l/person/day, while residents in Reno, NV use 1,166 l/person/day. Per-capita domestic water use tends to increase as the average income increases (FAO 2011). 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, poorer cities are more likely to have substantial populations without access to drinking water, decreasing aggregate demand for water. For instance, 27.6 % of Sub-Saharan urban residents lack access to clean drinking water, 12.3 % of Latin American and Caribbean urban residents, and essentially 0 % 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.

Three things must happen to ensure provision of fresh, clean water to urban inhabitants (McDonald et al. 2011a). First, enough water must be available. Availability, the absolute amount of surface or groundwater within a region that can be sustainably appropriated for urban use, is largely a function of climatic setting and land cover in the watershed. Second, the water must be of sufficient quality for use. Water that is polluted, either by upstream users or through pollution *in situ*, must be treated and purified before use in urban households. Third, a system must be in place to deliver that water to urban residents, usually via infrastructure such as piped water supplies, dams and canals, and wells.

Water availability is most likely to be a problem in cities in arid climates. One study (McDonald et al. 2011a) found that 21.7 % of urban dwellers, some 523 million, live in climates that would at least be classified as semiarid (Fig. 3.7). In the developed world these cities are clustered in the western United States, Australia,

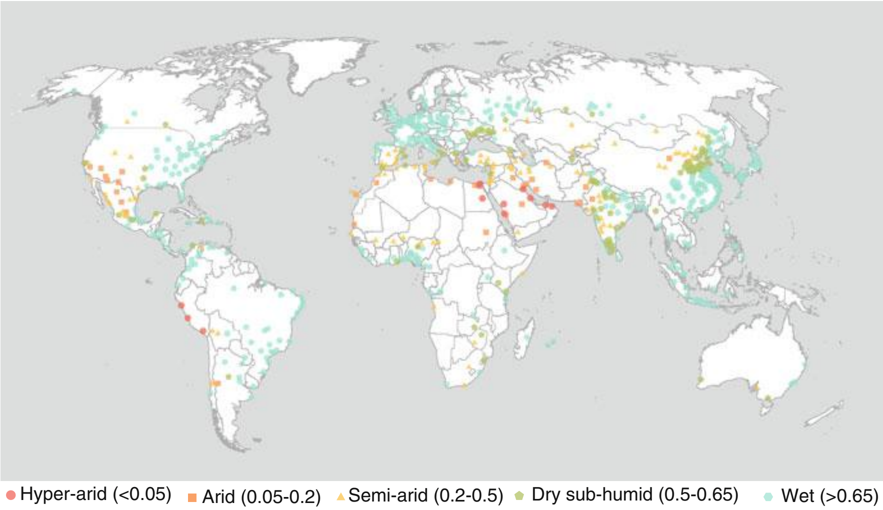

Fig. 3.7 Water availability for the world's cities. Water availability is measured by the aridity index, which is the ratio of precipitation to potential evapotranspiration (Adapted from McDonald et al. 2011a)

and parts of Spain. In the developing world most of these cities are located in northwestern Mexico, coastal Peru and Chile, North Africa, the Sahara, Namibia, the Middle East, and central Asia (see Fig. 3.7 for a map).

Water quality is most often a problem globally when there is significant human water use upstream. One useful proxy measure is the population density upstream, which correlates to several measures of water quality. One study (McDonald et al. 2011a) found that 890 million (36.9 % of the population of cities >50,000), are in cities with an upstream population density greater than 5.5 people/ha, the population threshold at which human activities often lead to nitrate concentrations that exceed the U.S. drinking water standard of 10 mg/l. Water quality issues affect all continents (Fig. 3.8), but tend to be concentrated in major river basins like the Ganges (India) and the Yellow River (China).

Water delivery is most a problem in rapidly growing cities with few financial resources. One study (McDonald et al. 2011a) found that 1.3 billion people (53.9 % of all urban population worldwide) live in cities with more than ten new residents per GDP per capita, mainly in sub-Saharan Africa, the Indian subcontinent, and Southeast Asia (Fig. 3.9). In contrast, some cities in developed countries have less than 0.5 new people per GDP per capita, and thus have roughly 20 times more financial capacity to deliver water to new urban residents than might a developing world city.

Cities have two broad sets of strategies to cope with insufficient water: those that involve building infrastructure to obtain more water than is currently available, and those that involve making wiser use of existing supplies.

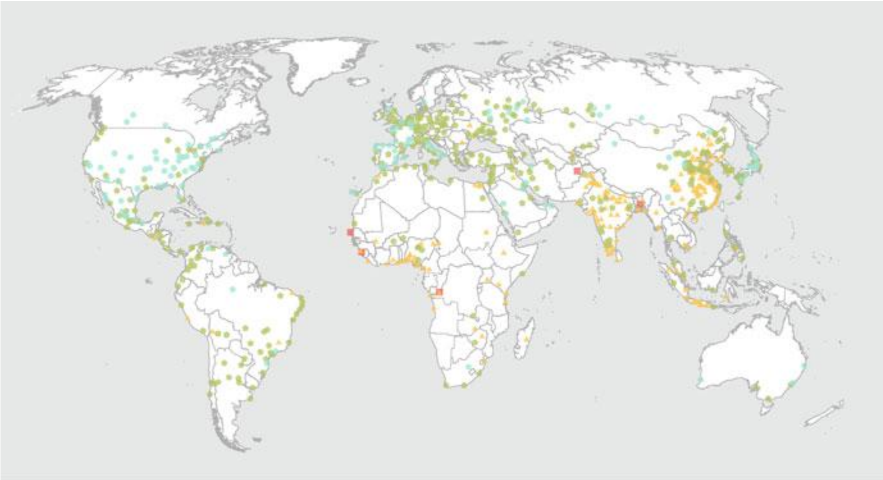

Fig. 3.8 Water quality for the world's cities. Water quality is measured as the density of people in upstream contributing areas (people/km^2), with population density and water quality exhibiting a negative correlation (Adapted from McDonald et al. 2011b)

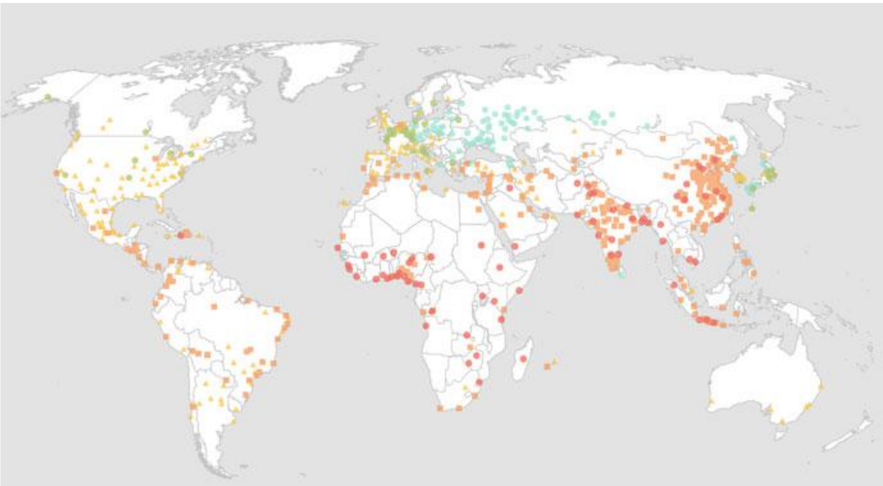

Fig. 3.9 Water delivery capacity for the world's cities. The ability of a city to delivery water to its citizens is measured as the number of people expected divided by per-capita GDP (Adapted from McDonald et al. 2011b)

The most common way cities try to obtain more water is tapping into groundwater to meet urban water needs. Groundwater use is sustainable if the rate of aquifer recharge exceeds the rates of withdrawals. However, for many arid cities, groundwater use exceeds the low rates of aquifer recharge. Mexico City has so overused its aquifer that the ground is subsiding 40 cm/year in some areas (Carrera-Hernandez and Gaskin 2007). Many other fast-growing cities face similar problems, but globally the extent of this groundwater mining by cities is unknown.

One common way cities try to make wiser use of existing supplies is by increasing water use efficiency, reducing the amount of water lost to leaks and trying to reduce per-capita water use for common tasks such as bathing and flushing the toilet. Another way is to improve watershed management upstream of reservoirs to prevent sedimentation and pollution from reaching reservoirs.

3.5 Summary and Conclusions

The broad global picture presented in this chapter suffices to show that global patterns of urbanization have had significant implications for biodiversity. In particular, urbanization as a driver of habitat conversion is already important and is expected to increase in importance in the future. Thus, urbanization is relevant to the Convention on Biological Diversity (CBD)'s Aichi Target 5 (*By 2020, the rate of loss of all natural habitats, including forests, is at least halved and where feasible brought close to zero, and degradation and fragmentation is significantly reduced*).

Habitat conversion driven by urbanization will be particularly important in tropical areas in the future and in coastal and island systems, as well as biomes that are disproportionately urbanized (e.g., Mediterranean habitat). CBD's Aichi Target 11 (*By 2020, at least 17 % of terrestrial and inland water, and 10 % of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and equitably managed, ecologically representative and well-connected systems of protected areas and other effective area-based conservation measures, and integrated into the wider landscapes and seascapes*) is unlikely to be met without addressing urbanization impacts in these places.

Similarly, the global analysis presented in this chapter shows that global urban growth will have significant implications for freshwater ecosystem services. Global urbanization will indirectly increase cities dependence on freshwater ecosystem services that control water quantity, quality, and timing. This has relevance to Millennium Development Goal's 7.B: (*Reduce biodiversity loss, achieving, by 2010, a significant reduction in the rate of loss*) and 7.C (*Halve, by 2015, the proportion of the population without sustainable access to safe drinking water and basic sanitation*). The remaining chapters will examine in more detail how cities depend on ecosystem services.

Finally, we suggest that urbanization should not be examined and framed solely as a problem or as a solution. It is dangerous for policymakers to consider urbanization solely as a problem, since it is an unavoidable part of economic development. A more useful way to think about global urbanization is as posing a series of social environmental challenges that must be overcome to achieve sustainability.

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Chapter 4
Regional Assessment of Asia

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Asia is home to 60 % of the world's population, and there are large variations in the region with regard to urbanization levels and urban growth rates. While some countries have populations that are predominantly urban (Singapore, 100 %; Malaysia, 72 %; Japan, 67 %; Indonesia, 54 %), others have populations that are predominantly rural (Bangladesh, 28 %; Vietnam, 29 %; India, 30 % Lao People's Democratic Republic, 33 %; Thailand, 34 %). Despite these variations, three characteristics define the region.

Many countries that are largely rural are undergoing massive *demographic and economic transitions*, resulting in a growing percentage of their populations living in urban areas. For example, the combined populations of Kolkata and Dhaka in the Ganges– Brahmaputra Delta increased from 4.9 million in 1950 to more than 30 million in 2010. The changing demography of these mega-deltas is also changing their economies, landscapes, and biodiversity (see Chap. 1, Fig. 1.1).

Half the increase in urban land across the world over the next 20 years will occur in Asia, with the most extensive patterns of change expected to take place in India and China.

The influx of large-scale capital to many Asian deltas has transformed the local economic base from a primarily agricultural one to a manufacturing and processing economy, bringing about *fundamental changes in landscapes and their ecologies*. For example, the Irrawaddy Delta economy in Myanmar was traditionally intensive rice cultivation, fishing, and forestry, supported by mangrove swamps. However, as Yangon, the largest city in Myanmar and the economic, financial, and trading hub of the country, increases in size on the periphery of the delta, it is affecting the coastal mangrove ecosystems. Urbanization and associated land practices—the damming of rivers, seasonal flood control, water diversions, agricultural practices, and construction of the built environment—have also transformed the supply and routing of sediments and changed the basic geomorphology and ecology of the delta (Textbox 4.1).

Textbox 4.1 Indonesia: Illustrating Asia's Three Development Characteristics

The 17,500 islands of the Indonesian archipelago and the surrounding seas contain the highest terrestrial and marine biodiversity on earth. However, the nation's ecosystems are severely impacted by land-use changes, which mainly follow two trends: urbanization and deforestation (Rustiadi and Panuju 2002). In the 1950s Indonesia's urban population was about 15 %; by 2010 half of the total population of 237.6 million was urban. The World Bank estimates that by 2025, Indonesia's urban population will represent 67.5 % of a total projected population of 270.5 million.

(continued)

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Textbox 4.1 (continued)

Across the archipelago, there are strikingly different patterns of urbanization and economic growth. About 68 % of the urban population lives in Java. Although there are many cities in Java, by far the largest is the capital city of Jakarta. The greater metropolitan region of Jakarta is the second largest in the world, 5,897 km^2 containing nearly 12 % of the entire Indonesian population (Hudalah and Firman 2012). Jakarta is now so densely populated that it is estimated that traffic will come to a complete standstill by 2014. However, as Indonesia's largest cities continue to expand, a reverse trend sees more affluent classes moving into the surrounding countryside to escape the excessive concentration, physical congestion and breakdown of urban services and amenities (Rustiadi and Panuju 2002). In Jakarta, the suburban population surpassed that of the city by the 1990s.

While the largest Indonesian cities and their suburbs have grown at a record-breaking pace, the annual population growth rate in small and medium-sized Javanese towns is far below the national average and even decreasing. This pattern is also found elsewhere in Indonesia: small cities with populations in the range of 100,000–500,000 actually lost population between 1993 and 2007, with average declines of more than 2 % per year.

These demographic patterns reflect several large historical trends. In 1967, Indonesia launched the Green Revolution in agriculture and the same year, the forests of the Outer Islands were opened to logging for export. At about the same time, other extractive industries like mining and petroleum began to take off. These neoliberal economic policies triggered large-scale migration from the countryside to the cities. The market-oriented policy of the 1980s boosted the economy but also led to an uncontrolled growth of large-scale private land development in the suburbs of Jakarta (Firman 2000). In the 1990s, economic growth became erratic, as a consequence of political and financial crises but as the economy recovered, so did urbanization. Between 2000 and 2005, an estimated 22,872 ha of land were converted to built-up areas (Hudalah and Firman 2012).

By the 1990s, much of Indonesian Borneo had been deforested, leaving logging debris in place of canopy trees. During the El Nino drought in 1998, 600 million tons of carbon were released from the forests of Borneo into the atmosphere. For comparison, that year the Kyoto target for reduction in carbon emissions for the Earth was 500 million tons. The smoke from the burning forests, so dense that at times the airport in neighboring Singapore had to be closed, created a crisis in air quality for Borneo.

By 2004, the volume of timber exports from Borneo exceeded all tropical wood exports from Africa and Latin America combined. Largely as a consequence of deforestation, Indonesia became the third largest source of carbon emissions, after the US and China. The pace of deforestation continued

(continued)

Textbox 4.1 (continued)

with the massive expansion of oil palm plantations. From 1990 to 2010, over half a million km^2 of oil palm plantations were planted in Indonesian Borneo, 90 % on formerly forested land, which is projected to significantly increase their contribution to Indonesia's 2020 CO_2-equivalent emissions of 0.12–0.15 GtC year^{-1}) (Carlson et al. 2013).

The disappearance of the forests and the expansion of oil palm plantations caused massive relocations of rural populations in Borneo, Sumatra and smaller islands. On Java and Bali, population growth and urbanization have increased while the expansion of monocrop agriculture has led to steep declines in biodiversity in the countryside.

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Chapter 5
Sub-regional Assessment of China: Urbanization in Biodiversity Hotspots

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China has been urbanizing rapidly since the early 1980s. This is manifested by large rural-urban population migrations and by the expansion of urban areas and the built environment. One consequence of urban expansion has been the loss of fertile agricultural land. Another, less noticed, has been the urban expansion within biodiversity hotspots. Throughout the country, expansion of urban areas have increasingly threatened habitats critical for conservation of biodiversity (McLaren 2011). Especially along the coast, many ecosystems have been destroyed as a result of continuous building and development (Zhao et al. 2006). On the other hand, further inland and especially along major rivers, the economic development and urban growth has increasingly been impacting ecologically sensitive lands (Li 2012).

China is also among the most biodiverse countries in the world (McNeely et al. 1990; López-Pujol et al. 2006). The country contains four biodiversity hotspots that are home to significant diversity of endemic species that are threatened by human activities. The number of protected areas (PAs) in the country has increased in recent decades; a recent study identified China as a nation with 1,865 nature reserves (Wang et al. 2012), covering more than 10 % of the country's territory. These PAs are particularly concentrated across the eastern half of the country where urbanization has also been the most dramatic.

Apart from studies that evaluate the past and current impacts of development on the country's biodiversity there has been sparse quantitative analysis of the implications of future urban expansion. A recent study predicted that proximity of urban areas to PAs in the country will dramatically increase by 2030 (McDonald et al. 2009). By 2030, the urban population of China is expected to be over 900 million, an increase of over 300 million (UN 2010). While there are uncertainties around this estimate, there is even greater uncertainty about the location and amount of future urban expansion.

Recent analyses indicate that nearly half of the increase in urban land across the world is predicted to occur in Asia, with the largest increases in China and India (Seto et al. 2012). Within China, urban expansion is predicted to create a 1,800 km coastal urban corridor from Hangzhou to Shenyang. As urbanization progresses towards the western regions of the country, more of the biodiversity hotspots are likely to be affected by development and urban land conversion. A recent study forecasts direct impacts of urban expansion on biodiversity in China, but it does not elaborate on how these forecasted impacts vary across the country (Güneralp and Seto 2013). On the other hand, while invaluable to develop our understanding between urbanization and biodiversity conservation at specific localities, case-based studies are too few to generate a comprehensive outlook across the country.

Of the 34 biodiversity hotpots identified around the world (Myers et al. 2000; Mittermeier et al. 2004), four remain partially within China's borders: Himalaya, Indo-Burma, the Mountains of Central Asia, and the Mountains of Southwest China (Fig. 5.1). In 2000, about 13 % of the total urban land in China – a little over 10,000 km^2 – were located within these hotspots. Importantly, the urban land in the Indo-Burma hotspot constitutes 92 % of the total urban land across all four biodiversity hotspots. The Indo-Burma hotspot extends across several provinces; for example, Guangdong province had around 85 % of its total urban land area located

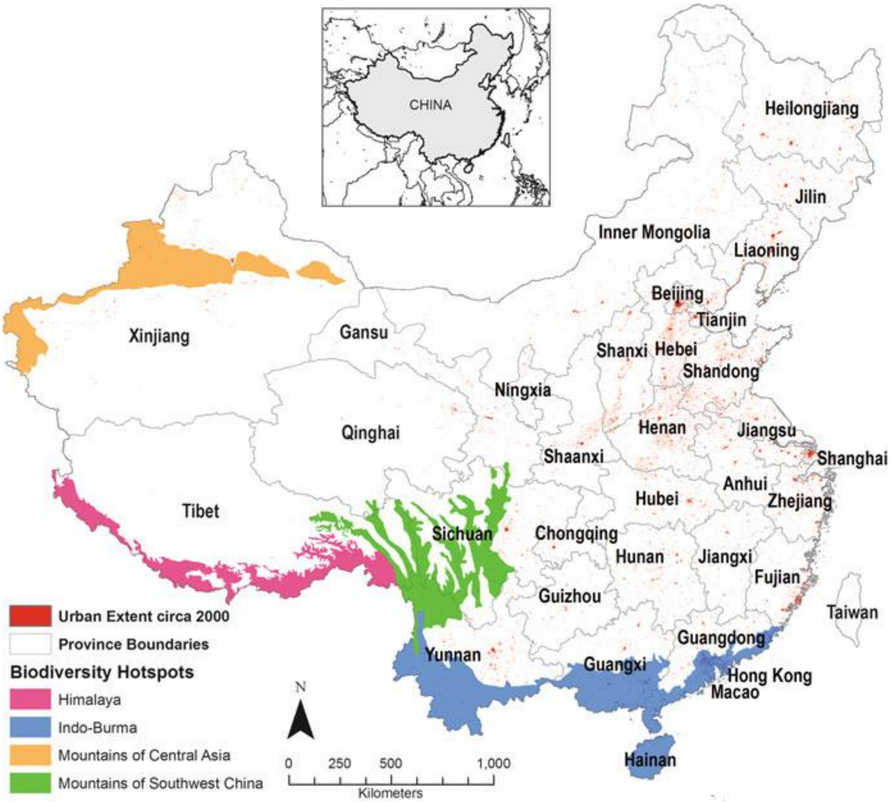

Fig. 5.1 Biodiversity hotspots and urban extent in China circa 2000 (Prepared by and published with kind permission of ©Burak Güneralp 2013. All Rights Reserved)

within the hotspot in 2000 (Fig. 5.2). Moreover, the province accounts for more than two thirds of the total urban land cover in this hotspot and the most urban land in any biodiversity hotspot across China. It is followed by Guangxi and neighboring Yunnan, both of which have southern portions of their land in the Indo-Burma hotspot. Xinjiang in the northwest of the country also has considerable urban land (about 500 km^2) in the hotspot Mountains of Central Asia; that is equal to about one fourth of the total urban land in the autonomous region (Fig. 5.2).

Based on the IPCC scenarios and projected urban expansion rates in Seto et al. (2011), the urban land in biodiversity hotspots is projected to increase from about 10,000 km^2 in 2000 to somewhere between 40,000 and 77,000 km^2 by 2030. Of the four hotspots, Indo-Burma, which contained by far the most urban land (more than 9,000 km^2) in 2000 (Fig. 5.1), is projected to have between 35,000 and 70,000 km^2 urban land by 2030.

Apart from the Mountains of Southwest China hotspot which is nearly completely within China's borders, the implications of urbanization in the other three

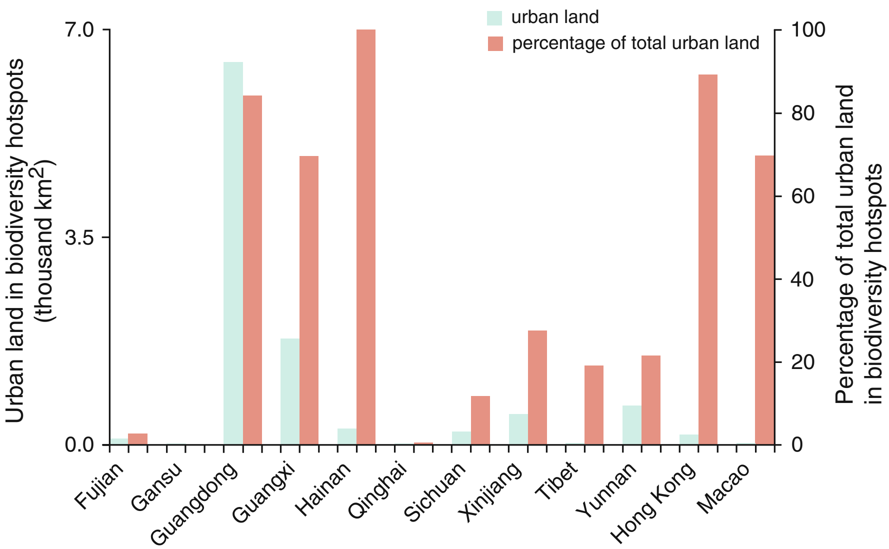

Fig. 5.2 The amount of urban land within biodiversity hotspots in 2000, and the corresponding percentage of the total urban land area in each of the provinces and special administrative regions that contain biodiversity hotspots (Published with kind permission of Burak Güneralp and modified by Maria Schewenius. ©Maria Schewenius 2013. All Rights Reserved)

biodiversity hotspots for their biodiversity and ecosystem functioning can be more accurately assessed through trans-border regional cooperation between China and its neighbors (Chettri et al. 2007). Such cooperative initiatives are especially pertinent for the Indo-Burma and Himalaya hotspots because urbanization is also rapidly progressing in those parts of these hotspots that are in other Southeast Asian countries including India (Seto et al. 2012). In addition, while not located within any hotspots, large urban agglomerations such as Chengdu in Sichuan and Ürümqi in Xinjiang are within less than 20 km of the Mountains of Southwest China and Mountains of Central Asia hotspots, respectively. The land use policies in such urban agglomerations should include strategies to direct growth away from the biodiversity hotspots.

Considering the primary importance of economic growth placed in performance evaluations of the local governments, proper evaluation of ecosystem services in these hotspots gains urgency so that they can be included in economic considerations of the local governments. In China, there are wide variations across the provinces in terms of the amounts and/or proportions of urban land in biodiversity hotspots. These differences across the provinces call for differentiated strategies to manage urban expansion to minimize its negative impacts on biodiversity and ecosystem functioning.

The threats to biodiversity hotspots come from direct land cover change that causes habitat loss and degradation of ecosystem functioning as well as indirect effects of urban encroachment. One such indirect effect is the increased incidence of colonization by introduced species as urban areas expand into these hotspots. Going beyond the physical expansion of urban areas in or near the biodiversity hotspots, the consumption patterns of urban inhabitants in general can adversely affect biodiversity and ecosystems in these sensitive areas even if they are not located in close proximity to each other (Seto et al. 2012). In particular, the reduction in household size with increasing urbanization has been shown to have large impacts on resource consumption and biodiversity (Liu et al. 2003). Moreover, urban expansion and population growth in one location may have knock on effects leading to land change cascades that can extend well into the more sensitive parts of biodiversity hotspots – both within the same country and across continents (DeFries et al. 2010). Such challenges cannot be met by local-level solutions only; they require policy responses at a much larger scale and thus call for appropriate strategies with sufficient breath, to be developed at the national and international levels.

Minimizing habitat and biodiversity loss and limiting degradation of ecosystem services require integrating ecological knowledge into urban and land use planning practices (Niemelä 1999) so that these practices become more attuned to conservation of biodiversity and preservation of ecosystem services (McDonald et al. 2008). However, if the past three decades are any indication, urban expansion dynamics in China will primarily be dominated by economic forces, which includes the role played by land transactions as a source of income for local governments (Frederic and Huang 2004; Yew 2012). Therefore, the current land market system needs to be reformed for urban planning to attain any meaningful level of success in conservation of biodiversity and preservation of ecosystem services.

Despite upbeat assessments on the trends of biodiversity loss in China (Xu et al. 2009), there is still cause for concern (Liu and Diamond 2005). How urban areas continue to expand may affect larger expanses of the biodiversity hotspots in the coming decades. There is thus a need for forward-looking studies to understand the likely rates, magnitudes, and patterns of urban expansion within the biodiversity hotspots at a range of spatial and temporal scales.

The scale of urbanization in China has so far been extraordinary and there is every indication that it will remain so in the coming decades. Thus, the impact of the country's urban growth on biodiversity and ecosystems may surpass the extent of impacts we have witnessed across the world so far. The preliminary forecasts reported here are limited to the biodiversity hotspots, one of several conservation prioritization concepts. Another forecasting study reported that proximity of urban areas to the nature reserves in China will also dramatically increase by 2030 (McDonald et al. 2009). Moreover, these forecasts inform about the potential direct impacts of urban expansion on biodiversity, but not about its indirect impacts due to increasing demand for natural resources originating in urban environments. One such indirect impact is the construction of dams and other infrastructure to meet the rising energy demands mostly originating from urban areas. For example, the existing and planned dams along the Chinese portion of the Mekong River will have significant impacts on biodiversity both through land changes (including inundation behind dam walls) and through alteration of river flow (Dugan et al. 2010; Barrington et al. 2012). Both direct and indirect impacts of urbanization need to be taken into consideration for a complete account of its environmental impacts. Nevertheless, there is a critical window of opportunity in the next few decades for China to implement more proactive approaches to guiding urban expansion in ways that least negatively impact biodiversity and ecosystems.

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Chapter 6
Sub-regional Assessment of India: Effects of Urbanization on Land Use, Biodiversity and Ecosystem Services

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6.1 Introduction

India is increasingly marked by the growing influence of urban areas, with large-scale, distal impacts on rural environments across the country. These changes will impact land cover, natural habitats, biodiversity and the ecosystem services that underpin human well-being.

Until recently, rural development was a major focus in India. This changed in 2005, when the launch of the Jawaharlal Nehru National Urban Renewal Mission shifted the focus to development of 63 urban centers throughout the country. Reforms in India and national policies now treat urbanization as central to economic and industrial development, and there is an explicit strategy to develop cities. One of the largest examples is the developing Mumbai–Delhi industrial corridor, which is approximately 1,500 km long and connects two of the country's megacities (United Nations Human Settlements Programme (UN-HABITAT)). The government is also establishing special economic zones, industrial and technology parks, and free-trade zones that will further focus urban expansion in specific locations. These urban clusters are likely to transform entire regions, with significant impacts on habitat and biodiversity.

Urbanization has major impacts on rural areas, reshaping lifestyles, livelihoods, and patterns of consumption and waste generation. Demands from urban populations decrease the supply of natural resources in far off areas, and increase pollution within and outside cities. This is often exacerbated by both lack of appropriate policies for managing these effects, and poor regulation and enforcement (Aggrawal and Butsch 2012). Thus, the ongoing and anticipated massive increases in urban population across India are bound to have significant implications for the country's environment, ecology, society and sustainability.

Urbanization in India also presents opportunities for the environment. For instance, at the national level following promotion of the transition from fuelwood to liquefied petroleum gas for household energy use in cooking, the urban fuelwood demand declined from 30 % of households in 1993, to just 22 % of households in 2005. This has reduced the pressure on forest habitats near urban areas. Cities can also serve as nodes for ecosystem recovery. For instance, in Navi Mumbai, decreased pressure on mangrove forests has lead to a remarkable recovery in the past two decades. In Bangalore, collaborations between municipal government and local communities have led to a growing movement towards the restoration of lakes. In Surat, a focus on integrated waste and sewage management has provided impressive results.

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This assessment provides an overview of the environmental and ecological implications of urbanization in India, discussing challenges as well as opportunities for future sustainability.

6.2 Patterns of Urban Expansion: Results from Remote Sensing Studies

Indian cities are expanding in number, density and size (Fig. 6.1). Currently, India's urban population is around 377 million people, or 30 % of the nation's total (JNNURM Directorate, Ministry of Urban Development and National Institute of Urban Affairs 2011). By 2031, the urban population in India is expected to nearly double, reaching

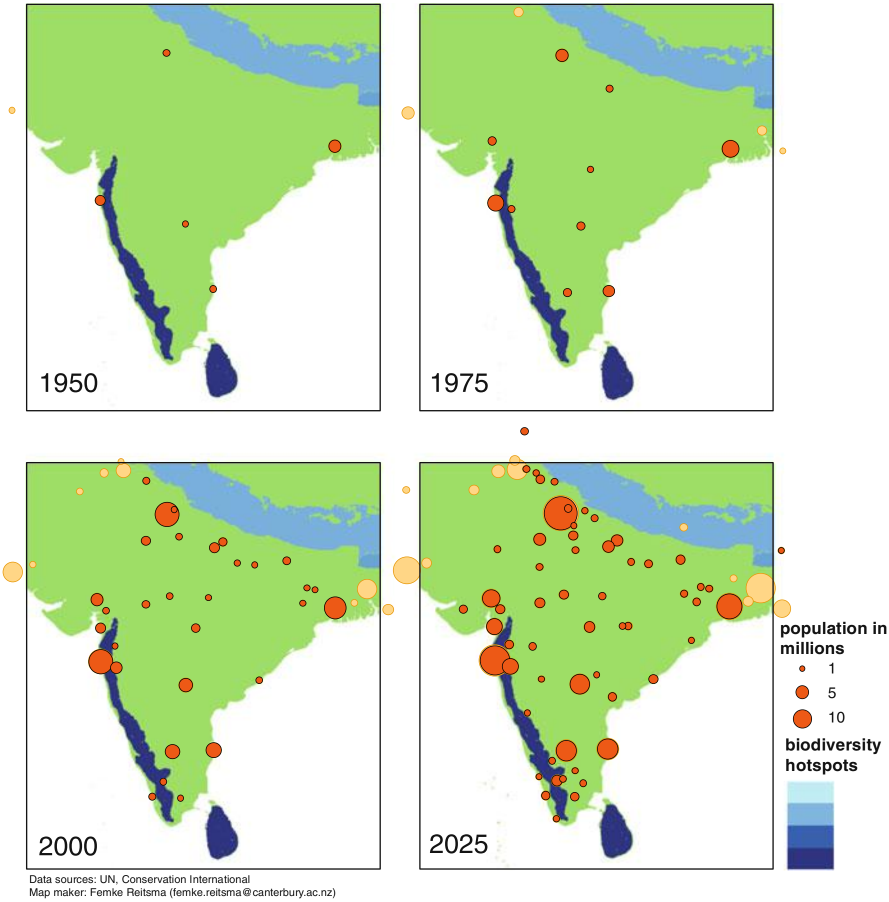

Fig. 6.1 Urban population growth in India (*red dots*) and the surrounding region (*orange dots*) 1950–2025 (Prepared by and published with kind permission of ©Femke Reitsma 2012. All Rights Reserved) (Color figure online)

600 million people (United Nations 2011). In the past 20 years, the built area in the top 100 cities alone has increased by almost 2.5 fold or over 5,000 km^2.

India already contains three of the world's ten largest cities, Delhi, Mumbai and Kolkata, as well as three of the world's ten fastest growing cities, Ghaziabad, Surat and Faridabad. Further impacts on ecosystem services, green spaces and connectivity will take place as large and growing cities merge into city clusters (such as Pune-Mumbai). However, about half the country's urban populations live in smaller urban agglomerations with under 100,000 people.

The development patterns of Indian cities are additionally shaped by their unique history, topography, planning and management. For instance, Pune and Bangalore retain significant green space in the city core despite rapid development and growth due to the presence of institutions such as the military and public sector companies, which protect large green patches.

Urban growth in India is often nucleated, with newly urbanized land usually seen in a tight band around the older parts of the city. In high growth cities like Bangalore and Pune, the city center maintains a fairly steady population because of a scarcity of land, while the city grows outwards, leading to increased fragmentation at the periphery (Taubenböck et al. 2009). In the smaller city of Lucknow, growth is largely in the city core due to infilling, which can lead to greater impacts on biodiversity in the center of the city, and can impede species movements through the urban landscape (Schneider and Woodcock 2008).

6.3 Impacts on Urban Ecosystem Services

Accelerated urban growth presents several difficult challenges for the natural environment in Indian cities. Increasing pollution of water and air degrades ecosystems. A continuous encroachment and transformation of ecosystems from woodlands, grass lands, coastal areas, wetlands and water bodies into urban concrete jungles further degrade them (Nagendra et al. 2012). The remaining green spaces in many cities have been transformed from their original state and species compositions to human-designed, landscaped and pesticide-intensive parks.

Further transformation of urban ecosystems is driven by their vulnerability to invasive species, such as the water hyacinth suffocating urban water bodies. Cities can also become nodes for the spread of invasive exotic species into surrounding non-urban habitats, such as the exotic *Lantana camara*, which was introduced to India as an ornamental garden plant, but now chokes forest understories throughout the country. Native bird species diversity has been shown to decline with an increase in exotic plant species in Delhi, and the same has been found in other cities in the world (Khera et al. 2009). This has disturbing implications for Bangalore, where 80 % of the trees found in parks are exotic (Nagendra and Gopal 2011). Enhancing the amount of green areas in cities with native species, as has been done in Mumbai, holds the potential to offset some of this development.

However, as cities and the climate change, some exotic species may have higher survival rates compared to native species and provide support to other species, or services for humans. It is thus highly important to understand how exotic and native species impact both humans and the ecosystems in urban areas.

The high population density in many Indian cities and towns creates particular challenges to mitigate the impact of climate change. A major challenge will be to manage scarcities and excesses of water. Coastal and inland cities located near rivers, such as Mumbai, Kolkata, and Delhi will have to deal with increased risk and intensity of flooding. The most vulnerable urban residents tend to be socio-economically deprived. They also tend to live in informal or traditional settlements, located in areas at greatest risk for flooding or landslides and at greatest risk of eviction during environmental crises.

Problems of water scarcity due to unpredictable rainfall will intensify as climate change accelerates, especially affecting cities in semi-arid areas such as Bangalore. Measures such as rainwater harvesting need to be intensified. Well-functioning ecosystems can be critical in ensuring greater food and water security for the most vulnerable in times of climate change. Urban forests have the potential to reduce air pollution and decrease urban heat island effects, while urban wetlands and lakes can reduce flooding, increase groundwater recharge, and stabilize soil. Improving solid waste management is also critical to maintaining the quality of urban ecosystems and life.

6.4 Impacts on Biodiversity

A major element of India's projected urbanization will take place along the coastlines through the growth of existing coastal cities and proposed and ongoing development of major new ports. This threatens important coastal regions through destruction of sensitive habitats such as mangroves and sea turtle nesting beaches, and increased demand for fish, turtle eggs and other seafood. Building construction close to the shoreline, along with mangrove destruction, also leaves cities more vulnerable to flooding and other damage from natural disasters like cyclones and tsunamis, and projected sea level rise from global climate change. Future development along the coastline must incorporate strategies to maintain and restore natural vegetation as a buffer along the water's edge.

Box 6.1 Landscape Transformation and Ecosystem Opportunities: The Example of Mumbai

What is today the city of Mumbai started as a group of islands but urbanization claimed land, which led to infilling of tidal flats, and conversion of mangroves for urban development. This has increased the city's vulnerability to flooding and anticipated sea level rises due to global climate change.

However, there has been some recovery of mangrove forests in the Navi Mumbai corridor along the eastern side of Thane creek since the mid-1990s.

(continued)

Box 6.1 (continued)

This can be linked to a decrease in the dependence on fuelwood by former villages that were overtaken by urban development, and a shift to alternatives such as compressed natural gas and electricity. Simultaneously, the creek also became an important wintering ground for a large population of Lesser Flamingoes. Unfortunately, the new proposed airport development in Mumbai threatens to destroy much of this newly re-created habitat. Thus, changes in human resource use can have immediate consequences for ecosystem degradation as well as for restoration.

6.5 Challenges of Governance

Governance of ecosystems in India is characterized and shaped by a complex network of actors interacting on multiple levels, including but not confined to the judiciary, elected officials, city municipalities, corporate and public sector agencies, Non-Governmental Organizations (NGOs), local community groups, research institutions, and activist groups.

Elected officials, judiciary, city municipalities, and planners can devise and seek to implement laws and regulations, but the involvement of community groups, corporate and public sector agencies and NGOs is important to ensure knowledge sharing, and willingness to follow regulations. In this context, informal, loose coalitions of different social, economic and interest groups are gaining increasing influence in negotiating local-scale agreements about resource use, and in providing important links with official institutions. They also strengthen the governance capacity of local municipalities, who face knowledge constraints and resource and manpower limitations that restrict their ability to effectively implement regulations limiting the over-use and exploitation of urban ecosystems.

Thus, a range of informal and formal institutions play important roles in making diverse perspectives and needs of different social and economic groups heard by decision-makers. They can also increase knowledge dissemination within their own groups and implement sustainability initiatives at a micro-scale that can become very valuable when accumulated at a city scale. Examples include wildscaping of local gardens in Pune, solid waste management in Chennai, and lake restoration and governance in Bangalore.

Another example is the case of India's capital, Delhi, which saw a rapid increase in air pollution in the 1990s. Interventions by the Supreme Court of India, followed by pressure from civil society groups, led to the implementation of a number of policies designed to reduce air pollution, resulting in an impressive drop in air pollution levels. Recent years have seen an increase in air pollution again, due to rapidly growing numbers of private vehicles. This is a challenge for most Indian cities, which lack sufficient and reliable public transport.

6.5.1 City Municipalities

The city of Surat, Gujarat state, Western India, is the fourth fastest growing city in the world. Yet, over the past couple of decades it has transformed into one of the cleanest cities in India, with an excellent public bus service, well planned water distribution and well functioning waste management and treatment plants. A key factor was the implementation of a well designed municipal management, brought about by streamlining of functional, administrative, financial and technological bodies within the municipality, in collaboration with NGOs, local community groups and the public. It is critical that municipalities work pro-actively to avoid problems rather than tackling them after they appear. It is also important to actively involve representation from a variety of social and economic groups, providing ecosystem and environmental protection and restoration, while also paying attention to issues of equity, social justice and human wellbeing.

6.5.2 Media and the Civil Society

Media play a key role in highlighting environmental and development issues for public awareness. In addition to traditional media such as newspapers and television, which are widely accessed across India, social media, such as Facebook, email list services, Twitter and blogs, have emerged as tools allowing people on multiple levels in society to share information, and monitor authorities' activities.

6.5.3 Sacred and Cultural Traditions of Conservation in India

History and cultural preferences for specific types of landscaping and biodiversity play a major role in shaping Indian urban ecosystems. In the capital city of Delhi, the trees in the old city where the British influenced landscaping, differ clearly in distribution and species composition from the trees in the new gated communities at the urban periphery, such as Gurgaon (King 2007). Similarly, in Bangalore, older parks are more wooded, while newer landscaped gardens tend to be dominated by neatly trimmed shrubbery, which may appeal more to the wealthier of the city's residents (Nagendra and Gopal 2011).

India has a long, rich tradition of conservation associated with sacred religious and cultural beliefs. Sacred groves are conserved in many peri-urban areas and smaller towns, while it is quite common to find massive, centuries-old sacred trees being protected in densely congested urban neighborhoods across India. These trees act as keystone species and provide important support for urban wildlife. Other habitats and species such as bat roosts, Bonnet macaques, hanuman langurs, and fish are protected in certain areas. People also feed urban wildlife during certain

Box 6.2 Cultural Influence Shaping Urban Ecosystems

Today one-third of the world's population lives in slums. In Bangalore, the awareness of the importance of biodiversity in slums was very high, with plant species providing crucial services to inhabitants, acting as sources of shade, physical support, food, and medicine. Trees and plants also had social and psychological significance, being important for cultural and religious ceremonies and beliefs.

While trees and plants in wealthier residential areas in Bangalore are of aesthetic and cultural value that can be seen as an extension of people's lifestyles, greenery in slums is very much a part of people's livelihoods. As a recent study found, the extremely difficult conditions under which the majority of the slum residents managed their daily activities meant most of their days were spent outside. Canopy trees provided shade, which is of increasing importance as the number of people increase and as summers are increasingly hot. The trees also supported a variety of professions: flower selling, broom making, incense sticks making, and the running of a mechanic shop, tea stalls and telephone booths (Gopal 2011). All slums had potted plants in addition to trees, grown in a variety of containers due to space constraints, mostly representing species that had direct value for consumption as food, for worship, or for medicinal use.

times of the day (Jaganmohan et al. 2012). Water, wetlands and lake ecosystems also occupy a prominent position in many Indian cultural traditions, with traditional restrictions on the conservation and management of fresh water resources, maintained through worship of local lake deities. Although disrupted by urbanization, many of these practices continue to survive in Indian urban areas. Such traditions can be very influential in providing a unique, India-specific path for sustainability in an urban future.

6.6 Conclusions

India is facing a massive increase in urban population, from 377 million people in 2010 to 600 million in 2031, in part because the country is investing heavily in large-scale infrastructures such as roads, telecommunications, water networks, and power and electricity grids. This increase is bound to create massive challenges for the environment, ecosystems and human well-being in India, and the challenges need to be addressed upfront. City planning, infrastructural development and the consumption patterns of urban inhabitants will impact ecosystems within cities as well as far beyond the city boundaries, with implications for the quality of life for people across the country.

Cities can and do harbor great biodiversity, in many cases managed and maintained by citizens of different levels of society, ranging from the wealthy to the underprivileged. This illustrates a great potential and opportunity within cities. Many Indian cultural traditions are associated with nature and its protection, which has added to the resilience of urban green and blue spaces.

Informed decision-making for ecosystem protection, management and restoration will be of increasing importance in the era of climate change. For this, sustainable planning and implementation is required, building on inclusion of people and groups from all levels and backgrounds. A network of official governance institutions, civil society groups and individuals can contribute to informed decision-making and effective implementation, with social and ecological well-being as the main focus.

As this chapter highlights, there are opportunities and success stories, as seen in the large scale involvement of NGOs, civil society groups and local communities from diverse socio-economic backgrounds, including the most underprivileged, in ecosystem protection and biodiversity maintenance. Such community initiatives to reduce urban ecological footprints, improve solid waste management, rainwater harvesting and lake restorations, need to be supported and scaled up to the national level to meet the challenges abound. It is essential and urgent that India finds ways to balance economic growth with reduction of pressure on ecosystems to ensure a secure, equitable, and sustainable future.

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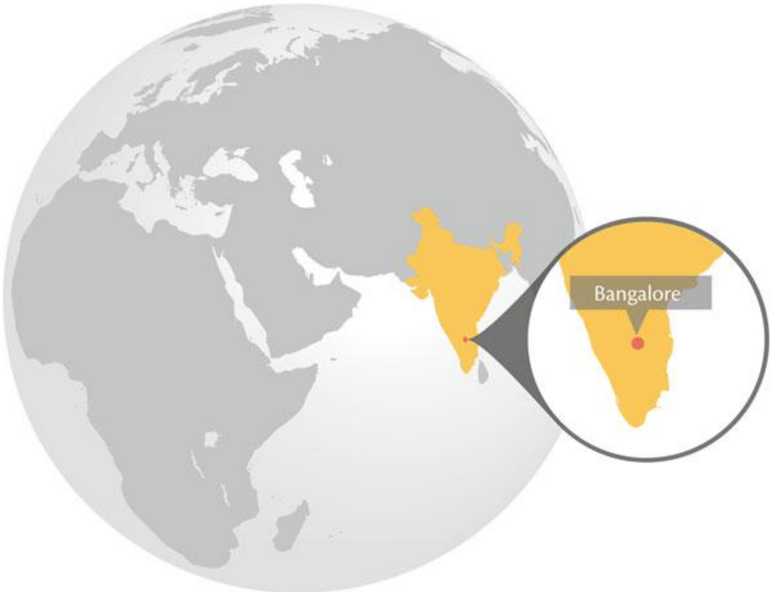
Chapter 7
Local Assessment of Bangalore: Graying and Greening in Bangalore – Impacts of Urbanization on Ecosystems, Ecosystem Services and Biodiversity

H.S. Sudhira and Harini Nagendra



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T. Elmqvist et al. (eds.), *Urbanization, Biodiversity and Ecosystem Services: Challenges and Opportunities: A Global Assessment*, DOI 10.1007/978-94-007-7088-1_7,

Abstract Bangalore is the principal administrative, cultural, commercial, industrial, and knowledge capital of the state of Karnataka, with a population approaching nine million. Economic growth has had a major impact on ecosystems and biodiversity, leading to the encroachment and pollution of water bodies, the felling of thousands of trees, and urbanization of green spaces. The city periphery experiences accelerated growth, with changes in ecosystems, land use and governance leading to impacts on ecosystems and biodiversity. Vegetation in the city core is species rich but less dense compared to other cities, with a high proportion of exotic plant species, and high faunal and insect diversity, although shaped by social preferences that vary across location and time. Bangalore's green spaces and lakes are embedded within multiple land use categories, and governed by a multiplicity of institutions with overlapping, often uncoordinated jurisdictional responsibilities. Civil society also significantly shapes the environmental agenda in Bangalore, taking an active and vibrant role in respect of environmental issues. In the coming decades, climate change and scarcity of access to clean water are likely to pose significant challenges for the city, exacerbated by the loss of lakes, wetlands and green spaces. Socio-economically vulnerable populations will be especially susceptible to these changes. In this context, Bangalore's cultural character, as a location of significant civic and collective action, will play a very important role in shaping urban environmental protection and conservation efforts, with collaborations between citizens of different economic strata and government agencies playing an increasingly critical role.

Keywords Economic growth • Urban sprawl • Traditional knowledge • Keystone species • Lakes

Key Findings

- Bangalore is simultaneously sprawling and growing rapidly in population
- The official governance structure is heavily compartmentalized and fragmented
- The lakes and green spaces in the city are rapidly transforming or disappearing
- Civic initiatives play an important role in providing protection for the urban ecosystems
- The city's poor often play a direct role in maintaining the biodiversity, while also being the most vulnerable to environmental hazards.

7.1 Introduction

Bangalore is the principal administrative, cultural, commercial, industrial, and knowledge capital of the state of Karnataka. The city is geographically located at 12.95° N latitude and 77.57° E longitude and situated on the Deccan plateau, at an altitude of 920 m above msl. Bangalore is a fast growing incipient megapolis, with an increase in population from 163,091 during 1901 to 8,499,399 as per the 2011 Census (Census of India 2011). With the advantages of booming economic activity, availability of land for expansion, and the city's year-round favorable climate due to its location at a higher elevation; population growth, migration and expansion has been extensive, leading to urban sprawl and landscape fragmentation in and around Bangalore (Sudhira et al. 2007; Nagendra et al. 2012).

Bangalore was known as a tiny village in the twelfth century and has grown through the intervening centuries, to emerge as the fifth largest city in India today. Popular belief associates this with an agricultural ecosystem service, considering the name "Bengaluru" (the city's name in the local language Kannada, from which "Bangalore" has been anglicised) to be derived from "*benda kaalu ooru*" – town of boiled beans. Tradition associates the twelfth century Hoysala King Vira Ballala with the origin of this name. The tale states that Vira Ballala once lost his way during a hunting expedition in this region and reached the hut of an old woman, who offered him cooked beans that were locally grown. In memory of her hospitality, the king is believed to have named the place as '*benda kaala ooru*' (town of boiled beans) (Rice 1897a). While this is an interesting story, it is unlikely to be true as there are references to the name seen much prior to the supposed date of this incident, in a nineth century war memorial stone inscription (Annaswamy 2003; Sudhira et al. 2007). Kamath (1990) offers an alternative explanation also linked to the local ecology, stating that Bangalore is said to have got its name from *benga*, the local Kannada language term for *Pterocarpus marsupium*, a species of dry and moist deciduous tree, and *ooru*, meaning town. Thus, the ecological character of the surrounding landscape appears to be closely linked with the very name of the city.

The founding of modern Bangalore is attributed to Kempe Gowda, a scion of the Yelahanka line of chiefs, in 1537 (Kamath 1990). Kempe Gowda is also credited with construction of four towers along four directions from the Petta, the central part of the city, to demarcate the boundaries of anticipated growth. The city has substantially surpassed these boundaries envisaged in the sixteenth century, with particularly rapid growth in the past half century. Since 1949, Bangalore has grown spatially more than ten times (see Fig. 7.1) (Sudhira et al. 2007).

Bangalore's prominence as a center of trade and commerce was established during the early nineteenth century, when the city supported a flourishing trade and commerce (Buchanan 1870). In the past two decades, Bangalore's economic growth, due to information and communication technology (ICT) industrial expansion, has placed the city on the global map (Friedman 2005). Bangalore's economic growth is powered by the presence of numerous higher education institutions, public sector companies and knowledge based industries (Glaeser 2011).

Despite the popular perception of the economic dominance of the ICT-based industries, the city's economy is highly diversified, being also characterised by textile, automobile, machine tool, aviation, space, defence, and biotechnology based industries, as well as numerous services, trade and banking activities. An important feature of the economic activities of Bangalore is the huge concentration of small and medium enterprises (SMEs) in diversified sectors across the city, with more than 20 industrial estates/areas comprising large, medium and small enterprises (Sudhira et al. 2007). The city is a source of wealth for many, with a net district

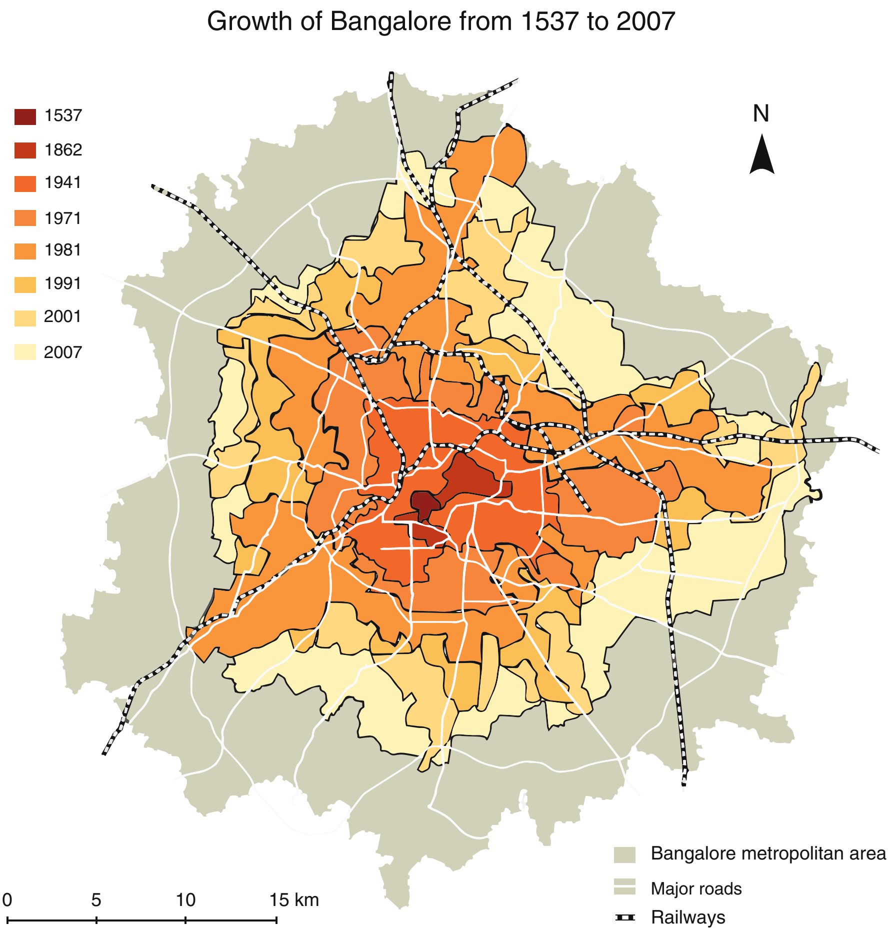

Fig. 7.1 Spatial growth of Bangalore from 1537 (*red*) to 2007 (*light yellow*) (Source data from Census of India. Prepared by H.S. Sudhira and modified by Jerker Lokrantz/Azote. Published with kind permission of ©H.S. Sudhira 2013. All Rights Reserved)

income of Rs. 2,625,920 lakhs (approx. US$ 5.8 billion) and a per capita income of Rs. 39,420, little more than twice of the state's average per capita income of Rs. 18,360 (Government of Karnataka 2005). Yet, economic inequities remain strong with prevailing housing poverty for an estimated 25 % of the city's population.

7.2 Urbanization, Ecosystem Services and Biodiversity; Scenarios and Trends

The economic growth in the city has had a major impact on ecosystems and biodiversity. Bangalore attracts a high traffic of migrants each year, and many of them indicate that they come to the city in part for its cool climate and greenery

(Nair 2005; Sudhira et al. 2007). Yet ironically, while the city was once known for its wide tree-lined avenues, historic parks, and expansive water bodies, this influx of growth has led to the encroachment and pollution of water bodies, the felling of thousands of trees, and large scale conversion of open areas and parks into commercial, industrial and residential settlements (Nair 2005; Sudhira et al. 2007; Nagendra and Gopal 2010, 2011). This is not a new phenomenon, and urban expansion has led to the disappearance of some patches of the city's iconic oldest botanical gardens even as far back as the early nineteenth century (Iyer et al. 2012), but the scale of impact has exploded in recent decades.

Further, urban growth in Bangalore – in common with many other Indian cities – is much less directed by state policies or colonial legacies than for many other parts of the world. This has resulted in patterns of growth that are irregular and complex, with reduced urbanization in the city core, but accelerated and fragmented processes of change at the periphery (Schneider and Woodcock 2008; Taubenböck et al. 2009). High land prices and scarcity of land in the city centre have led to the location of most new development at the city periphery (Sudhira et al. 2003; Shaw and Satish 2006). Consequently, patterns of change in green areas appear to be similarly constrained, with increased loss in vegetation and fragmentation in the city periphery compared to the city core (Nagendra et al. 2012). Urban expansion has also greatly transformed the land-use patterns and institutional forms of governance of many ecosystems located in former agricultural hinterland areas (Nair 2005; D'Souza and Nagendra 2011). The consequences of these combined changes in ecosystems, land use and governance have been manifold, with deterioration of biodiversity and soil quality, aggravation of urban heat island effects, increased pollution, flooding, water scarcity and epidemics, and consequent impacts on human health and well-being.

Bangalore contains a diversity of green spaces located within multiple land use categories including in parks, home gardens, office complexes, wooded streets, wetlands, and remnant forests (Sudha and Ravindranath 2000; Nagendra and Gopal 2010, 2011). Vegetation in the city core tends to harbour greater heterogeneity and species richness as well as a larger proportion of exotic species compared to rural and forested areas (Issar 1994; Neginhal 2006). Within the central, older parts of the city, few patches of remnant natural vegetation exist, and most ecosystems have been significantly modified by human influence, responding to social preferences that vary across location and time (Sudha and Ravindranath 2000). For instance, older wooded streets and parks tend to be dominated by large-canopied, slow growing long lived tree species that provide greater shade, biodiversity support, pollution reduction and microclimatic buffering while recent planting has focused more on ornamental species and short statured, small canopied, relatively short lived species that are easier to maintain but less likely to provide the same range of environmental and ecological services (Nagendra and Gopal 2010, 2011). Bangalore has a relatively high tree diversity but relatively low tree density compared to many other cities (Nagendra and Gopal 2010). As with other parts of the world, home gardens in Bangalore are rich in plant diversity but in contrast to cities in Europe and the USA, these tend to contain a large proportion of plants selected for their cultural, medicinal and culinary properties (Jaganmohan et al. 2012). The city flora contains a large proportion of exotic species, with three out of four park trees coming from introduced species (Nagendra and Gopal 2011). Many of these species have however been planted for well over a century, with the view to creating a spectacular, scenic urban landscape with a succession of species flowering across all seasons. Many of the introduced species thus support a wide diversity of birds, insects and other fauna (Issar 1994; Neginhal 2006).

Trees in Bangalore provide a diversity of ecosystem services, decreasing ambient air temperatures in the summer by 3–5 °C, and road asphalt surface temperatures by as much as 23 °C, substantially reducing levels of noxious air pollutants including SO_2 and Suspended Particulate Matter (unpublished data), providing critical habitat for a diversity of birds, insects and other urban wildlife, and constituting important sources of recreational and sacred cultural services to city residents, especially in poor neighborhoods such as slums (Gopal 2011). Thankfully, despite the extensive clearing and fragmentation of vegetation in many parts of Bangalore, the city core still supports substantial vegetation. Given the city's colonial history as a former British military establishment, many of Bangalore's large green spaces are managed by a variety of public institutions including the military and defence establishments, public sector industries, and educational institutions (Nagendra et al. 2012). Bangalore also hosts two large, historic botanical gardens, a number of educational institutions and a number of historic cemeteries (Nair 2005). These locations harbour large numbers of majestic, visually spectacular trees that provide important biodiversity and environmental/ecosystem services to the city and have provided significant protection against vegetation clearing and fragmentation in recent years (Nagendra et al. 2012). A faunal checklist compiled in 1999 documented the presence of 40 species of mammals, over 340 species of birds, 38 species of reptiles, 16 species of amphibians, 41 species of fishes and 160 species of butterflies within a 40 km radius from the Bangalore city centre (Karthikeyan 1999). Pockets of native vegetation cover persisting in academic institution campuses and botanical gardens contribute significantly to faunal biodiversity. There is high insect diversity, with reports of rare species in the campuses of Bangalore University, the Indian Institute of Science and University of Agricultural Science in Bangalore, as well as in the city's two botanical gardens – Lal Bagh and Cubbon Park (Gadagkar et al. 1997; Kumar et al. 1997; Nayaka et al. 2003; Varghese 2006; Swamy et al. 2008). There has even been a report of the discovery of a new ant species in an Indian educational institution campus (Varghese 2006).

A number of small neighbourhood parks have also come up in the core of Bangalore in the last two decades. Their size and focus on exotic, landscaped features does appear to provide limitations in terms of the range ecological services they can provide (Nagendra and Gopal 2011). Yet, even these parks provide important recreational services for local neighbourhoods, and field surveys indicate that these constitute important habitats for migratory birds and other local biodiversity (Swamy and Devy 2010). In these and other human-impacted urban habitats in Bangalore, some taxa such as ants and earthworms are able to persist because of the availability of specialized microhabitats, leaf litter and soil organic matter (Kale and Krishnamoorthy 1981; Swamy et al. 2008), while other taxa such as lichens indicate patterns of species turnover and replacement by species more tolerant to pollution (Nayaka et al. 2003). Yet, worryingly, fragmentation of vegetation connectivity has increased over time, within the city core as well as the periphery (Nagendra et al. 2012). This is especially critical for the Bangalore urban landscape, where many migratory birds and other species need to move between a number of small, scattered habitats of variable resource quality, making their survival and persistence especially challenging.

Although satellite remote sensing indicates the presence of recent greening in some peripheral areas, field research indicates that much of this can be traced to the short term plantation of water hungry, fast growing exotic timber species such as *Acacia auriculiformis* and *Eucalyptus* on formerly agricultural lands, which are eventually converted to urban development (Nagendra et al. 2012). Fortunately, some peripheral parts have witnessed citizen-government interactions efforts at plantation and restoration of urban environments, and are beginning to provide a positive example for other peripheral areas in the city (Nagendra 2010).

The expansion and intensification of Bangalore has also transformed the land-use patterns and institutional forms of governance of the city's wetlands and water bodies. There were once thousands of reservoirs in the area surrounding Bangalore, used for a number of purposes including agriculture, fishing, cattle washing, drinking, and domestic uses (Buchanan 1870). These water bodies were largely created and maintained by human effort, through damming rainfed streams to create networks of freshwater reservoirs topographically distributed throughout the region (Rice 1897a, b). These wetlands supported an impressive diversity of birds, fish, amphibians, reptiles, insects, and micro-organisms until quite recently (Krishna et al. 1996). Originally managed by adjacent village communities, lakes in Bangalore are now managed by a large number of government departments with overlapping jurisdictions and responsibilities (Gowda and Sridhara 2007; D'Souza and Nagendra 2011). Public perceptions and uses of lakes have also transformed as a consequence of urbanization, from community spaces valued for water and cultural services, to urban recreational spaces used largely by joggers and walkers (Srinivas 2004; D'Souza and Nagendra 2011).

Currently, over 200 lakes are located within greater Bangalore (BBMP 2010), while a much larger network of lakes surround the city at its periphery (Fig. 7.2). Rapid changes in land use have taken place around lake and wetland areas (D'Souza and Nagendra 2011) as water bodies have been encroached upon for conversion to urban land use, subjected to drying because of disruptions in the drainage networks, and polluted by domestic and industrial waste (ESG 2009).

7.3 Governance and Institutions

An important aspect of a city is how well it plans, manages and administers, activities which form the core part of an urban agenda – governance. Yet, environment hardly becomes a priority in most instances. In Bangalore, there has been an overall emphasis on economic development at the cost of environmental degradation.

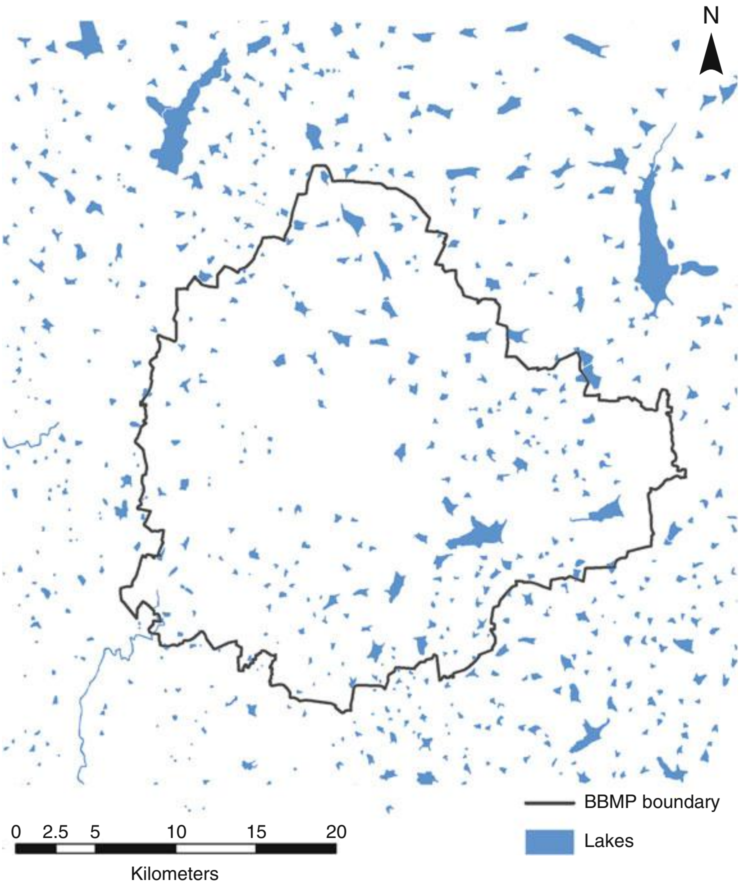

Fig. 7.2 Distribution of lakes within and surrounding Bangalore. Note the lack of lakes in the city center, indicating their encroachment and conversion to other land uses (Prepared by and published with kind permission of © Harini Nagendra 2012. All Rights Reserved)

Appropriate state mechanisms through institutions, policies and programs can enable the protection and maintenance of ecosystems. However, apart from the formal administrative structures, the presence and involvement of civil society significantly shapes the environmental agenda in Bangalore. The city, in the recent past, has witnessed non-governmental organizations (NGOs) and community based organisations (CBOs) taking an active and vibrant role in respect of environmental issues (Sudhira et al. 2007; Khandekar 2008).

A multiplicity of laws and institutions are known to have had control over provisioning of different ecosystem services including protection of lakes. Subramanian (1985) notes that even about a century ago, there were multiple agencies including the Government of Mysore (Public Works Department, Dewan), Government of India

Table 7.1 Institutions and their functional areas concerning the environment in Bangalore

Sl. No.	Institutions	Functional areas
1	Bruhat Bangalore Mahanagara Palike (BBMP)	Urban local body responsible for overall delivery of services – Development and Maintenance of Parks and Playgrounds (all open spaces), Solid Waste Management, Health, Storm Water Drains, Manages only 4 lakes currently, Responsible for Tree Cover as well
2	Bangalore Development Authority (BDA)	Land use zoning, planning and regulation within Bangalore Metropolitan Area; Develops and Maintains few Parks; Notification of Green Belts and Valley Zones implicates amount of development
3	Bangalore Water Supply and Sewerage Board (BWSSB)	Drinking water – pumping and distribution, sewerage collection, water and waste water treatment and disposal
4	Lake Development Authority (LDA)	Regeneration and conservation of lakes in Bangalore urban district
5	Department of Forests, Ecology and Environment	Formulation of programmes in the state on activities causing impact on ecology and environment, Responsible for setting up LDA
6	Bangalore Urban Division, Karnataka Forest Dept.	Has the key role in maintaining the green cover in and around the city, task of planting trees and increasing tree cover
7	Karnataka State Pollution Control Board- KSPCB	Responsible for enforcing various acts and rules concerning the Environment, monitors for air pollution, water pollution, solid waste (municipal, bio-medical and hazardous) disposal, and noise pollution. Also responsible for conducting public hearing in accordance with the EIA notification for any major projects that can potentially have environmental impacts

(Engineering Department, Secretary of State), Resident at Bangalore, Municipalities of the City and the Civil and Military station, and the Military Department, who were all involved in the water supply system, resulting in considerable delays in decision-making. The situation prevailing is no different today, with the involvement of the Bruhath Bangalore Mahanagara Palike (BBMP), Bangalore Development Authority (BDA), Bangalore Water Supply and Sewerage Board (BWS&SB), Lake Development Authority (LDA), Karnataka State Pollution Control Board (KSPCB), Department of Major and Minor Irrigation, Fisheries Department, Karnataka State Council for Science and Technology (KSCST), Agenda for Bangalore Infrastructure Development (ABIDe), Ministry of Environment and Forests, Government of India (MoEF), and Department of Science and Technology (DST), Government of India, all playing considerable roles in the various facets of ecosystem services in Bangalore (Table 7.1). However, the 74th Constitutional Amendment Act by the federal government mandates only the urban local body for protecting the environment.

With a prevailing political vacuum and the clear state-capture of the urban local body, the City Corporation appears to be in no position to re-assert its position in managing environmental issues (Sudhira 2008).

Formally, among the abovementioned institutions, the environment agenda is seen as primary only by the Karnataka State Pollution Control Board (KSPCB) and the Lake Development Authority (LDA). The KSPCB was initially set up under the provisions of the Water (Prevention and Control of Pollution) Act passed in 1974, and it has been extending the responsibility for enacting the Air (Prevention and Control of Pollution) Act, 1981 and Environment (Protection) Act, 1986. Thus, today, KSPCB is responsible for enforcing various acts and rules concerning the Environment. In the context of Bangalore, KSPCB monitors air pollution, water pollution, solid waste (municipal, bio-medical and hazardous) disposal, and noise pollution. KSPCB is also responsible for conducting public hearings in accordance with the Environmental Impact Assessment notification for any major projects that can potentially have environmental impacts. Since the jurisdiction of KSPCB extends throughout the state its focus and attention on environmental issues within the city of Bangalore is not commensurate to the importance of the problems and issues.

In a move to establish formal institutional structures towards the management of lakes and water bodies, and taking cognizance of the N. Lakshmana Rau Committee (1987), the Government of Karnataka set up the Lake Development Authority (LDA), as a registered society under the provisions of the law (Karnataka Co-operative Societies Registration Act, 1959), after the Government Order No. FEE 12 ENG 2002, Bangalore, dated 10th July 2002, as a non-profit organisation working solely for the regeneration and conservation of lakes in and around Bangalore city. LDA initially developed 5,00,000 in Bangalore using funding from the National Lake Conservation Program and later extended its jurisdiction to all major water bodies in Bangalore.

Recent attempts by LDA in a few lakes to explore public-private partnerships have also been extremely controversial, leading to uncontrolled disruptive activities such as motorized boating in some lakes, which have been effectively challenged by litigations by local individuals, civic activist groups and non-governmental organizations (Khandekar 2008). These litigations, including a series of long term efforts by the Environment Support Group, have been extremely successful in garnering support from the Karnataka High Court. In a series of extremely progressive rulings, the Court has intervened on a number of aspects relating to conservation and protection of lakes (High Court of Karnataka 2011). Coupled with recent efforts by the city government to fence, protect and rejuvenate many of the city's urban water bodies, with significant funding earmarked for these initiatives, these series of actions are beginning to reshape Bangalore's polluted wetland and lake environment and ecology. Despite expensive government restoration projects, many lakes continue to be degraded, encroached, silted, and contaminated by sewage. In contrast, a number of government efforts at lake restoration conducted in collaboration with local communities have resulted in the significant recovery of formerly polluted lakes, increase in ground water tables, and increase in native faunal and floral biodiversity (Nagendra 2010). Although some of these community efforts have been rightly critiqued for their exclusion of specific socio-economic and livelihood groups (Sundaresan 2011), others hold significant potential for the development of similar restoration efforts in other parts of the city.

7.4 Urban Dynamics and Future Development

Bangalore's growth in the last few decades is particularly compelling as the city has, over two centuries, been continuously adapting from being a centre of trade during the early nineteenth century (Buchanan 1870) to a Network City (Heitzman 2004). While the city continues to evolve spatially, socially, economically and culturally at a rapid pace; for the ecological systems, these are too fast to adapt in such short time spans. The consequences are glaring: the change in land-use with loss of water bodies and green cover are evident. Clearly, the trend is not encouraging.

Transportation is another major factor impacting environmental sustainability in Bangalore, which has a vehicle population of about 3.8 million for a population of 8.5 million (Regional Transport Authority 2012) – a vehicle-to-person ratio that is far higher than other Indian cities. Much of this can be explained by the lack of sufficient public transport. CO_2 emissions from the road transport sector in Bangalore have been estimated at 24 million tonnes for the year 2005–2006, with emissions from other greenhouse gases, CH_4 and N_2O, estimated at 325 and 19 tonnes respectively. The study suggests that the total emissions projected for 2012 and 2017 would be 3.03 and 4.06 MT of CO_2 equivalents respectively (Greenhouse Gas Inventory of Karnataka 2007). In recent years, there have been efforts to address this by improving the public bus service system, and building a Metro. Bangalore is also embracing non-motorized transport, through a number of community-led cycling initiatives, and efforts by the government such as the ongoing implementation of a 45 km network of bicycle friendly streets within a residential neighbourhood, and support to a recently launched campus-based bicycle sharing system in an educational campus. These efforts show promise as alternatives to encourage Bangalore's citizens to use non-motorized modes of transport for shorter trips, and to reduce the city's environmental and ecological footprint.

The dynamics influencing land-use change resulting in urban sprawl and loss of water bodies and vegetation can be explained by adapting a causal loop diagram initially developed to explain the drivers of urban sprawl in Bangalore (Sudhira 2008). In Bangalore, master planning is the primary instrument that determines the land-use policy. Specifically, in the course of master planning exercise, one of the prominent measures to limit growth is by way of enforcing building height restrictions formalised through the floor area ratio (FAR) and corresponding zoning regulations tied to different land-use categories.

The analysis on land-use change resulting in urban sprawl suggests water dynamics is a crucial balancing feedback that can limit the outgrowth. In the prevailing scenario for Bangalore, wherein water is primarily sourced about 100 km away over a gradient of about 100 m for about 1,000 MLD (million litres per day), there is also

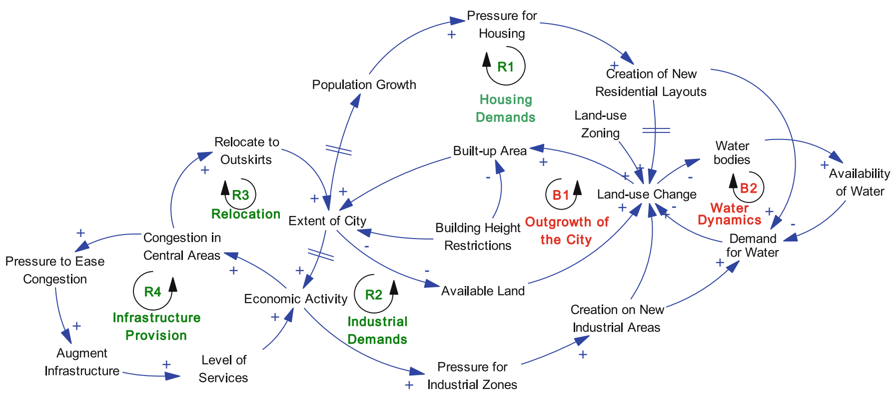

Fig. 7.3 Causal loop diagram for urban sprawl and resulting water dynamics (Prepared by and published with kind permission of © H.S. Sudhira 2013. All Rights Reserved)

huge energy costs involved in pumping this to different parts of the city. The piped water supply does not meet the demand for water, and many parts of the city depend on ground water extracted from borewells and supplied through private tankers, and over extraction of ground water, coupled with the shrinking of waterbodies and conversion of many open areas to impervious urban surfaces have led to alarming levels of depletion in the ground water table. This has a particularly severe impact on poor settlements, whose inhabitants are unable to afford the high costs of privatized water supply. With significant land-use changes in the recent past resulting in loss of water bodies, primarily, prevalence of water bodies and availability of water is going to be a crucial factor for the growth of the city. Accordingly, the key variables identified were: population growth, economic activity, pressure for new housing and industrial areas, land-use zoning, available land, availability of water, built-up area, water bodies, level of services and building height restrictions. The corresponding causal loop diagram is shown in Fig. 7.3.

The six feedbacks generated in this system were:

(i) Reinforcing feedbacks:

(a) Housing demands (R1)
(b) Industrial demands (R2)
(c) Relocation (R3) and
(d) Infrastructure provision (R4)

(ii) Balancing feedback:

(a) Outgrowth of the city (B1)
(b) Water dynamics (B2)

The reinforcing feedbacks, R1 and R2, quite naturally for most cities, set the demand for development of land into residential layouts/apartments or industrial estates, respectively. As the city grows, there is congestion in central areas forcing residents to relocate in the outskirts. This is captured by the reinforcing feedback, R3. The other reinforcing feedback, R4, for infrastructure provision suggests that augmenting infrastructure in a congested area can be counter-productive. Indeed this insight is well established by the classic Braess paradox (Braess 1968). This was demonstrated by adding a link within the network to ease congestion, which would be counter-intuitive in a congested road network. Reinforcing feedbacks often generate exponential growth and then collapse. Systems that are self-regulating or self-correcting have been provided with balancing loops. The balancing feedback, B1, results in the outgrowth of the city through the process of land-use change limited by the availability of land for development. The other balancing feedback, B2, rise in demand for water and land-use change affects water bodies and will be limited by availability of water. The other control variable as envisaged by the land-use planning is land-use zoning and building height restrictions, which suggests that it influences the extent of outgrowth and built-up areas negatively.

Bertaud and Brueckner (2005) have shown that in unrestrictive building height conditions, the extent of the city spread is contained with higher densities in central areas. Further, the BangaloreSim model (Sudhira 2008) that was primarily developed as a spatial planning support system was used to generate scenarios for 2025. The model was used to generate forecasts of land cover for 2025 with higher FAR and siting of growth poles at key economic activity centres. The simulations revealed an increase of built-up areas (417 km^2) amounting to 31.55 % of the land cover and the water bodies declined to less than 3 km^2 with about 2.26 % of the land cover. The simulations also suggest a trend towards an increase in high-density built-up and decrease in low-density built-up areas. The increase in built-up areas with high densities suggests more congestion in certain parts of the city. While this exercise was useful in generating a scenario of urban land-use, there can be many more scenarios generated by undertaking the simulations for different policy settings to understand their respective consequences. However, as noted earlier, the reduction of water bodies is of particular concern.

With the current trends of urban growth Bangalore is experiencing, the forecast for 2025 gives pointers on possible implications despite several limitations in the model, including limited policy levers for testing. However, on the one hand, the increase in built-up areas and reduction of water bodies seem like a matter of concern, on the other hand the increase of high density and decrease of low density built-up (or sprawl) can be of some consolation.

7.5 Concluding Remarks

Further research is critically needed to understand how the larger climatic and ecological context influences urban ecosystems and ecosystem services in Bangalore (D'Souza 2011). In the context of increasing global warming, while the Indian monsoon has remained relatively stable, an increasing frequency of extreme rain events has been noted, ranging from floods to droughts (Goswami et al. 2006).

The fragmentation of green spaces and of lake connectivity, coupled with extreme rainfall events, is likely to result in an increase in the frequency of droughts as well as flooding in urban areas. Vulnerable socio-economic settlements, in particular slums, are likely to be especially affected by this double whammy of urbanization and climate change (D'Souza 2011). Women and children in slums in particular have to deal with the consequences of lack of access to clean water, lack of sanitation, and consequences in the form of increased susceptibility to infectious diseases in Bangalore. Adaptation measures, particularly those of ecosystem restoration and rejuvenation, can help to mitigate some of these impacts, but unfortunately these have been largely ignored by planners so far (Sudhira 2008). Recent studies have indicated that greenery and plants play an extremely significant role in the lives of slum residents in Bangalore, providing critical social, cultural, religious medicinal and food related ecosystem services (Gopal 2011). Thus, city planning needs to take into account the needs of vulnerable socio-economic populations, which are especially susceptible to the short term and long term impacts of urbanization including loss of green spaces, microclimatic variations, air and water pollution, flooding, lack of sanitation and epidemics (D'Souza 2011; Gopal 2011). Unfortunately, such approaches are currently missing in Bangalore and community action has largely been successful in engaging with city planners and administration only in wealthier localities, sometimes resulting in the further exacerbation of existing inequities and vulnerability (Ranganathan 2011; Sundaresan 2011).

Bangalore is known for its strong and active community networks. Thus a number of individual citizens, resident associations, civil society networks, naturalist groups, and citizen science and educational programs have also been influential in promoting awareness of, and organization around issues related to biodiversity and conservation in the city (Sudhira et al. 2007; Nagendra 2011). The environmental activist group "Hasiru Usiru" (loosely translated as "Greenery is Life") has been very active in Bangalore since their formation in 2005, with activities ranging from green awareness to protests against tree felling and urban governance. Social networks such as Hasiru Usiru have contributed substantially to keeping issues of urban conservation in the forefront of public awareness in recent years, in addition to initiating a number of Public Interest Litigations that have resulted in influential court rulings on issues of tree felling (Sudhira et al. 2007; Enqvist 2012). The bird-watching community in Bangalore, comprising mostly of amateurs with a few experts, has been very active, meeting at least once every month for over two decades, with an active email discussion group, "bngbirds", hosted at yahoo.com (a similar group exists on butterflies that also uses mailing lists and discussion groups to communicate and organise). Since the past 5 years, there has also been an annual Bird Race, with participants having cumulatively logged over 230 species of birds in and around Bangalore in a single day, and helping to provide systematic data that can eventually aid in long term assessment (available at http://birdrace.dhaatu.com/bangalore). Other popular citizen science initiatives that have been primarily developed and housed in Bangalore, though applied at wider geographic scales, are the 'Migrant Watch' program (http://www.migrantwatch.in/), and the Citizen Sparrow program, (http://citizensparrow.in/).

As urbanization poses greater challenges, compounded by effects of urban primacy on Bangalore, it remains to be seen how the city adapts and evolves in decades to come. Eventually, the sustenance of the city will depend on how effectively it retains its natural resources, particularly green spaces and water bodies. In this context, Bangalore's cultural character as a location of significant civic and collective action will play a very important role in shaping urban environmental protection and conservation efforts, with collaborations between citizens and government agencies playing an increasingly critical role in placing pressure on, negotiating with and shaping government responses to planning for ecosystem management and conservation (Sudhira 2008; Khandekar 2008; Nagendra 2010, 2011; Ranganathan 2011; Sundaresan 2011).

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Chapter 8
Local Assessment of Tokyo: Satoyama and Satoumi – Traditional Landscapes and Management Practices in a Contemporary Urban Environment

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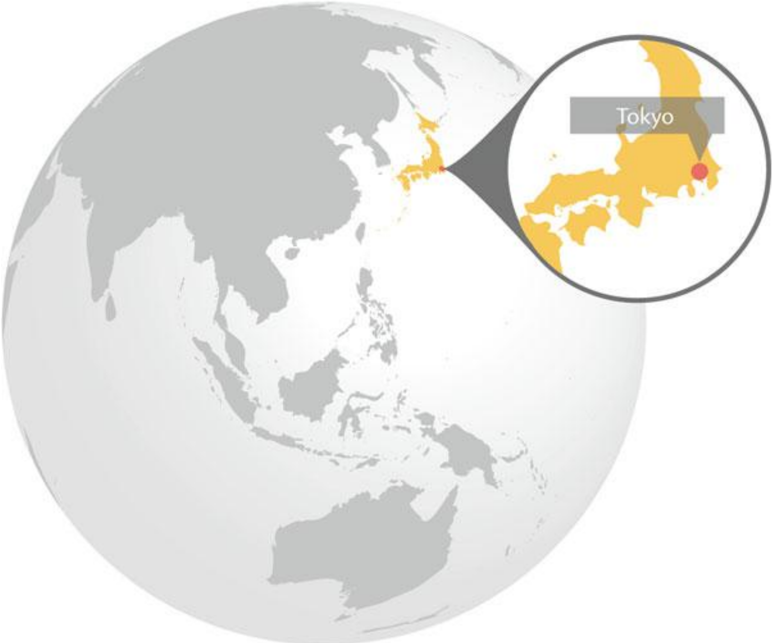

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Abstract This assessment explores the Japanese concepts of satoyama and satoumi (land and coastal), as possible strategies for sustainable management and governance of common urban ecological resources. Satoyama and satoumi are described as landscape types, and management approaches to land and coastal areas that build on a mosaic composition of ecosystem types and their inherent interlinkages. The management practices and the rich biodiversity of the landscapes are thus mutually interdependent. It is acknowledged in the assessment that local governments play a critical role for the management of urban ecosystems and conservation of biodiversity, which is especially important in the face of the unprecedented urban growth currently ongoing globally. This assessment provides an overview of the urbanization trends in Japan, with related challenges to ecosystem provisioning, and the opportunities for sustainable management that a satoyama and satoumi approach can present. Some international examples of ecosystem management that in different ways can inspire transformation of governance structures in Japan to support urban satoyama and satoumi are highlighted.

Key Findings

- Satoyama and satoumi landscapes are biodiversity-rich landscapes in peri-urban to rural areas, and contain a mosaic of ecosystems sustained by human management.
- Extensive rural-to-urban migration and expanding cities in Japan have caused a decrease of use of the natural resources, resulting in a degradation of the quality and quantity of satoumi and satoyama landscapes.
- Implementing satoyama and satoumi in urban areas can support cultural, provisioning and regulating ecosystem services, thereby also well-being for humans and ecosystems alike.
- Current urban green space planning policies do not sufficiently take the satoyama and satoumi concepts into account, and have mostly focused on national and regional levels, whereas support from both local governments and communities is key for implementation and long-term management.
- Two elements are argued to support implementation and long-term management of urban satoyama and satoumi: an increased involvement of local communities in the policy-making and management of the urban green areas, and incentives to make urban satoyama and satoumi financially profitable.

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8.1 Introduction

The values of urban ecosystems for human well-being are increasingly recognized in research, although they may not always be recognized by the general public (see for example Mougeot 2006; Vandruff et al. 1995; Bolund and Hunhammar 1999; Berkowitz et al. 2003) (see also Chaps. 11 and 27). In Japan, the interdependence between traditional activities and natural environments is the foundation of Japanese culture. Following this, cultural services such as educational and social services are increasingly appreciated by urban residents.

Satoyama (land) and satoumi (coastal) are Japanese concepts that refer to long-standing management traditions based on the symbiotic interaction between ecosystems and humans (Duraiappah et al. 2012). Satoyama and satoumi landscapes are defined as “a dynamic mosaic of managed socio-ecological systems producing a bundle of ecosystem services for human well-being” (Saito and Shibata 2012, p. 26).

The four primary characteristics of these landscapes are: (1) satoyama is a mosaic of both terrestrial and aquatic ecosystems comprised of woodlands, plantation, grasslands, farmlands, pasture, irrigation ponds and canals, with an emphasis on the terrestrial ecosystems; (2) satoumi refers to aquatic ecosystems and is a mosaic of both terrestrial and aquatic ecosystems comprised of seashores, rocky shores, tidal flats, coral reefs, and seaweed/grass beds; (3) satoyama and satoumi landscapes are managed with a mix of traditional knowledge and modern science; and (4) biodiversity is a key element for the resilience and functioning of satoyama and satoumi landscapes.

Satoyama and satoumi are primarily found in rural and peri-urban areas and satoyama alone is estimated to cover 40 % of Japan’s total landmass, in some prefectures reaching up to 58 % (Ministry of the Environment, Japan 2001). The current trend is going towards degradation in quality and quantity of the landscapes: satoyama primarily due to land usage change, i.e., conversion of farmland and forests to built-up urban areas; and satoumi due to overfishing, pollution, and changes to the physical environment caused by intentional development or unintentional human activities of coastal areas, such as land reclamation, port constructions and coastal embankments. However, transferring satoyama and satoumi to cities holds the potential to maintain these landscape management approaches, while at the same time provide effective ways to meet some of the challenges of the growing cities. For example, ensuring food security and production of fresh water can be particularly important in developing countries where urbanization rates are high.

The transition of satoyama and satoumi landscapes to urban areas is, however, faced with some key obstacles: (1) a lack of awareness of the potential social and ecological benefits of satoyama and satoumi; (2) a lack of legislative and policy support to control the urban expansion, conserve satoyama and satoumi in the new urban areas, and implement new such landscapes in the already built-up environment; (3) a lack of technological, human, and financial resources for implementation; and (4) a lack of coordination across institutional levels (JSSA 2010a; Ministry of the Environment, Japan 2012).

The next section provides an overview of urbanization and its effects on biodiversity and ecosystem services in Japan, focusing on satoyama and satoumi landscapes located in or nearby urban areas. The text will then explore some approaches to biodiversity governance that manage the changes associated with urbanization around the world, with the aim to assess possible ways to strengthen implementation of an urban satoyama and satoumi approach.

8.2 Urbanization, Ecosystem Services and Biodiversity: Scenarios and Trends in Japanese Cities

The intensive growth of industries during the rapid economic growth period from the late 1950s to the early 1970s in Japan created employment opportunities, led to population concentration, and ultimately urbanization. Particularly the economic globalization of the last decades of the 1900s, created urban concentrations and eventually metropolises on the Pacific coastal zone of Japan, such as Kobe, Osaka, Nagoya, and Tokyo.

The rapidly expanding cities resulted in conversion of farmland to urban land, first along the coastline, then into forests and in-land (Himiyama 2004). In the process, people's traditional ways of making their livelihoods significantly changed (Okuro et al. 2012). In the rural areas, modernization and mechanization of agriculture created monocultures, which decreased biodiversity (Okuro et al. 2012). At the same time, forestry and fisheries collapsed due to the expanding cities and the import of timber and seafood.

Together with the degradation of satoyama management practices, the capacity of the ecosystems to provide crucial services such as nutrient cycling, food and timber production, water retention, and water purification has decreased. In addition to agricultural production, the change in satoyama also affected the ecosystem composition and local culture. From the 1990s, an ageing rural population and shrinking working population within agriculture, forestry and fisheries have both become increasingly apparent nationwide.

At the same time, the changes in land cover from natural landscapes to built-up areas also create challenges for the service providing ecosystems. The population concentration in the Tokyo metropolitan area, which includes the neighboring prefectures around Tokyo (i.e., Kanagawa, Saitama, and Chiba), consequently forms the world's largest city zone with over 30 million people. The Kanto region, i.e., the Tokyo metropolitan area and its three neighboring prefectures: Gunma, Tochigi and Ibaraki, has been exposed to the most rapid and extensive urban development and population growth in Japan. When the urban population increase was the most dramatic, i.e., in the 1960s until the 1980s, the forest cover in the region decreased by 4.3 % (618 km^2) (Saito 2004).

Japan is today facing several changes within cities as well as in the peri-urban areas. Some of the largest cities are expected to decrease in population size, as the country's overall population began to decline in 2005. It is predicted that the urban populations will decrease rapidly and at the same time the mean age will increase, especially in big cities (National Institute of Population and Social Security Research 2012). Securing a workforce to maintain the production of ecosystem services has become difficult not only in rural satoyama but also in satoyama located in or close to the metropolitan areas, (i.e., the cities and the neighboring prefectures).

However, overconcentration in the center of cities is expected to continue in some metropolises like the Tokyo metropolitan area. At the same time, population numbers in the peri-urban areas around the city centers have dwindled due to depopulation and migration from the crowded city centers to peripheral areas. As the population in the city centers decrease, the urban sprawl increases.

8.2.1 Changes and Feedbacks in Ecosystems

The decrease of land and marine resources management in Japan, has caused a degradation of products provided by satoyama and satoumi such as timber, crops and fish. This, in turn, has caused degradation of other types of ecosystem services including supporting, regulating, and cultural services (Duraiappah et al. 2012).

The changes in ecosystem services provisioning in areas outside and around metropolitan areas can come to increase the effects of other expected changes, as they reduce the natural disaster buffer capacities. Climate changes are expected to increase the risks of extreme weather phenomena and related disasters in Japan, and the metropolitan areas like Tokyo and Osaka are expected to experience the biggest changes. Heat island effects are already advancing, and extreme weather phenomena such as guerrilla downpours and intense heat, occur frequently.

As the quality of the rural land decreases, so does the biodiversity, and thereby also the resource base that supports the cities with bundles of ecosystem services. These changes have already led to increased frequencies of floods and threats to the availability of indispensable high-quality fresh water of upstream regions.

Satoyama landscapes in peri-urban areas have been faced with new challenges including fragmentation of landscapes, loss of mosaic land use, increase of alien species like *Solidago altissima* and common raccoon (*Procyon lotor*), and loss of traditional knowledge to manage the landscape. The loss of traditional knowledge is particularly significant. In a study by Shimada et al. (2008), four past and present management models of typical secondary woodlands in satoyama near Tokyo were identified: traditional management, non-traditional management, ad hoc management, and no management. They found that traditional management maintained relatively higher species richness with little variation between surveyed plots, compared to non-traditional management models.

Japan, as with many countries around the world, imports a substantial proportion of its consumed resources. This pattern of reliance on resources from around the world, to a large extent driven by demands by urban dwellers, has strongly contributed to degradation of ecosystem services in rural areas globally, such as soil loss, ground drainage, reduced amounts of carbon storage because of deforestation, and global biodiversity loss. The role of urban areas as strong contributors to global environmental problems needs to be acknowledged. It is, however, also crucial to acknowledge and realize the potential of cities to act as areas for adaptation and mitigation strategies and as hubs for innovations (Chap. 33).

8.3 Governance and Institutions

Although policies and ordinances to conserve and maintain urban green spaces for multiple ecosystem services, such as food security, local climate regulation, and biodiversity conservation exist, they are not necessarily associated with the concept of satoyama and satoumi. This part of the assessment intends to address this gap and use existing examples from other countries to facilitate the discussion of how good governance of satoyama and satoumi at a local or city level in Japan may be designed.

An assessment aimed to provide policy-makers with scientifically credible information on the values of ecosystem services provided by satoyama and satoumi for economic and human development, the *Japan Satoyama Satoumi Assessment* (JSSA), was initiated in Japan in 2006. The final report was published in 2012 (Duraiappah et al. 2012). The JSSA primarily focused on Japan on a national and regional scale, but also included a local urban and peri-urban scale. In addition to discussing changes in ecosystem services, the JSSA also covered institutional mechanisms, socio-economic challenges, public participation, and associated appraisal of biodiversity governance.

The JSSA shows how satoyama and satoumi emphasize sustainable use of ecosystems in order to support the provisioning of ecosystem services for human well-being. However, in practice, the general ecosystem management and governance discussions have thus far mostly focused on regional and local levels, whereas the discussions on satoyama and satoumi in the JSSA focused on national and subnational levels. There is thus a need to develop strategies that address local management and governance of urban satoyama and satoumi in Japan.

Based on the findings in the JSSA, four aspects that may facilitate the development of local governance in support of urban satoyama and satoumi are discussed: (1) effectiveness of biodiversity policy; (2) coordination between development and biodiversity and ecosystem services conservation; (3) available financial mechanisms to support long term implementation; and (4) capability to build up partnerships and encourage participation. Furthering the discussion on strategies that may improve the dynamic balance between societies and ecosystems in cities, this assessment draws upon examples from Nagoya, Japan; eThekwini/Durban, South Africa; and the UK (Europe).

8.3.1 Strategies for Effective Biodiversity Conservation Policies

Biodiversity conservation is a national goal in Japan as clearly defined in the *Japanese National Biodiversity Strategies and Action Plan* (NBSAP) (the Fifth Edition has been recently launched on September 2012). The Japanese NBSAP has explicitly taken satoyama and satoumi and urban green spaces into account but separately of each other. However, satoyama and satoumi landscapes, whether rural or urban, are thereby still treated as separate from urban nature and lack conceptual alignment in policies. For example, the mosaic characteristics of forests and agricultural lands make it difficult to categorize the satoyama into one category of land classification. Local ordinances have been proposed by prefectures and cities in Japan to promote conservation, regeneration and utilization of satoyama since 2000 (Takahashi et al. 2012). However, regulations and policies at a local level tend to favor economic growth and development before protection of existing urban satoyama and satoumi. In this context, the loss of natural and semi-natural areas, which are the basis of satoyama and satoumi, is inevitable without effective legal or non-legal binding instruments aimed at protection of the natural landscapes.

Following a segmented governance structure, national policies and laws in Japan are largely separated from the local governance and thus provide little support for satoumi and satoyama, although it is recognized in the national guidelines that an integrated approach is needed for managing such a system in a sustainable manner (Takahashi et al. 2012). Legal responses have been respectively developed to address the need of sustaining satoyama and satoumi landscape at national levels. The conservation of satoyama in Japan is promoted under the newly established Act on the Promotion of Conservation for Biodiversity Activities through the Cooperation among Regional Diversified Actors (enforced in 2011). However, the development of specific legal strategies for conservation of satoumi and by that marine biodiversity, is still only in the early stages. Management of coastal areas is marked by a highly complex web of a wide range of stakeholders from fisheries, construction, nature conservation, and recreation sectors, among others.

An international and seemingly promising method is the *Strategic Environmental Assessment* (SEA), which since 2004 is mandatory under the *SEA Directive* for the member states of the European Union to conduct in selected plans and programs. SEA originates from the National Environmental Policy Act (NEPA) of the USA in 1970, and has since mostly been developed and implemented in European and North American countries (Dalal-Clayton and Sadler 2005). SEA is designed to evaluate the possible accumulative environmental consequences of proposed policies and plans, and thus ensure already in the early stages of decision making that biodiversity support is considered (ICLEI 2010). Although SEA systems vary considerably between countries or even cases, the concept has rapidly evolved and been applied to public plans and programs, such as land use, transport, energy, waste and agriculture, to support sustainable development.

In the past decades, the concept of SEA has been introduced in several East Asian countries, including China, South Korea, Taiwan, and Japan. As in many East Asian countries, the applications in Japan remain limited, but some SEA components have already been implemented as local governments use the SEA to screen environment-related plans and programmes (Dalal-Clayton and Sadler 2005). SEA can provide an opportunity to include satoyama and satoumi approaches into development processes, if the SEA system would be further developed and adapted to a Japanese social-ecological context. For example, the importance of using traditional knowledge for environmental management, as well as the interaction between human maintenance activities and their consequent ecosystem functioning will have to be highlighted and valued in the SEA.

8.3.2 Development That Supports Biodiversity and Ecosystem Services Conservation

Laws, regulations, and policies proposed for green space conservation and the sustainable use of natural resources have shown to some extent to contribute to the maintenance of satoyama and satoumi landscapes (Takahashi et al. 2012). However, implementation and long-term management of satoyama and satoumi in and around cities is complicated by the increasing competition for land in Japan. One of the key challenges to make satoyama and satoumi an integral part of the urban landscape is thus to find strategies that can balance land-use for financial prosperity, with the implementation and conservation of satoyama and satoumi. Another key challenge is to design plans and strategies of which satoyama and satoumi are integral parts.

Although plans and strategies that aimed to include biodiversity and ecosystem services in urban development have been designed, for example a Master Plan for Greenery, their relation to urban satoyama and satoumi is not specified. Their full potential to support satoyama and satoumi is thus still to be realized.

One way to meet the limitations of current plans may be to design more comprehensive development plans that systematically coordinate biodiversity protection and development initiatives. Taking the UK's planning system as an example, the *Biodiversity Supplementary Planning Documents* (BSPDs) are developed in conjunction with local development documents. A BSPD provides explicit guidance to actors such as developers, households and planners on protecting, creating and improving biodiversity during the development process (SCDC 2012). Another systematic planning approach that facilitates biodiversity and development coordination is the South African *Integrated Development Plans* (IDPs) which are mandatory for all local authorities in South Africa to prepare and continue over 5 years. By aiming to coordinate the work between different sectors, such as housing, environments and transportation, the IDPs can provide a cross-sectoral planning mechanism that integrates development with conservation of biodiversity and ecosystem services (eThekwini Municipality 2007). These planning systems take biodiversity and ecosystem services into account at the early stage of local development and stand better chances to influence and to be compatible with other sectoral plans.

The BSPDs and the IDPs represent two top-down planning measures that might facilitate the integration of satoyama and satoumi proposition throughout existing development frameworks and influence sectoral plans. However, both of these measures are relatively new and it is too early to conclude which measure is more applicable in the Japanese context. More studies to address the characteristics of Japanese planning and institutional systems are needed.

In an interesting bottom-up development, on the other hand, a revival movement of satoyama landscapes since the late 1980s, has increasingly drawn public attention and interest to nature conservation and protection. This movement has been especially active in municipalities containing urban satoyama and satoumi, with several tens of thousands municipal inhabitants, located within approximately 50 km from urban centers (Saito 2005). The activities focus mostly on the values of cultural services such as education and recreation. This hints to that supporting cross-scale planning mechanisms where urban inhabitants are involved in planning and management, can promote urban satoyama and satoumi on several levels.

8.3.3 Available Financial Mechanisms to Support Long-Term Implementation

International initiatives to give ecosystem services monetary values have for example estimated the urban plantations in Canberra, Australia, at a combined energy reduction, pollution mitigation and carbon sequestration value of US$20–67 million during the period 2008–2012 (Brack 2002). If satoyama and satoumi can credibly be given a monetary value, this can be an incentive to support the conservation of satoyama and satoumi landscapes in Japan (e.g., TEEB 2011).

Several economic interventions have already been proposed by Japanese local governments to mitigate or reverse the decline in satoyama and satoumi. The interventions include taxation of illegal industrial waste dumping, encouragement of the use of biomass energy from thinning woods of satoyama, and direct payments for stewardship, i.e. management aimed to maintain and conserve the rural natural resources. The implementation of the incentives has, however, been limited due to the decreasing production value from agricultural, forestry and fishery activities (Takahashi et al. 2012).

Voluntary involvement plays an increasingly large role for maintenance of satoyama and satoumi, such as payment for ecosystem services, stewardship sharing with citizens, and certification systems for products (Takahashi et al. 2012). A case in point is the *Greenification Certificate System* initiated in Nagoya, Japan in 2008 (*cf*. Kohsaka 2010). Under the overall framework of the *System of Greening Areas*, the *Greenification Certificate System* serves as a voluntary tool by which private landowners receive lower interest rates on loans from local banks, while conserving or creating green spaces when they develop their land. This experimental tool is expected to encourage more green spaces, including trees, green facades and green roofs, in private owned properties and might be useful for enhancing biodiversity in densely urbanized cities.

8.3.4 Capacities to Build Partnerships and Encourage Participation

As human activities play a vital role in the management of satoyama and satoumi, participation from citizens, local non-profit organisations (NPOs) and non-governmental organisations (NGOs) should be acknowledged by decision makers and in planning as a key component for achieving sustainable management of urban satoyama and satoumi. Involving a wide range of partners, such as government agencies, academia, conservation groups, local businesses, amateur naturalists and private corporations, can play a key role in defining priorities in satoyama and satoumi management and sustain it over time.

One international example of a successful public-private partnership that may serve as inspiration for cities in Japan, is the Buffelsdraai Community Reforestation Project in South Africa. The eThekwini municipality and the Wildlands Conservation Trust together engage local communities to create tree nurseries at their homes and provide tree seedlings for reforestation. By selling seedlings, participants receive credit notes to exchange of food, basic goods and school fees at regular tree stores in participating communities (eThekwini Municipality 2012).

In Japan, Kawasaki City developed a Conservation Management Plan to engage citizens in stewardship activities of designated conservation areas and to reduce the conservation burden in terms of labor and expenditure on landowners. The city also builds a partnership with universities to encourage research on urban ecosystem conservation areas and to open up a dialogue between academics, policy-makers and managers. The capacity to build such cross-level and cross-sectoral partnerships where local inhabitants actively participate in the management of the urban green areas can prove to be critical in terms of creating and sustaining urban satoyama and satoumi (Chap. 27).

8.4 Concluding Remarks

Although satoyama and satoumi are Japanese terms, these landscapes of mosaic ecosystems where human-nature interaction is central are not unique to Japan. Such landscapes are found throughout many regions of the world, though the terminology and managerial philosophy might vary from one area to the other (Duraiappah et al. 2012).

This text has assessed satoyama and satoumi landscapes as possible strategies to support and urban development that builds on conservation of urban ecosystems.

The concepts originally referred to specific landscape types as well as management structures on a regional scale in Japan. As cities are expanding on traditional satoyama and satoumi landscapes, however, the concepts increasingly refer to existing or planned areas in or around cities.

Management of satoyama and satoumi has traditionally built on involvement by entire communities. Translated to urban areas, satoyama and satoumi may thus provide support for inclusive, multi-level management structures as well as management of landscapes. Including the local inhabitants and their ecological knowledge can be a means of supporting and maximizing the ecosystem functioning, as well as ensuring long-term management.

However, several factors challenge both conservation of existing satoyama and satoumi areas and successful implementation of urban satoyama and satoumi landscapes. These challenges include weak legislation where the existing regulations give insufficient support for ecosystem conservation in general, and in addition lacks integration of satoyama and satoumi aspects. The increasingly stiff competition for land in Japanese cities, and even more so in cities in the most rapidly urbanizing countries in the world, pushes land prices up and challenges green area conservation. Finally, climate changes affect the prerequisites for native species to exist, and their inherent balance, thus also affecting entire ecosystems. To highlight possible ways to meet the challenges, the assessment has shown through international examples that entail satoyama and satoumi elements, where local initiatives have contributed to successful ecosystem management.

Although many green initiatives exist in cities such as Yokohama (JSSA 2010b; Sadohara et al. 2011), Nagoya (Nagoya City 2010) and Kanazawa (Ishikawa Prefecture 2011; UNU-IAS 2011), in the long term these obstacles require changes at multiple levels, from governments to societies. Local governments will play a critical role in initiating appropriate changes to favor biodiversity and ecosystem health in the course of urban development. Therefore, a proactive assessment framework that directs local governments to address problems and to monitor performance is needed for further development.

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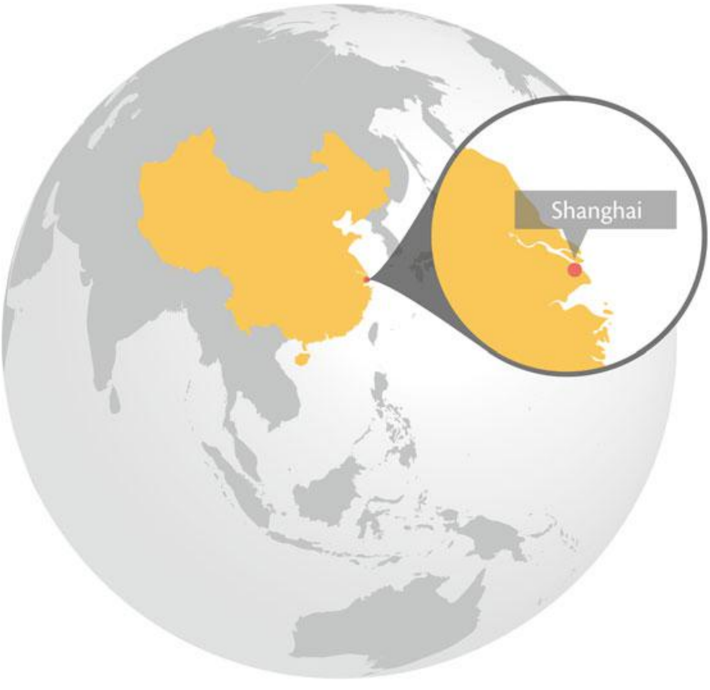
Chapter 9
Local Assessment of Shanghai: Effects of Urbanization on the Diversity of Macrobenthic Invertebrates

Wenliang Liu, Xiaohua Chen, and Qiang Wang



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Abstract Shanghai is the largest industrial and commercial city in China. It is a coastal metropolitan city located on- and surrounded by several types of natural and constructed wetlands. The rapid growth of the city over the past three decades and rapid economic development have caused a number of ecological problems. Macrobenthic invertebrates play an vital role for the wetland ecosystem structure and function in Shanghai. As macrobenthic invertebrates quickly respond to water and habitat quality, they can be key indicators of the state of the wetland ecosystems. However, tidal flat reclamation, alien plant spread, and sewage discharge pollution caused by the growth of the city and the urban population, have changed their habitats and affected their capacities to produce ecosystem services. This assessment discusses the effects of urbanization on macrobenthic invertebrates in Shanghai and measures that may contribute to their conservation. The results show that the growing city has changed the species composition and abundance. They also show that ecological restoration can yield positive results for the macrobenthic invertebrate populations, but is a long process that needs the support of well-designed policies.

Keywords Shanghai • Urbanization • Biodiversity • Macrobenthic invertebrates

Key Findings

- Shanghai rests on tidal flats and is surrounded by natural and constructed wetlands
- Macrobenthic invertebrates living in the wetlands produce crucial ecosystem services and are important as indicators of the state of the aquatic ecosystems
- The expansion of the city, primarily since the 1970s, has changed the macrobenthic invertebrates' species composition and decreased the population abundance
- Ecological restoration is one means to support re-introduction of the macrobenthic invertebrates, but is a long-term process
- Supporting mechanisms need specially designed policies, which today are lacking

9.1 Introduction

Shanghai is a coastal metropolitan city, and the largest industrial and commercial city in China. Natural and constructed wetlands account for 23.5 % of the city's total area (Gao and Zhao 2006). The wetland ecosystems support a diverse array of macrobenthic invertebrates which are important as commercial resources, and as providers of ecosystem services such as water purification, by recycling nutrients, detoxifying pollutants, and dispersion (Gray 1997; Snelgrove 1997).

Shanghai has experienced rapid urbanization over the past three decades, accompanied by rapid acceleration of economic development. The growth of the city has caused a number of ecological problems, including the degradation of air and water quality, alteration of the local climate, a decline in native plant species, and an increase in the numbers of alien plant species (Zhao et al. 2006). Furthermore, tidal flat reclamation by the expanding city, and sewage discharges have led to further degradation of the aquatic natural habitats. This assessment explores the effects of urbanization on biodiversity in Shanghai, focusing on macrobenthic invertebrates, and discusses measures that may contribute to their conservation.

9.2 Shanghai's Demography, Economy, and Geography

Shanghai is a coastal city resting on the estuary of the Yangtze River. It borders the East China Sea in the east, the Yangtze Estuary in the north, Hangzhou Bay in the south and the Jiangsu and Zhejiang Provinces in the west (Fig. 9.1).

Shanghai is the largest industrial and commercial city in China, with its municipal jurisdiction encompassing 17 districts and one county (Chongming Country, including three islands: Chongming, Changxing, and Hengsha Islands). The jurisdiction covers an area of 6,340 km^2, including 6,219 km^2 of land and 121 km^2 of water. Shanghai sits on the alluvial plain known as the Changjiang River Delta, whose foundation was formed in the late Mesozoic Era about 70 million years ago. The Changjiang River deposits large amounts of silt in its estuary, which over time accumulated over 6,000–7,000 years ago to form a growing sand bank. During the Tang Dynasty (618–907) most of the land area that modern-day Shanghai city covers today became dry land. During the Ming Dynasty (1368–1661), land emerged on the eastern bank of the Huangpu River with a coastline that still today remains largely intact. The Changjiang River Estuary in the north has three islands: Chongming, Changxing, and Hengsha. The 1,267 km^2 Chongming Island is the third largest island in China and the largest alluvial island in the world. In 1996, a fourth island called Jiuduansha began to form in the estuary (Han et al. 2010).

With the rapid expansion of the city, the urban land area increased exponentially from 159.1 km^2 in 1975 to 1,179.3 km^2 in 2005 (Fig. 9.2). The slowest annual rate of urban area expansion, 17.7 km^2, occurred between 1975 and 1981; the rate then increased to 52.4 km^2 between 1990 and 1995; and again to 54.9 km^2 between 2000 and 2005 (Zhao et al. 2006). This is consistent with the changes in China's economic policies, since the country began its economic reform in 1978 and accelerated the process in 1992 (Lin 2002).

According to the statistical yearbook of Shanghai in 2011 and 2012, the permanent resident population of Shanghai has reached about 23.71 million. In recent years, the migrant population, i.e. people who come to the city to work but are not registered as Shanghai residents, has grown steadily in peri-urban and suburban areas. In the five districts of Minhang, Jiading, Songjiang, Qingpu and Fenxian, the migrant population has exceeded the registered population. The permanent resident population,

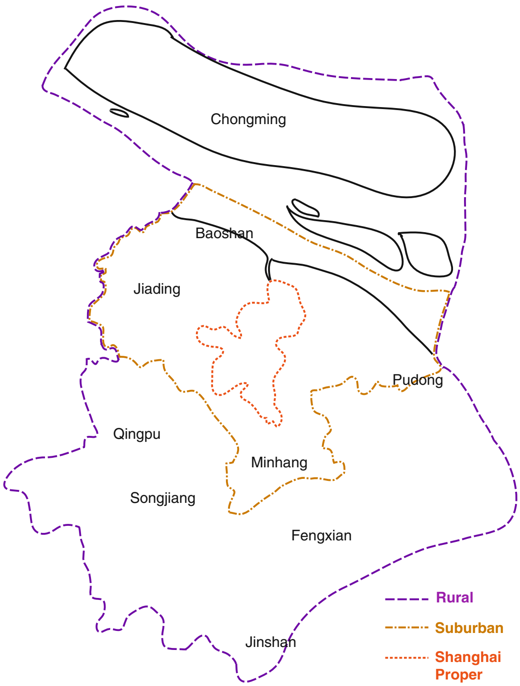

Fig. 9.1 The city of Shanghai and its main districts. The three demarcated areas are from the center and out: Shanghai Proper (*red*), Suburban Shanghai (*yellow*), and Rural Shanghai (*purple*) (Source data from Han et al. 2010. Prepared by and published with kind permission from ©Wenliang Liu 2013) (Color figure online)

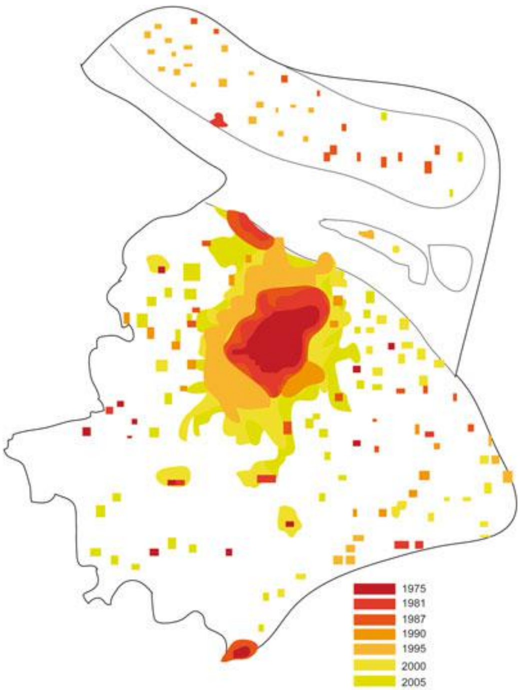

Fig. 9.2 Growth of Shanghai from 159.1 km^2 in 1975 to 1,179.3 km^2 in 2005 (Source data from Zhao et al. 2006. Prepared by and published with kind permission from ©Wenliang Liu)

i.e., the registered population and the permanent migrant population, is expected to increase in the near future, following an expected increase in the number of newborns. For example, in the first three quarters of 2012 the number of newborns among permanent residents was 160,000; this represents an increase by 13,000 compared with the same period the year before.

Shanghai was the largest and most prosperous city in the Far East during the 1930s, and beginning in the 1990s, development surged. The city's per capita GDP (Gross Domestic Product) exceeded US$1,000 for the first time in 1990. It rose to US$2,000 in 1995 and surged above the US$10,000 mark in 2008. In 2010, it leveled at US$11,809, which roughly equaled the GDP of a medium-developed country (Information Office of Shanghai Municipality and Shanghai Statistical Bureau 2011).

Situated in the subtropical zone and the East Asian monsoon belt, Shanghai has a mild and moist climate and experiences four distinct seasons. It has an average annual temperature of 16 °C, a yearly rainfall of 1,164.5 mm and a frost-free period of 222–235 day per year (Han et al. 2010). The rapid urbanization can increase temperatures considerably in the city and adjacent areas (Zhou et al. 2004). A correlation analysis of the relationship between the differences in mean temperatures in urban versus rural areas, has shown that the differences in temperature between urban and rural areas has increased substantially. Furthermore, the increase has been faster for the monthly mean maximum temperature than for the monthly mean minimum temperature (Zhao et al. 2006).

9.3 Biodiversity of Macrobenthic Invertebrates in Shanghai

Macrobenthic invertebrates are organisms that live on the bottom substrates of aquatic habitats and are larger than 1 mm (Shen and Shi 2003), or 0.5 mm (Liu 2000), depending on the classification scheme used. The organisms include sponges, nemerteans (ribbon worm), annelids (earthworm, bristle worm and leech), mollusks, cnidarians (sea anemone, coral and sea pen), echinoderms (starfish, sea urchin, and sea cucumber), ascidians (sea squirts), and arthropods (crustaceans, insects) among others.

Macrobenthic invertebrates produce a wide range of valuable ecosystem services. In freshwater ecosystems, they can be good indicators of water quality (Weigel et al. 2002; Cristina et al. 2009; Simone and Rui 2010). Furthermore, they can improve water quality, sustain commercial fisheries, and they support general ecosystem functioning that can provide people with leisure and recreational opportunities and inspiration for artistic expression (Shen and Shi 2003). Biodiversity and distribution of macrobenthic invertebrates are influenced by water temperature, salinity, primary productivity by plants, depth, sediment type, and physical disturbance (Coles and McCain 1990). Changes in macrobenthic community biodiversity and relative spatial distribution can influence primary (productivity of autotrophs such as plants) and secondary production (productivity of heterotrophs such as animals) (An 2010).

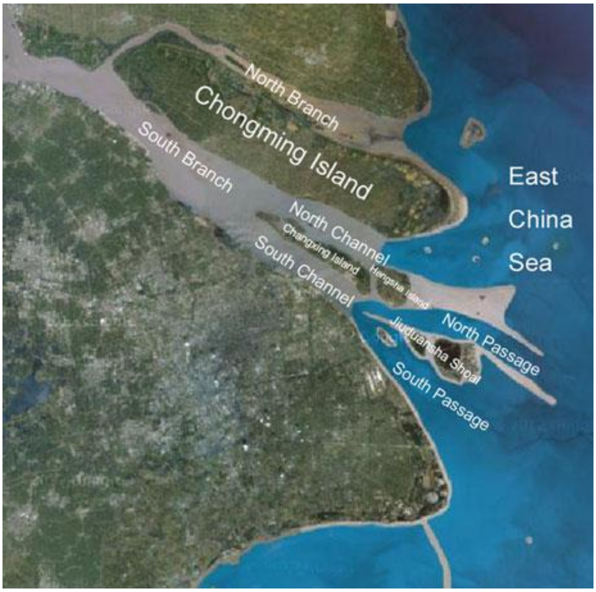

Fig. 9.3 Network of the Yangtze Estuary (Reproduced from Wang and Ding. 2012. Published with kind permission of ©Coastal Engineering Proceedings 2012, under the Creative Commons license: http://creativecommons.org/licenses/by/3.0/)

9.3.1 *Macrobenthic Invertebrates of Yangtze Estuarine Tidal Flat Wetlands*

There are about 2,699 km^2 of tidal flat wetlands in the Yangtze Estuary (Yun 2004) that provide good habitat for macrobenthic invertebrates. The channels in the Yangtze Estuary have an ordered-branching structure (Fig. 9.3): the estuary is first divided by the Chongming Island into the North Branch and the South Branch. Then the South Branch is divided into the North Channel and South Channel by the Changxing Island and the Hengsha Island. The South Channel is again divided into the North Passage and the South Passage by the Jiuduansha Shoal, which is now developing into an island (Wang and Ding 2012).

Macrobenthic invertebrates in the Yangtze Estuary are abundant; in 2004, their total biomass was about 5.29–6.73 × 10^4 t (wet weight) (Tong 2004). A total of 126 species of macrobenthic invertebrates belonging to 101 genera, 71 families, 22 orders, 8 classes and 5 phylum were recorded in the Yangtze estuarine wetlands (Liu and He 2007). Crustacea (shrimps, crabs, etc.) and mollusca (snails and mussels) were dominant, accounting for 58 and 29 % of the species collected, respectively (Fig. 9.4).

Many of the species are used as food and have high value, such as mitten crab (*Eriocheir sinensis*), marine shrimp (*Exopalamon carincauda*), and mud snail (*Bullacta exarata*), etc. (Fig. 9.4). The total value of important economic species was about US$3,155 per ha per year, and the highest one of them was *Eriocheir sinensis*, which could provide about US$1,412 per ha per year (Zhu 2004).

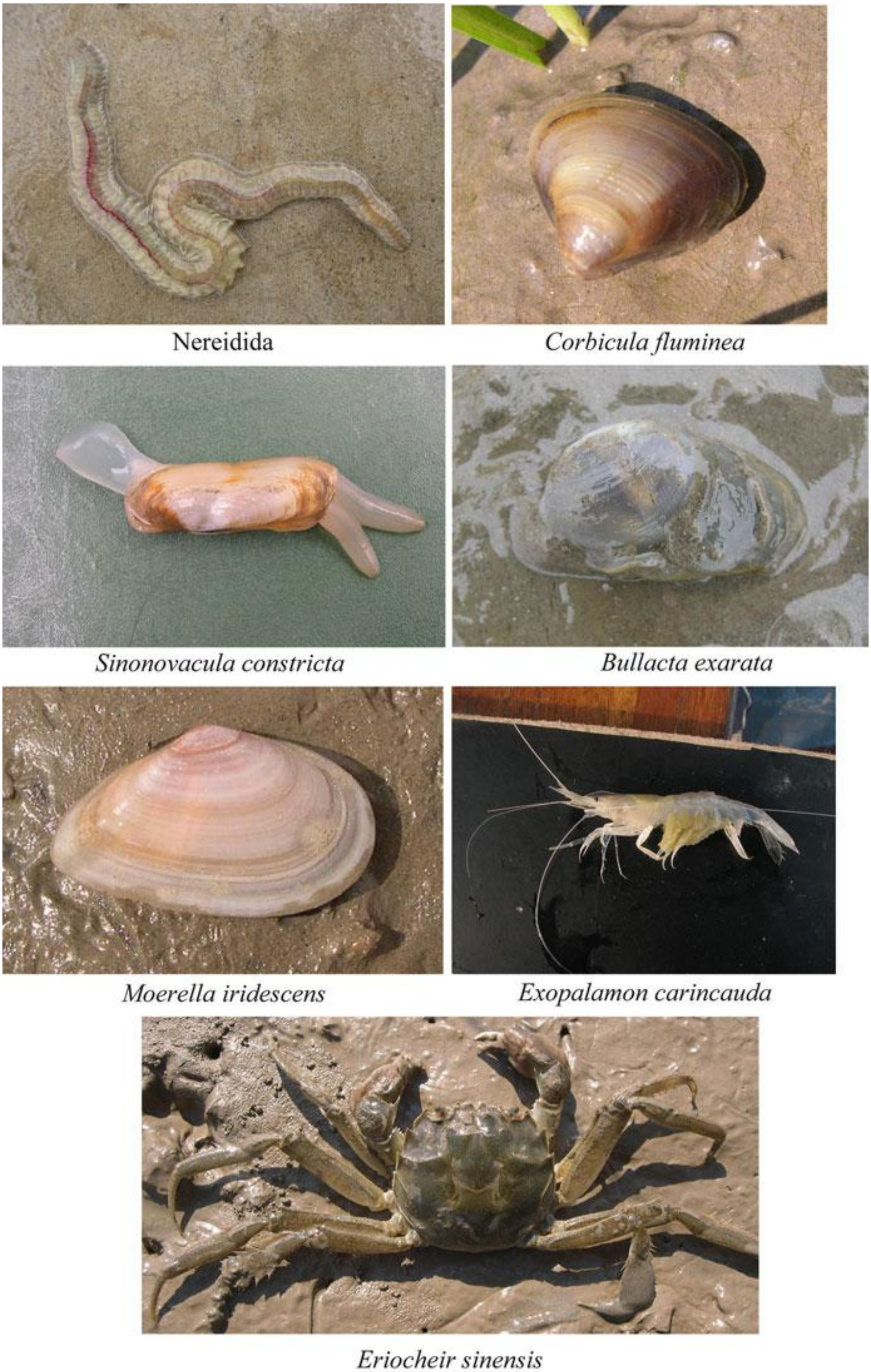

Fig. 9.4 The main commercial benthic productions in the Yangtze Estuary (Photographs by ©Wenliang Liu 2012, and published with his kind permission)

Table 9.1 Species composition and occurrence frequency of macrobenthic invertebrates of rivers in Shanghai

Taxonomy	Urban area	Suburban area	Outer suburban area	Frequency (%)
Annelida				
Limnodrilus hoffmeisteri	+++	++	++	50.6
Branchiura sowerbyi	++	+	+	18.1
Rhyacodrilus sinicus	++	+	+	25.3
Hirudo nipponia	++	+	+	10.8
Neanthes japonica	–	+	++	12.0
Nephtys galbra	–	–	+	1.2
Mollusca				
Bellamya purificata	+	++	+++	34.9
Bellamya quadrata	+	–	–	1.2
Bellamya angularis	–	+	–	1.2
Bellamya aeruginosa	–	–	+	4.4
Parafossarula eximius	–	–	+	1.2
Semisulcospira cancellata	–	–	+	1.2
Corbicula fluminea	–	++	+	14.5
Corbicula largillierti	–	–	+	1.2
Limnoperna lacustris	+	++	+	9.6
Anodonta woodiana	–	–	+	2.4
Arthropoda				
Chironomidae larvae	+	+	+	15.7
Grandidierella sp.	–	–	+	6.0
Sinocorophium sp.	–	–	+	1.2
Cyathura sp.	–	–	+	2.4

9.3.2 Macrobenthic Invertebrates of Rivers in Shanghai

Shanghai proper is bisected by the Huangpu River, a tributary of the Yangtze, and there are many rivers, canals and streams in the city. Macrobenthic invertebrate assemblages in rivers were investigated at 83 sites in the Shanghai metropolitan area in 2012. A total of 20 species were recorded, including 4 species of annelids (earthworms, bristle worms and leeches), 10 molluscs (snails and mussels), and 6 arthropods (crustaceans, insects) (Table 9.1).

The study showed on a strong correlation between the water quality and the species composition of the macrobenthic fauna. In the central urban area, the levels of organic pollution in the water were higher than in the peri-urban to rural areas, and the levels of dissolved oxygen were lower. The number of macrobenthic invertebrates species was shown to be relatively low as in total 11 species were collected, but the mean density high, at 8,776.3 ind./m^2. The majority of the species in the central urban area were characterized by high pollution tolerance. In suburban and outer suburban areas, the species diversity was found to be higher, at 15 collected species, but the density lower, at 690.3 ind./m^2. High population densities of pollution tolerant macrobenthic species (oligochaetes) were found in the west and north areas of the larger rivers, but in the relatively less polluted east and south areas of the larger rivers, the population densities were low. Pollution sensitive macrobenthic species including the mollusks and arthropods dominated in the suburbs and rural areas, where the organic pollution levels were lower, and dissolved oxygen levels were higher (Chen et al. 2013).

9.4 Effects of Urbanization on Biodiversity of Macrobenthic Invertebrates; Scenarios and Trends

9.4.1 Habitat Fragmentation and Loss: Tidal Flat Reclamation

The Yangtze estuarine tidal flats are important potential land resources for urban development and economic projects and reclamation of tidal flats is regularly carried out to meet the demand of rapid regional development. The activities increased the area of built-up land by 843 km^2 from 1949 to 2000 (Yun 2004). Reclamation has been recognized as the primary solution to secure land to meet future needs of the city's inhabitants, which indicates that the coastal environment will continue to be the major focus for development projects in the future (Naser 2011).

However, reclamation and dredging involves the direct removal of macrobenthic invertebrates, resulting in the physical smothering of the intertidal and subtidal habitats, and deoxygenates the underlying sediments (Allan et al. 2008; Newell et al. 1998). These physical and chemical alterations may reduce the overall biodiversity, i.e., species richness, abundance, and biomass of macrobenthic invertebrates in tidal marsh habitats (Smith and Rule 2001).

Data collected from samplings of the macrobenthic invertebrate community in the natural and diked tidal flats in the south bank of the Yangtze Estuary, showed that the species richness decreased in the tidal flats, and the composition of species changed after diking. The number of crustacean species decreased from 7 to 1, and polychaetes decreased from 4 to 3. The proportion of molluscs and insect larva species composition increased from 29.4 and 5.9 % to 50 and 25 % respectively (Yuan and Lu 2001).

Surveys of macrobenthic fauna before and after reclamation of the East Nanhui tidal flat were conducted from 2004 to 2009 (Ma et al. 2012). The results showed a decline in the number of species, from 32 to 26; the biomass values decreased from 38.8 to 1.97 g/m^2; and the diversity of dominant species declined significantly from 2004 to 2009. The results suggest that reclamation has a damaging effect on macrobenthic communities, by changing the elevation, hydrodynamics and characteristics of sediment and succession of vegetation of the tidal flat (Yuan and Lu 2001).

However, the macrobenthic invertebrates communities have shown to have a large capacity to self-rehabilitate following engineering projects in the tidal flats of the Yangtze Estuary, as the community structure and biomass recovered 270 days after engineering (Zhang et al. 2007). The influence of reclamation on macrobenthic invertebrates may decrease if enough new wetland regions would be created to replace the habitats lost by reclamation. In an analysis of the wetland evolution from 2000 to 2010 (Yang et al. 2011), the results showed that many new wetland regions have formed in recent years, such as the rapid and slow accretions in the east and northeast respectively of the Chongming Island, and the continuous increase in the Jiuduansha shoal. The study data (unpublished data, 2011 and 2012) suggested that in a parallel process, the predominant species, *Eohaustorius cheliferus*, disappeared from the eastern part of Hengsha Island after reclamation, while it became increasingly dominant in the growing Jiuduansha shoal.

9.4.2 Invasive and Exotic Species: **Spartina alterniflora**

With the acceleration of urbanization, city development demands more and more land. *Spartina alterniflora* (*S. alterniflora*), an invasive North America species of perennial grass growing on intertidal flats, was introduced to the Yangtze estuary in the 1990s as an ecological engineering species involved with coastal stabilization and land reclamation (Chung 2006). It is now a dominant species in estuarine salt marshes. *S. alterniflora* changed natural plant zonation patterns by expanding into *Scirpus* × *mariqueter* and *Phragmites australis* (*P. australis*), two native dominant perennial grass species, stands because of *S. alterniflora's* high tolerance of salinity and tidal immersion (Li et al. 2009).

The succession of vegetation could affect the species composition and distribution of macrobenthic invertebrates in the Yangtze estuary (Yang et al. 2007). The spread of *S. alterniflora* in *Scirpus* × *mariqueter* communities significantly decreased the species diversity of macrobenthic invertebrates in the earlier phase (Chen et al. 2005). However, a study of the macrobenthic fauna associated with *S. alterniflora* in the Yangtze estuary suggests that as *S. alterniflora* eventually formed a stable distribution, the species number and abundance of macrobenthic invertebrates increased, and a new structure of macrobenthic invertebrates community was formed in *S. alterniflora* zones. The new macrobenthic invertebrates community structure was different from that found in native salt marsh, and it had taken several years for the new macrobenthic invertebrate communities to establish and become stable (Xie et al. 2008). Research on benthic fauna in different marshes (*S. alterniflora*, *Scirpus* × *mariquete*r, and *P. australis*), showed that macrobenthic community structures differed in the proportion of native to exotic plants, but that the effects of plant types on species richness and densities were generally weak (Chen et al. 2009).

Many wetland mitigation plans require a 5-year monitoring period of the flora and fauna after development projects, but many macrobenthos need longer, even up to 25 years, to recover after changes to their habitats (Craft and Sacco 2003). Long-term monitoring studies are thus needed in the future. A model was constructed to simulate vegetation changes over time resulting from the changes in sediment loads and zonation in Jiuduansha shoal. Its simulations predict that areas of *P. australis* will continue to increase, and that *S. alterniflora* areas will decrease following a rapid initial increase (Wang et al. 2013). If these changes are realized, the overall community structure of macrobenthic invertebrates associated with in *S. alterniflora* will change in the future, which also would have effects on the ecosystem services produced and the income resource base.

9.4.3 Pollution: Sewage Discharges

Sewage effluents are major sources of river pollution in Shanghai (Cheng et al. 2006). Macrobenthic invertebrate assemblages in rivers were investigated at 83 sites in Shanghai metropolitan area in 2012 (study data 2012). The study showed a reduction in species diversity, richness and evenness of macrobenthic invertebrates in the areas where levels of organic enrichment, mainly ammonia and phosphate, were increased. No living samples were detected at nine sites that had particularly high levels of pollution, thus lacking suitable habitats. The water quality of the other 74 sites with living samples was evaluated by the Goodnight-Whitley Index (= Number of Tubificidae/Total number of benthic organisms × 100) (Goodnight and Whitley 1961). The index allowed separation of the sites into three groups: (1) 33 sites were severely polluted, with low richness and only dominated by high pollution-tolerant oligochaetes; (2) two sites were moderately polluted, dominated by oligochaetes but also with other species present; and (3) 39 sites were lightly polluted with high species richness.

The changes of the macrobenthic biodiversity during the ecological restoration process in Suzhou Creek (a river that passes through the Shanghai city centre) from 1999 to 2006 were analyzed (Liu 2007). The results indicated that the Shannon-Weiner index and Pielou's evenness index were positively related to water quality, as both indices increased with the improvement of water quality. However, the macrobenthic biodiversity showed no distinct improvement downstream if macrobenthic communities had deteriorated upstream (Cheng et al. 2006). The ecological restoration in Suzhou Creek is thus a long process.

9.5 The Governance Framework for Ecological Conservation and Restoration

Since the 1990s, the Shanghai Government has adopted a series of corresponding policies to protect the city's environment. Firstly, policies have been developed that aim to promote clean energy use and reduce water pollution, improve the sewage treatment infrastructure, remove exhaust emission sources, and improving transportation systems. Secondly, nature reserves and wetland parks have been established, which provide wetland protection as parts of the regional development.

9.5.1 Nature Reserves

Nature reserves can be an effective measure to conserve biodiversity, as they offer protection for the repository of genetic diversity, speciation and the source for metapopulation dispersal (Gong et al. 1993). Two national nature reserves associated with biodiversity conservation of macrobenthic invertebrates have already been established in Shanghai.

Shanghai Chongming Dongtan National Nature Reserve was established in 2006 to help strengthen the management of a 60,000 acre (24,000 ha) wetland reserve recognized under the International Ramsar Convention, through improved design and implementation of conservation strategies. To date, the managing body has conducted research with the government and academic partners on the distribution of species and ecological zones within this dynamic, continuously shifting estuarine environment. It has also trained government and academic partners in Conservation Action Planning, an approach to planning, implementing and measuring conservation strategies. Finally, it has assisted with an environmental awareness campaign targeted at reserve visitors from around the nation and the world (Xu and Zhao 2005).

Shanghai Jiuduansha National Wetland Nature Reserve was established in 2005 and covers an area of 103,833 acres (42,020 ha) at the junction of the Yangtze River and the East China Sea. *The Jiuduansha Reserve* is a typical estuarine tidal flat wetland, and the reserve is located in an area where the flows of the Yangtze River and tides of the East China Sea meet. Because of its particular geographical location and rich biodiversity, Jiuduansha is considered one of the most important estuarine wetlands in China. The results showed that estuarine wetland ecosystems are extremely sensitive to hydrological changes and other types of environmental changes. Hence, conserving Jiuduansha and its ecosystems can offer unique opportunities to explore the potential impact that water conservancy projects along the Yangtze River may have on ecological processes in the watershed and the estuary (Chen 2003).

9.5.2 Wetland Parks

Wetland parks conserve wetland landscapes with especially high ecological, cultural, aesthetic and biodiversity value, and maintain the ecological processes and ecological services functions (Wang 2008). They differ nature reserves in that visitors are allowed to enter, under special conditions, for inspirational, educative, cultural, and recreative purposes. Creating wetland parks can serve as an effective way to alleviate the contradiction between urban development and the protection of wetland biodiversity.

Chongming Xisha National Wetland Park, which is a typical estuarine wetland, is the only national wetland park in Shanghai, and is also a base for research on wetland ecological restoration. It is located in the southwestern part of the Chongming Island and has a total area of 4,500 ha. The Xisha wetland was previously flat and scattered with numerous ponds. Human activities (both development projects and individual actions) caused a reduction of biodiversity, which inhibited a stable, rich and dense bio-community and the ecosystems' functioning.

An optimized ecosystem was constructed and the biodiversity increased through environmental engineering of the wetland park that now includes an artificial forest swamp (Liu et al. 2009) and restored bird habitat (Gao and Lu 2008). It was opened to the public in 2005 and provides an example of a regional development strategy that also supports wetland protection.

9.6 Concluding Remarks

Shanghai has experienced rapid urbanization over the past three decades, accompanied by rapid acceleration of economic development. From 2000 to 2020, the urban area of Shanghai is predicted to increase at an annual rate of 3 %, and reach a total of 1,474 km^2 by 2020 (Han et al. 2009). By then, 92.6 % of the population in the Shanghai region is expected to be urban. The city is facing many challenges as its urban growth rate continues to accelerate.

Shanghai is a coastal metropolitan city with various types of natural and constructed wetlands which account for 23.5 % of its total area (Gao and Zhao 2006). Macrobenthic invertebrates play an important role for wetland ecosystem structure and functioning in and around the city. They are good indicators of the water quality and can play an important role for monitoring the impact of urbanization on the marine environment. The macrobenthic invertebrates also fill several other important functions, for example as a food resource for humans, a food resource for vertebrate predators, for filtering the water column, conducting sediment turnover, and acting as an organic consumer, as they eat plants, and contributor, as they become food for other animals. However, reclamation of the tidal flats by the expanding city, alien plant spread, and sewage discharge caused by rapid urbanization in Shanghai has changed the natural habitats of the macrobenthic invertebrates. This has changed the composition of species and abundance of macrobenthic invertebrates, and affected their capacity for ecosystem services provisioning. To support a transition to a socioeconomically and environmentally sustainable urbanization process, the Shanghai government has agreed on a series of policies aimed to protect wetlands in recent years, and established protected areas, i.e., wetland reserves and parks. The policies and protected areas have been helpful in restoring and improving the habitats of macrobenthic invertebrates. However, ecological restoration is a long process (see Chap. 31) and for conservation measures to be effective, they need to be able to respond to future development pressures as the city is expected to continue to grow.

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Chapter 10
Patterns and Trends in Urban Biodiversity and Landscape Design

Norbert Müller, Maria Ignatieva, Charles H. Nilon, Peter Werner, and Wayne C. Zipperer

Abstract Urbanization destroys or modifies native habitats and creates new ones with its infrastructure. Because of these changes, urban landscapes favor non-native and native species that are generalists. Nevertheless, cities reveal a great variety of habitats and species, and, especially in temperate cities, the diversity of vascular plants and birds can be higher than in the surrounding landscapes. The actual occurrence of a species, however, depends on habitat availability and quality, the spatial arrangements of habitats, species pools, a species' adaptability and natural history, and site history. In addition, cities are particularly human-made ecological

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systems. Top-down and bottom-up activities of planners, land managers, and citizens create the urban biodiversity in general and in detail. Plants and animals in cities are the everyday life contact with nature of the most humans on our earth. The intrinsic interplay of social and ecological systems with a city often forms unique biotic assemblages inherent to that city. To support native biodiversity, landscape architects, conservation biologists, and other groups are linking landscape design with ecosystem structure and function to create and restore habitats and reintroduce native species in cities.

10.1 Introduction

Urbanization is a double-edged sword. On one edge, urbanization destroys and fragments natural ecosystems, introduces non-native species, degrades and alters ecosystem processes, and modifies natural disturbance regimes. On the other edge, urbanization creates social and economic opportunities, centers of art and culture, and truly unique ecological spaces through design. Cities are not landscapes depauperate of plants and animals, but rather novel places teaming with unique plant and animal communities. In fact, cities can play an essential role in meeting the Convention on Biological Diversity (CBD) target of stemming biodiversity losses. This role includes three complementary components: (1) sustaining ecosystem goods and services for and within cities; (2) conserving biodiversity within towns and cities and promoting the sustainable design of all urban areas to maximize their ability to support biodiversity; and (3) promoting awareness and influencing decision-making to create livable spaces not only for humans, but also plants and animals.

This chapter will examine how urbanization affects biodiversity at local and regional scales, how novel biotic communities and habitats are created, how social contexts influence species patterns and richness, and how landscape-design is influencing biodiversity in cities. The chapter also examines the role of non-native species in urban landscapes and how species are evolving in cities. The term "urban landscapes" is used in this chapter to capture the diversity of human communities ranging from small settlements such as villages or towns whose populations are less than ten thousands to megacities whose populations are greater than ten million humans.

10.2 Biodiversity Patterns

An important component of evaluating biodiversity in urban landscapes is defining biodiversity (Box 10.1). For this chapter, the distribution of species and species richness across the urban landscape is used as one measure of diversity. To assess species richness, it is also necessary to define what a native and non-native species is and examine the types of sampling protocols to measure richness.

The term 'non-native species', used throughout this chapter, is the equivalent of 'alien species' as used by the CBD. It refers to a species, subspecies or lower taxon,

Box 10.1 What Is Urban Biodiversity?

Urban biodiversity is 'the variety or richness and abundance of living organisms (including genetic variation) and habitats found in and on the edge of human settlements'. Species range from the rural fringe to the urban core (see Chap. 1). The following examples of habitats found in human settlements:

- Remnant vegetation (e. g., remnant habitats of native plant communities, rock faces)
- Agricultural landscapes (e. g., meadows, arable land)
- Urban-industrial landscapes (e. g., wastelands and vacant lots, residential areas, industrial parks, railway areas, brown fields).
- Ornamental gardens and landscapes (e.g., formal parks and gardens, small gardens and green spaces)

introduced (i.e., by human action) outside its past or present natural distribution and includes any part—gametes, seeds, eggs, or propagules—of such species that might colonize, grow and mature and subsequently reproduce. The chapter does not address genetically modified organisms (GMOs) or non-native fungi, bacteria, and viruses.

To measure biodiversity in an urban landscape, one must account for the relative age and area of the urban landscape as well as inventorying methodology. In general, older urban landscapes have more non-native species than recently settled landscapes (Pyšek and Jarošík 2005). Furthermore, larger urban landscapes (e.g., cities) have more non-native species than small urban landscapes (e.g., villages and towns) (Pyšek et al. 2004). Within a given city, different sampling designs may have been applied at different times, yielding different species richness values, and thereby limiting comparability. In conjunction with sampling protocol, the area being sampled and the intensity of sampling needs to be considered. City boundaries, landscape heterogeneity, and ownership patterns change over time and these changes can affect not only areas being inventoried but also species distribution. For example, biodiversity studies in urban areas are often conducted on public spaces where access is not limited. Yet, often more than 70 % of the land in urban areas is privately owned. Because private landowners control the vegetation structure on their properties, these properties can influence urban biodiversity tremendously (van Heezik et al. 2012). Their absence from sampling can affect the overall recorded species for an area. Likewise, an important element of sampling is what constitutes a count—a single individual of a species or a viable population?

Although different protocols can be used to describe and quantify the effect of urbanization on biodiversity, the two primary techniques are the urban-rural gradient and comparisons among land uses. Urban-rural gradients represent anthropogenic gradients that result from patterns of human development. Based upon techniques used by plant ecologists to study the influence of an environmental gradient on community composition, Sukopp and Werner (1982) and McDonnell and Pickett (1990) propose using the gradient approach to capture changes in land use, the bio-physical environment, and alternation of disturbance as one moves from the urban core to the rural fringe. Sukopp (1973) and Sukopp et al. (1980) illustrate these changes for a typical city in the northern hemisphere (Fig. 10.1) (Box 10.2). These gradients, however, may not be linear (e.g., high to low) as one moves from the urban to the rural landscape but rather are dependent on the organizational structure of the city—concentric, sector, or multiple nuclei (Harris and Ullman 1945). Quantifying the spatial and temporal scales of urban components (i.e., what is urban) along the gradient is paramount for comparability (McIntyre et al. 2000).

The gradient approach can also be applied within a city by comparing different land uses. Blair (1996) uses this urban gradient approach to study how avian diversity varied by land use. Land use is used to capture urban morphology and generally includes residential, agricultural, transportation, industrial, commercial, recreational, institutional, and 'natural' cover such as forest (see Anderson et al. (1976) for a detailed description and definitions of land usage). Scientists often treat land-use as a homogenous area with respect to environmental and anthropogenic factors, but heterogeneity within a land use often exists because of different building types and social contexts (Kinzig et al. 2005). These differences can influence the presence and distribution of species. Like defining an urban-rural gradient, how the various land uses are defined and the scale of at which measurements are taken are critical for comparing species patterns across different studies.

Despite the issues associated with definitions and sampling, general patterns of species richness are discernible. For instance, native species richness declines and non-species richness increases as one moves from the rural fringe to the urban core with approximately 30–50 % of the plant species in the urban core being non-native (Dunn and Heneghan 2011). Similarly, under some conditions of low to moderate levels of urban development (i.e., suburbanization), species richness may actually increase (McKinney 2002). The increased number of species in suburbanizing landscapes results from high habitat heterogeneity, high number of introduced species, socio-economic factors, and altered disturbance regimes (see Kowarik 2011). Another species pattern observed in urban landscapes is that species tend to be non-native invasives and native generalists, which are tolerant to the urban conditions. The literature, however, provides studies that are contrary to these generalities. For example, Hope et al. (2003) report that species richness in Phoenix, Arizona, a city in the desert, increases with urbanization because of human influences such as irrigation and ornamental landscaping. In a review of gradient studies, McDonnell and Hahs (2008) actually identify five response curves for native species as urbanization increases: (1) no response, (2) negative response, (3) punctuated response, (4) an intermediate response, and (5) a bimodal response. Although not stated by McDonnell and Hahs (2008), a native species may also show a positive response to urbanization (see Sect. 10.4.1). To examine more closely how urbanization affects species richness, a detailed discussion of plants and birds and—to a less detailed degree—mammals, amphibians reptiles, and invertebrates is provided in the subsequent sections.

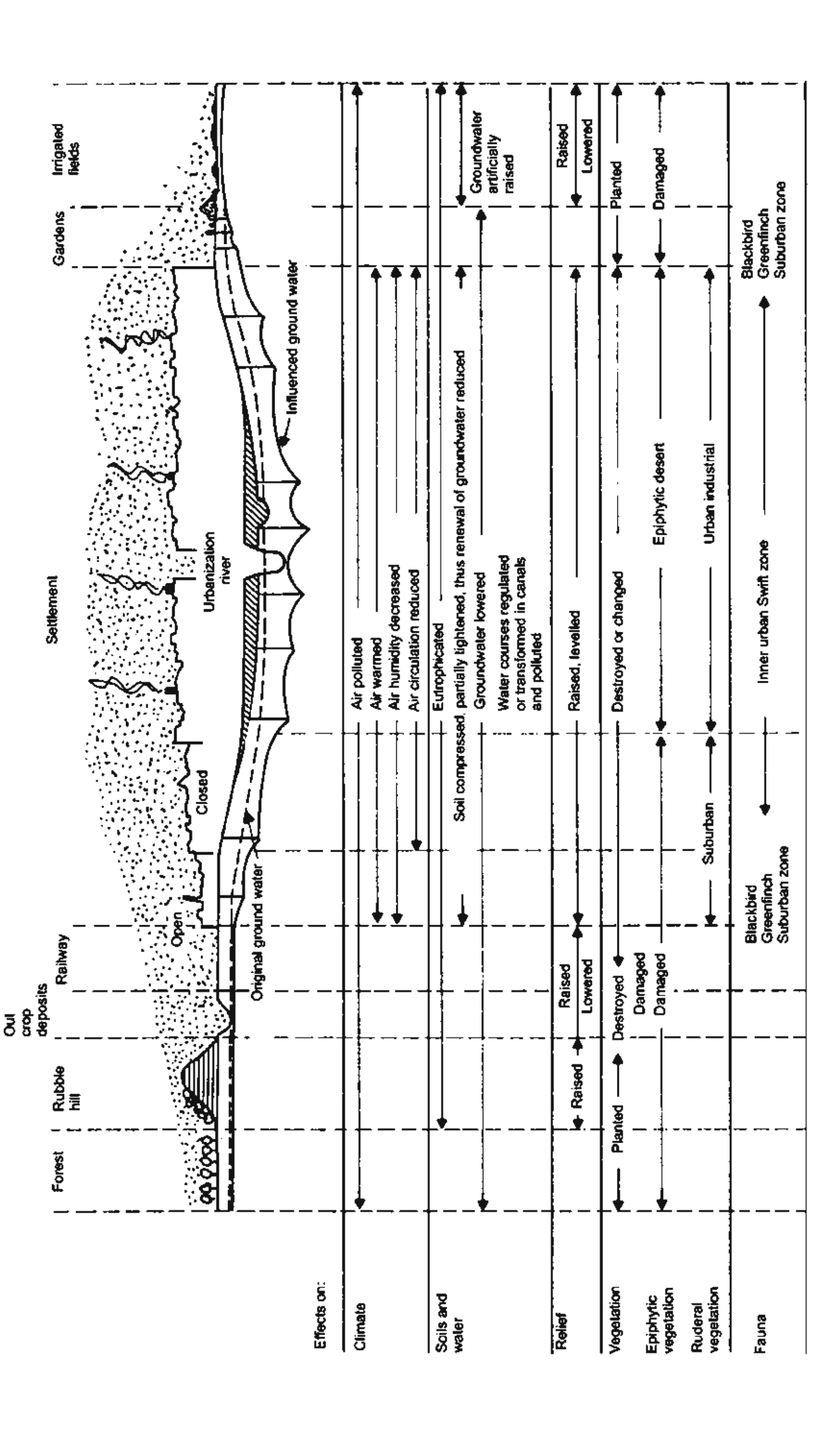

Fig. 10.1 Variations in biotic and abiotic components of a city's biosphere in the northern hemisphere (Reproduced from Sukopp et al. 1980. Published with kind permission of © Herbert Sukopp 2013. All rights reserved)

Box 10.2 A Number of Attributes That Define an Urban Area and Can Subsequently Affect Biodiversity (From Müller and Werner (2010) After Sukopp and Wittig (1998) and Pickett et al. (2001)) (See Chap. 1 for a Definition of Urban)

1. Configuration of buildings, technical infrastructure and open spaces where the extent of hard surface (including buildings, paving and other structures) covers an average of 30–50 % of the land surface in the urban fringe and suburban areas, and well in excess of 60 % in the core areas.
2. Formation of an urban heat island effect in temperate and boreal zones with longer periods of plant growth, warmer summers and milder winters than the surrounding countryside.
3. Modification of the soil-moisture regimes, tending to become drier in temperate zones, but with opposite effects in desert areas due to irrigation.
4. High levels of nutrient input at both point source and broad-scale.
5. High biomass production in parks, private and community gardens, and similar intensively cultivated or managed areas.
6. Intentionally and unintentionally elevated food availability for animals both wild and domesticated.
7. Soil contamination, air pollution, and water pollution; with particular impacts on soil organisms, lichens, and aquatic species.
8. Disturbance such as trampling, construction (often with removal of all vegetation), mowing, radical soil change, light and sound pollution, and litter or illegal dumping.
9. Fragmentation of forests, grasslands and waterways as well as existing green spaces.
10. High proportion of introduced plant- and animal species.
11. High proportion of habitat generalists and common plant and animal species.

10.3 Plant Species Richness in Cities

Humans have a long history of transporting plant species and affecting local biodiversity. Since the Neolithic period, 12,000 species have been introduced into Central Europe for ornamental and cultural purposes and approximately 10 % (1,100) of those plants have become naturalized (Lohmeyer and Sukopp 1992). The 10 % naturalization of non-native species appears to be a general rule for continental flora (Reichard and White 2001); however, the effect of naturalization may be more dramatic on islands. For instance, on New Zealand, Ignatieva et al. (2000) and Stewart et al. (2010) document only 48 native species of a total of 317 vascular plant species in Christchurch. Pyšek (1998) also reports that the area extent of the urban landscape affects flora diversity. Villages have greater proportion of native species,

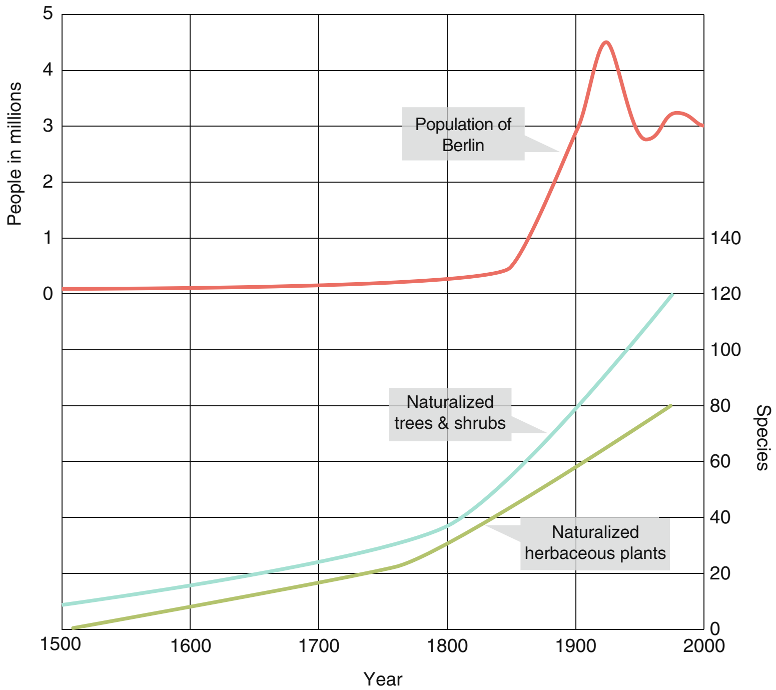

Fig. 10.2 Correlation between human population growth and naturalized exotic plants in Berlin (Modified from Sukopp and Wurzel 2003. Published with kind permission of © Urban Habitats 2003 and Herbert Sukopp 2013. All Rights Reserved)

whereas cities have greater proportion of non-native species with respect to the total number. Similarly, Sukopp and Wurzel (2003) correlate the increase in the number of naturalized species (trees, shrubs and herbaceous plants) with city expansion of Berlin, Germany during the nineteenth and twentieth centuries (Fig. 10.2).

The relationship between human population size and biodiversity is more complex. Luck (2007) reviews the relationship between human population density and biodiversity and reports a positive correlation between human population density and species richness (primarily plants and birds) because of the co-occurrence of human settlements and species-rich areas (see Sect. 10.7 and Chap. 3). Scale of the geographical areas plays an important role in analyses with positive correlations between human population density and species richness occurring for sampling areas greater than or equal to 2,500 km^2. For sites less than 2,500 km^2, the correlation is less apparent because of geographical biases, scale of sampling, and sampling protocols; all are factors identified in the previous sections. Geographically, most biodiversity studies are conducted in the Northern Hemisphere with a high proportion in the United States (29 %) (see Chap. 27 for a discussion of a similar trend exhibited for studies on urban governance for biodiversity, and see Chap. 33 for a general examination of this northern bias). With projected human population growth

(Chap. 21), regional planning and landscape design become paramount if cities are to reduce threats to threatened and endangered species, protect existing conservation areas, and minimize habitat loss and degradation (Luck 2007). In fact, the large forests in many urban landscapes will become increasingly more important for biological conservation. Examples of these spectacular forests include the Tijuca Forest in Rio de Janeiro; the Bukit Timah Nature Reserve in Singapore; Riccarton Bush in Christchurch; the El Ávila National Park in Caracas; remnants of Australian bush land habitats in Perth, Sydney and Brisbane; natural forest remnants in New York, Stockholm, St. Petersburg and Moscow; the Ridge Forest in New Delhi; and rock faces and outcrops in Edinburgh.

Williams et al. (2009) identify four primary factors or "filters" influencing the distributions of plant species in urban landscapes. They include (1) habitat availability, (2) the spatial arrangement of habitats, (3) the pool of plant species, and (4) evolutionary pressures on populations. With the exception of spatial arrangement, these filters mirror the factors influencing vegetation dynamics as posited by Pickett et al. (1987)—site availability, species availability, and species performance. An aspect of habitat availability is site history, which encapsulates ownership legacy and use. This history can be extensive, especially for ancient cities (see Celesti-Grapow et al. 2006 and Chap. 2). Furthermore, these filters or factors work synergistically and simultaneously, rather than independently, and their effects will vary by species (Williams et al. 2009). The next subsections examine species availability, unique habitats, and species traits for plants in urban landscapes.

10.3.1 Species Availability

Williams et al. (2009) identified three sources of species in urban landscapes: (1) native species originating in the area itself, (2) native species occurring regionally, and (3) non-native species introduced by humans or naturalized in the region. Wittig (2004) recognizes a fourth source, anecophytes, which are species with European origins that have no natural habitats but have evolved to adapt to agricultural, urban, and industrial landscapes. All of these sources are ecologically and anthropogenically dynamic. Changes in any of them may affect species diversity in a city (Tait et al. 2005).

Analyses of long-term species records provide insights into how these sources change. Chocholoušková and Pyšek (2003) examined the vegetation of the city of Plzen, Czech Republic and its surrounding area for three periods of time: 1880–1910, the 1960s, and the 1990s. Over the 120 year period, 805 species were permanently present, 368 disappeared, and 238 were new additions. Total species richness of the city and surrounding area decreased from 1,173 recorded in 1880–1910, to 989 in the 1960s, then increased to 1,043 in the 1990s, a 17 % total change over time. Interestingly, species richness in the surrounding area declined from 1,112 to 745 species, whereas the city's species richness increased from 478 to 773 species, primarily through the introduction of non-native species. Of the 1,459 total number of species inventoried, 13.6 % were archaeophytes (introduced before 1500), 15.4 %

neophytes (introduced after 1500), and 71.0 % native species. Similarity coefficient (Jaccard) between the surrounding area and the city increased from 35 % for the period of 1880–1910 to 46 % in the 1990s. Woody species, both shrubs and trees, increased in the city over the study period. A closer examination of the woody vegetation showed that neophyte woody species increased from 2 to 8 to 33 species (Chocholoušková and Pyšek 2003).

At a finer scale, DeCandido et al. (2007) examined the history of species change for Central Park in New York, NY, USA. Central Park is 341.2 ha and was established in 1853. Based on nineteenth-century plant lists, herbarium specimens, and field surveys in 2006–2007, DeCandido et al. (2007) reported that during the late nineteenth and early twentieth centuries 356 species—255 (74 %) natives and 91 (26 %) non-native—were recorded. From the 2006–2007 survey, 362 species—145 (40 %) native and 217 (60 %) non-native—were recorded. A cumulative list of total species from all sampling periods was 583 species—331 (57 %) native and 252 (43 %) non-native. Over the study period, 260 new species (64 % non-native) were added and 198 species (90 % native) were lost. Of the lost native species, 117 were annuals associated with wet meadows and woods; these are habitats-types that were lost during park development.

Other authors have reported similar patterns of shifts in species richness with turn-over rates ranging from 3 to 55 % (DeCandido 2004; Godefroid 2001; Landolt 2000; Werner and Zahner 2009). These studies indicate that turn-over rates are more complex than just non-native species replacing native species. Although non-native species can out compete native species in shared habitats, loss of native species often results from habitat loss, shifts in land use and site history, or changes to environmental conditions such as altered disturbance regimes (e.g., fire suppression), altered hydrological patterns, increased desiccation, and reduced light availability (Hahs et al. 2009; Gregor et al. 2012). In general, herbaceous plants (primarily wetland species or species associated with wet soils) are the dominant native species being extirpated (Ricotta et al. 2009). These studies also highlight the need to examine species by life form or functional groups to gain a better understanding of how species are responding to urbanization and the effect of species loss on the ecosystem. Although the general pattern is of native species richness declining and non-native species richness increasing over time, collectively, native species can comprise 50–70 % of total species richness in a city, albeit sometimes as rarer species (Kowarik 2011).

With increased dominance of non-native plant species and the extirpation of native species in urban landscapes, McKinney and Lockwood (1999) postulated that biotic homogenization was occurring—an increased similarity of species composition between sites, which, historically, had disparate floras. Based on 20 localities in the United States, McKinney (2008) observed localities with a relative high number of total non-native species (>200 species) were more similar compositionally than localities with fewer non-native species, and regardless of distance between localities, non-native species had higher similarities among localities than native species. In other words, with increased urbanization, the urban environment promoted the proportion of total shared species by promoting more shared species among non-natives (McKinney 2008).

Biotic homogenization appears to be scale and site related. Rejmánek (2000) examined the flora of states in the United States and reported that non-native plants species actually increased floral distinctiveness for adjacent states. Similarly, in examining the flora of Germany, Kühn and Klotz (2006) observed greater heterogeneity of native species in urban than in rural sites, and urbanization did not have the overall effect of homogenization of all species. Overall, at the regional scale, urbanization did not contribute to homogenization (Kühn and Klotz 2006). Kühn et al. (2004) attributed the lack of homogenization to the occurrence of human settlements in biological hotspots for native species, the greater diversity of available habitats in urban areas, and different invasion rates for non-natives species (also see Olden et al. 2004). Collectively, these studies point out that homogenization is a more complex phenomenon in urban landscapes than previously thought and warrants greater investigation.

In addition to biotic homogenization, Olden et al. (2004) also identified three other types of homogenization: genetic, taxonomic, and functional. Genetic homogenization reduces the spatial separation of genetic variability within a species or population through direct introductions of species outside their normal range or through extirpation of local populations. Horticultural practices directly facilitate the introduction of species outside their normal ranges which has led to intraspecific (i.e., within a species) and interspecific (i.e., occurring between species) hybridization and often, the creation of invasive species (see Schierenbeck and Ellstrand 2009). For instance, Culley and Hardiman (2007) document the intraspecific hybridization of Callery pear (*Pyrus calleryana*), a commonly planted street tree, which has resulted in the invasive species currently colonizing natural areas in the Midwest United States. Similarly, Trusty et al. (2008) document the interspecific hybridization of *Wisteria sinensis* and *W. japonica*, species imported because of showy floral displays and sweet fragrance, and the resulting invasive progeny. Bleeker et al. (2007) identify 134 hybrids resulting from the hybridization between 81 non-native species and 109 native species. Interspecific hybridization between a non-native species and a rare-native species is especially problematic because of the dilution of genetic material (swamping gene flow) by the non-native species and outbreeding depression (a reduction in progeny fitness). Each of these issues needs to be considered when developing conservation strategies for rare, native species (Bleeker et al. 2007).

Taxonomic homogenization, largely from a phylogenetic perspective, refers to an increase in compositional similarity among communities (Olden et al. 2004). Knapp et al. (2008) illustrate taxonomic homogenization in an urban context using the Kühn et al. (2004) data set for Germany. As previously mentioned, Kühn et al. (2004) identify high species richness and the lack of biotic homogenization in Germany's urban landscapes. A closer examination of species data reveals that the urban landscapes may have been more species rich than corresponding rural landscapes, but phylogenetically, urban landscapes are less diverse than rural landscapes. In other words, because of the urban filters (see Williams et al. 2009) acting on available species in urban landscapes, species are more closely related functionally than species in rural landscapes (Knapp et al. 2008). Ricotta et al. (2009) discern a similar pattern when comparing 21 floras from European and U.S. cities, and report that that non-native species had a significantly lower phylogenetic diversity than native species. Consequently, the flora in urban landscapes, with its lack of phylogenetic diversity, may be less adaptable to environmental change (e.g., climate change) than flora in rural landscapes.

Functional homogenization, a measurement of the increase in spatial similarity of functional variables over time, is based on the assumption that the simplification of species (through the loss of specialists to generalists) and the simplification of phylogenetic diversity leads to a reduction in ecosystem function (and subsequently, ecosystem benefits and services) (Olden et al. 2004; Clavel et al. 2011). Unfortunately, as opposed to well-documented effects of urban environment on functional homogenization (see Pickett et al. 2011), there is a lack of information on changes in functional diversity resulting from a simplification of species richness across the urban landscape. Research at fine scales indicates that species can alter biogeochemical processes (Ehrenfeld 2005) and carbon sequestration accumulation (Escobedo et al. 2010), but how functional homogenization manifests itself across the urban landscape still needs to be determined.

10.3.2 Habitats

Urbanization transforms landscapes. It fragments or obliterates natural vegetation resulting in habitat loss and isolation. It alters the spatial arrangement of landscape components and modifies heterogeneity thereby disrupting ecological pathways. It modifies the climate by creating urban heat islands. These changes often result in the loss of native plant and animal species (Dunn and Heneghan 2011).

Assessments of patches of remnant vegetation show that patch configuration plays a significant role in determining plant species richness (Burgess and Sharpe 1981). In general, larger remnant patches contain more native species than smaller patches in urban landscapes (see Godefroid and Koedam 2003). Consequently, conservation strategies in urban landscapes favor preserving larger patches over smaller ones. Smaller patches, however, can play significant roles in maintaining overall richness in an urban landscape by containing unique habitats (Florgård 2007; Forman 1995), and serving as stepping-stones or increasing connectivity for species that migrate among habitats and through the landscape (Forman and Collinge 1996). Actually, these small patches, in combination with backyard habitats, form a habitat network in urban landscapes that is critical to species conservation (Rudd et al. 2002).

Patch history plays an equally important role in determining the species composition. By distinguishing remnant sites (i.e., those never cleared for urban use)—from emergent sites (i.e., those cleared for urban use and allowed to reforest), Zipperer (2002) shows that the emergent forest patches have a greater plant species richness and greater number of non-native species than the remnant patches. Emergent patches are also dominated by wind-dispersed species, whereas remnant forest patches are dominated by animal-dispersed species. Analysis of wastelands and derelict sites (i.e., abandoned land where plants grow without any human

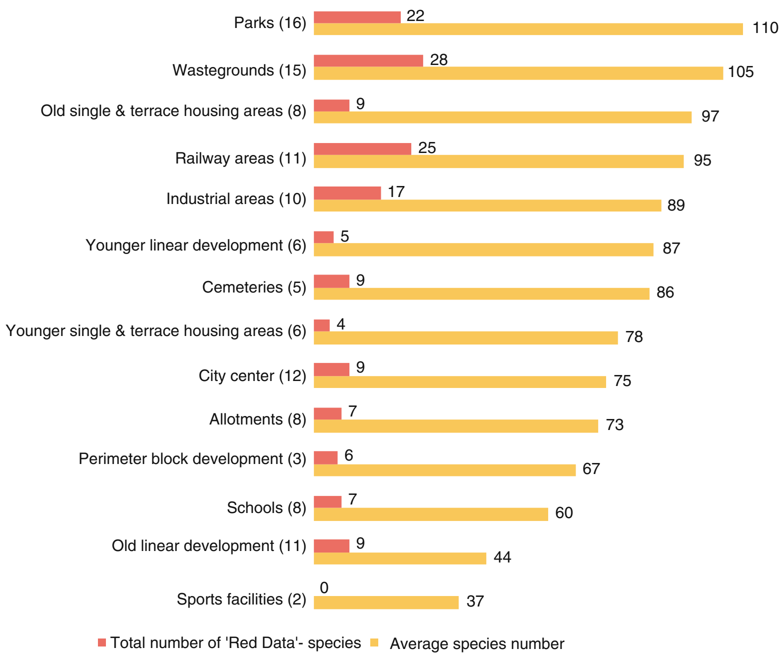

Fig. 10.3 Average number of vascular plant species and number of IUCN Red List plant species found within different land-use types in Augsburg, Germany. The number of times each land-use type was sampled is listed in *parentheses* (Modified from Müller 2011. Published with kind permission of © Springer 2011. All Rights Reserved)

control (Muratet et al. 2007)) in Europe show a similar pattern of non-native, wind-disseminated species dominating the site (Godefroid et al. 2007). The assemblage of native and non-native species on emergent forest patches and on wastelands forms novel ecosystems whose vegetation dynamics, biogeochemistry, and ecological functions are only now being identified and evaluated (Hobbs et al. 2006; Sukopp et al. 1979).

Urbanization also creates new habitats such as road verges, vacant lots and wastelands, hard surfaces and walls, parks, and gardens (Fig. 10.3). These habitats contribute not only to the overall plant species richness of a city, but also to the preservation and conservation of endangered native species. Although most of these habitats have only recently been studied from an ecological perspective, some have been extensively studied during the last century (see Gilbert 1989; Sukopp and Wittig 1998). For instance, the effect of roads has been extensively studied from multiple perspectives such as biogeochemistry, wildlife mortality, and chemical disposition (Forman 1995); however, Trammell and Carreiro (2011) only recently conducted a structural and functional analysis of road verges. In Louisville, Kentucky, USA, they observed that distance from city center was a primary determinant of plant species composition and structure. Plots located further from the city had lower stem density but higher species richness than plots located in the city

(Trammell and Carreiro 2011). They also observed an increase in non-native species, primarily Amor honeysuckle (*Lonicera maackii*), closer to the city.

Vacant lots, wastelands, and derelict sites include: sites where infrastructure once occupied but since have been removed, sites that have been abandoned and are no longer managed, and sites created from war (see Sukopp 2002). Sites often are poorly drained because of compacted soils or rapidly drained because of the additional construction debris mixed into the soil. Construction material also increases alkaline concentrations. These sites are often short-lived habitats because of irregular patterns of disturbances and new buildings being erected. Nonetheless, vacant lots and wastelands can be quite species-rich and can contain species native to the area, species from agricultural sites, and ruderal, non-native species (Kelcey and Müller 2011; Muratet et al. 2007). Prach and Pyšek (2001) report that soil fertility can play an important role in vegetation dynamics on wastelands in Central Europe with European aspen (*Populus tremula*) dominating on poor sites, and black elderberry (*Sambucus nigra*) and goat willow (*Salix caprea*) dominating on moderately fertile sites. Del Tredici (2011) calls this suite of species occupying vacant lots a "cosmopolitan assemblage of early-successional, disturbance-tolerant species that are pre-adapted to the urban environment".

Hard surfaces are not unique to urban landscapes but proliferate because of building construction, stone and brick walls, and pavements as well as the presence of ruins (Lundholm 2011). Although these sites often are hostile environments for plants (e.g., due to the lack of soil and moisture, and extreme temperatures), they can harbor a unique array of species that contribute the overall native species richness of a city. Two key factors influencing vegetation on walls are age of the surface and moisture availability (Darlington 1981). Compared to newer walls, older walls and mortar tend to have more species because they have weathered more, have had time to neutralize alkaline conditions, and accumulate organic material in cracks (thus creating a rooting zone for vegetation). Darlington (1981) also reports that oceanic climates with high rainfall and relatively low temperature fluctuations favor vegetation developing on walls. By comparison, walls in arid climates have limited vegetation on surfaces because of desiccation. Wall vegetation includes angiosperms as well as algae, cyanobacteria, lichens, bryophytes and ferns (Lundholm 2011). Although exceptions do exist, Lundholm (2011) reports the following species patterns on hard surfaces in urban landscapes: hemicryptophyes (i.e., perennials with their buds at or near the soil surface) are dominant in Atlantic and Central Europe, chamaephytes (i.e., woody species with resting buds at or near the soil surface) in Mediterranean Europe, therophytes (i.e., annual species) in India, and phanerophytes (i.e., woods species with resting buds above the soil surface) in Israel. Interestingly, the shift of construction material from stone and concrete to glass and metal for construction surfaces threatens the occurrence of this biota in cities.

Another set of novel habitats in urban landscapes are parks and gardens. Of all the habitats in a city, parks and gardens truly demonstrate human expression and creativity. Urban parks are not only credited for their ecosystem services and positive aesthetical and social values (Bolund and Hunhammar 1999; Chiesura 2004), but also act as hot spots of biodiversity in urban areas (Cornelis and Hermy 2004).

For instance, old historic parks in Europe are a complex combination of habitats of native vegetation, historical cultural landscape and typical urban vegetation such as lawns. They often contain and support the preservation and conservation of endangered and rare taxa (Kümmerling and Müller 2012). On the other hand, parks can be sources for plant invasions through extensive use of non-native plants (Dehnen-Schmutz et al. 2007). This is especially true for parks outside of Europe (Ignatieva 2010) (also see Sect. 10.6).

Private residential gardens in the United Kingdom, USA, and other colonial countries with similar urban planning structure, may represent as much as 27 % of the land area in a city (Smith et al. 2006; Thompson et al. 2003). Although generally ignored by ecologists as significant habitats in urban landscapes, gardens contribute significantly to plant species richness and to insect and avian species diversity by providing critical habitat for nesting, food, and cover (Smith et al. 2006). Because of their importance to city biota and humans, we will examine gardens in a greater detail than other habitat types (also see Sect. 10.6.1).

Ecologically, gardens are species rich. Thompson et al. (2003) inventoried 60 gardens in Sheffield, UK and observed 438 species, 33 % of which were British natives. Overall, native species richness was not correlated with garden size, but total species richness was. Thompson et al. (2003) also reported that total species richness was greater in gardens than any other natural community, principally because of the addition of non-native species. Management also plays an important role in maintaining species richness in gardens. Through active management, more species can be maintained in a given area than otherwise would have occurred naturally (Thompson et al. 2003). Loram et al. (2008) inventoried five UK cities—Belfast, Cardiff, Edinburg, Leicester, and Oxford—and observed similar patterns of species richness in gardens. Across these cities, native species represented 34 % of the total species inventoried. Surprisingly, the most frequently sampled species were 20 native species. In fact, Loram et al. (2008) reported no differences for species richness, diversity, and composition across cities, which varied climatically and geographically. Similar results were reported from biodiversity studies for gardens in front of homes in Germany (Müller 2010a). Quigley (2011) recognizes the contribution of gardens to species richness, but questions the ecological functionality of gardens. Generally, gardens are developed for visual appeal and do not increase trophic diversity and often are not self-sustaining (see Sect. 10.5).

Like gardens, lawns are ubiquitous and unique habitats, which cover large areas in urban and urbanizing landscapes all over the world (Müller 2010b; Stewart et al. 2009). Lawns are found in parks, playing fields, golf courses, along streets and roads, in plazas and schoolyards (Ignatieva and Stewart 2009). Lawns are nearly universal in front and back yards in suburban gardens in UK, USA, Australia, and New Zealand. However, lawns and gardens (as a whole) differ with respect to species richness and effect of management. Thompson et al. (2004), sampled 52 lawns in Sheffield, UK, and identified 159 species with 94 % being native. In fact, the 24 most common species were native. By comparison, 'colonial lawns' in New Zealand showed the opposite trend with non-native species dominating (Ignatieva and Stewart 2009). Stewart et al. (2009) studied 327 lawns in Christchurch, NZ and identified

127 species with the majority (81 %) being non-native. They observed that of the 25 most common species, 22 were non-native whose origins were primarily Eurasian and some from North America. The majority of native species were forbs which were often removed because they were regarded, along with non-native forbs, as 'weeds' (Ignatieva and Stewart 2009).

Meurk (2004) conducted an inventory of lawns in both northern and southern hemispheres. In the northern hemisphere, "core" grass species were Kentucky bluegrass (*Poa pratensis*), English ryegrass (*Lolium perenne*), common bent (*Agrostis capillaris*), and red fescue (*Festuca rubra*), and forbs species being white clover (*Trifolium repens*), common dandelion (*Taraxacum officinale*), annual blue grass (*Poa annua*), and common plantain (*Plantago major*). In fact, 94–97 % of all species in European lawns were indigenous. By comparison, results from lawn sampling in the Southern Hemisphere indicated that the percentage of indigenous species in lawn floras was highest in the tropics-subtropics or arid environments (e.g., Bolivia (80 %), South Africa (42 %)) and lower in temperate environments (e.g., Chile (20 %), Southern Australia (11 %) and New Zealand (19 %)) (Meurk 2004). South Africa had the greatest proportion of annuals/biennials in sampled floras (31 %) for the Southern Hemisphere. In the northern hemisphere UK had the highest proportion of annuals/biennials with 21 % (Stewart et al. 2009). These results suggested a homogenization of lawn flora around the globe as a result of globalization (see Sect. 10.6.2).

Lawns also differed from gardens with respect to management types and intensity. High species richness in gardens was attributed to management intensity, but intensive management in lawns reduced species richness. Falk (1980), studying only two lawns—one intensively managed (i.e., fertilized and irrigated) and mowed, and the other less intensively managed and just mowed—observed that the intensively managed lawn had 50 % fewer species than the less intensively managed lawns. A comparison of percent cover showed that turf grass species (e.g., tall fescue (*Festuca arundinacea*), Kentucky bluegrass, and Bermuda grass (*Cynodon dactylon*)) occupied nearly 90 % cover in the intensively managed lawn and only 70 % in the less intensively managed lawn. In addition, percent cover of dominant non-grass species differed between sites. In the intensively managed lawn, white clover dominated, whereas smooth crabgrass (*Digitaria ischaemum*) dominated in the less intensively managed lawn suggesting management intensity may also influence species occurrence and performance.

Examining the effect of lawn care further, Stewart et al. (2009) conducted a detailed analysis of Christchurch, New Zealand lawns and identified seven distinct communities. Each community reflected differences in lawn care such as mowing, irrigating, removal of clippings, and litter accumulation rather than environmental and social variables. Primarily, species richness declined significantly with an increase in litter, lawn area, and loamy soil, and the presence of grass clippings. Hence, park lawns had lower species richness than residential lawns (Stewart et al. 2009). By comparison, since the 1980s lawn management in parks and gardens within many European cities has shifted towards practices that support biodiversity. For instance, instead of being cut 10–15 times annually, lawns are cut only twice per year. The shift in management has created species-rich meadows which contribute significantly to local biodiversity and reflected historical cultural landscapes (Kelcey and Müller 2011; Kümmerling and Müller 2012) (see Sect. 10.6).

10.3.3 Species Traits

The urban environment is a unique environment in which species are exposed to environmental effects that do not occur collectively in other ecosystems. Environmental effects include elevated soil and air temperatures due to the urban heat island effect, higher concentrations of heavy metals in the soil, atmospheric pollution, increased water stress, and greater nitrogen and calcium deposition (Grimm et al. 2008; Lovett et al. 2000; McDonnell et al. 1997) (see Box 10.2). These environmental effects, in the context of the urban morphology, affect not only the gains and losses of species (Pärtel et al. 1996; Williams et al. 2009), but also serve as a filter for specific plant traits and selective pressures on species adaptions and evolution (Hunter 2007).

Plant traits can play a critical role in the survivability of a species in an urban environment. Traits associated with plants growing in human settlements include being biennial or perennial, C-strategists (competitors), and wind-pollinated; flowering in mid-summer; reproducing by seed and vegetatively; dispersing by wind or humans; and having a high demand for light and nutrients (Lososová et al. 2006). As opposed to arable lands which are disturbed annually, urban sites (e.g., vacant lots and wastelands) tend to have irregular disturbances, which create a patch mosaic of various stages of successional development. These irregular disturbances favor biennial and perennial rather than annual species. Similarly, Müller (2010b) reports that the most common plant species in six large cities of the northern hemisphere were from grasslands and riparian habitats (Fig. 10.4). These species may have a pre-adaptation to the droughty and anaerobic conditions found in urban landscapes.

Based on Grime's (1979) plant life strategies, (Lososová et al. 2006) reported that C-competitors were being selected in urban landscapes. In addition, Chocholoušková and Pyšek (2003) observed CSR-competitors/stress-tolerators/ruderals, CS-competitors/stressors, and CR-competitors/ruderals as being the dominant strategies in their historical analysis of Plzeň, Czech Republic.

A similar pattern of traits are being identified in structural and compositional shifts towards wind-dispersed, fast growing, shade intolerant species in remnant forests in urban landscapes. Rudnicky and McDonnell (1989) re-inventoried a historic remnant forest in New York City. The site had not been cut in historic times. All stems ≥15 cm diameter at breast height were inventoried and mapped in the mid-1930s, and again, in 1985. In the 1930s, 70 % of the forest was composed of two forest types: a hemlock forest type and an oak forest type. In 1985, these forest types only occupied 30 % of the forest and a maple/cherry/birch type was the dominant forest type. The shift in structure and composition from large conifer and oak

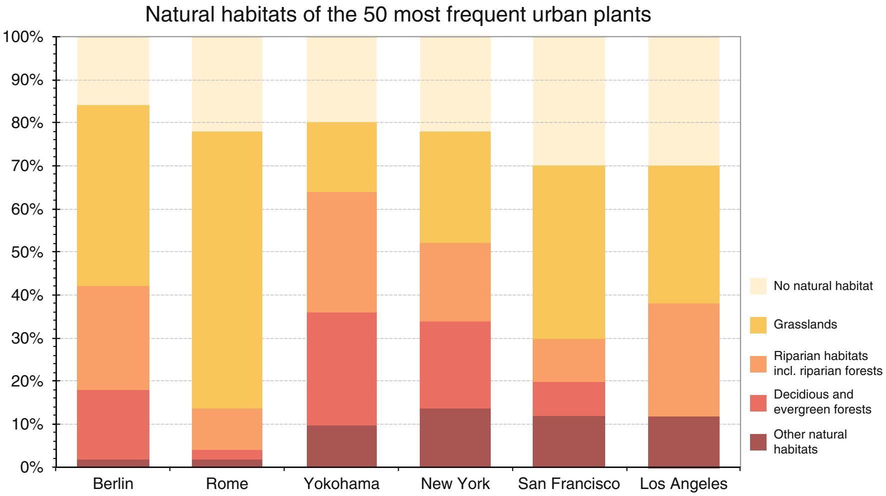

Fig. 10.4 Natural habitats of the 50 most frequent plants within six large cities of the Northern Hemisphere (Modified from Müller 2010b, p. 13. Published with kind permission of © Wiley-Blackwell 2010. All Rights Reserved)

species to wind-disseminated, fast-growing species was attributed not only to natural disturbances such as hurricanes but also to human activities such as arson, vandalism, and trampling. By comparison, the conifer and oak species were more susceptible to human disturbances than the wind-disseminated species (Rudnicky and McDonnell 1989). In addition, Godefroid and Koedam (2007) and Vallet et al. (2008) report a preponderance of nitrophilic species in remnant forest patches.

In addition to shifting structure and composition, the ecological novelty and the evolutionary consequences of the urban environment have altered genotypes of species. For instance, Wittig et al. (1985) and Müller (2010b) report a list of species specifically restricted to urban landscapes—anecophytes, species with no apparent natural habitat (Fig. 10.4) (also see Scholz 1991; Sukopp and Scholz 1997). Examples of such species include a shepherd's purse (*Capsella bursa-pastoris*), lambsquarters (*Chenopodium album*), Bermuda grass, mouse barley (*Hordeum murinum*), common plantain, annual bluegrass, prostrate knotweed (*Polygonum aviculare*), common groundsel (*Senecio vulgaris*), common chickweed (*Stellaria media*), and common dandelion, which are worldwide some of the most frequent plants in urban areas (Fig. 10.4) (Müller 2010b). Sukopp et al. (1979) report also that more than 15 species of primrose (*Oenothera* spp.) have evolved since the introduction of the American parent species 350 years ago in Europe. Similarly, since their introductions in Great Britain, Michaelmas daisies (*Aster novi-angliae*, *A. novi-belgii*, *A. lanceolatus*, *A. laevis* and hybrids) appear to be more variable both morphologically and in their ecological amplitude than the same species in North America (Gilbert 1989). Over time, new species will evolve through natural selection and hybridization, and novel ecosystems will continue to develop, potentially changing ecosystem benefits (both positively and negatively) affecting humans in urban landscapes (see Chap. 11).

The dominance of non-native species in urban landscapes has led to the evaluation of species traits to identify why invasives are so successful in urban landscapes. The identification of these traits, however, can be problematic because of the approaches used in the analyses; the types of comparison, scale, and data character used; and what constitutes the occurrence of a species (Pyšek and Richardson 2007). Accounting for these factors, Pyšek and Richardson (2007) observe several general patterns when comparing non-native species to native species: faster growth, taller plant height, more vegetative reproduction and lateral growth, more often hermaphroditic, earlier germination and germination under a wider range of conditions, higher water, nitrogen and/or phosphorous use efficiencies, and more extended periods of flower timing. For pollination and dispersal, no differences are observed, and mixed findings occur with respect to seed size. In their analyses Pyšek and Richardson (2007) point out that traits do matter, but caution that traits that are successful at one stage of the invasion process and in a specific habitat may be neutral and even detrimental in other stages.

In their landscape designs and management, many cities, landscape firms, and nurseries are moving away from non-native species and going native (Ignatieva et al. 2008). The current thought is that native species are adapted to the region and will be better suited for plantings than non-native species. Like all species, native species have evolved to live in a set of environment conditions involving soil moisture, temperature, nutrient and light availability, and shade tolerances. The suite of these environmental conditions needed by a species may not be collectively represented in the urban landscape. Consequently, native plantings often fail because species are not adapted to the urban environment—it is the wrong plant in the wrong place (Quigley 2011). Nonetheless, matching the right native species for the right place can improve survivability and enhance native species representation in urban landscapes and design.

10.4 Animals: Vertebrates and Invertebrates

The previous section examined how the urban environment influenced plant species richness, patterns, and distributions. This section focuses on vertebrates and arthropods. First, the section examines humans, the dominant mammal in urban systems and then evaluates how other mammals, birds, reptiles and amphibians, and arthropods respond to urbanization.

10.4.1 The "Other" Mammal

Homo sapiens is the dominant mammal in urban and urbanizing landscapes. Often one does not think of humans as being part of the ecological system, but rather views humans as the cause of environmental degradation, habitat fragmentation, and altered disturbance regimes (see Chap. 2). Section 10.5 examines socio-ecological systems and the reciprocal effects that ecological and social systems have on each other in greater detail. In this subsection *Homo sapiens* are highlighted as the keystone species in urban systems. Humans modify nearly every aspect of our abiotic and biotic environment through the following behaviors: (1) constructing barriers to dispersal, (2) fragmenting habitats, (3) introducing non-native species, (4) introducing domestic pets, (5) altering ecosystem structure and processes, (6) altering disturbance regimes, (7) changing competitive relationships and trophic structure, and (8) generating multiple-scale effects (Adams and Lindsay 2011). An artifact of this modification is the built infrastructure to sustain human activities.

Infrastructure creates habitat for some wildlife species but may present hazards for other species. For instance, buildings serve as nesting habitats for raptors, but also are obstacles to migratory birds. More than 100 million migrant and resident birds are estimated to be killed each year by colliding into windows (Adams and Lindsay 2011). Similarly, collisions with communication towers cause approximately 1.2 million birds deaths annually (Adams and Lindsay 2011). Roads also pose a major threat to wildlife species, especially small and slow moving fauna (Forman and Alexander 1998). For instance, for 12 linear kilometers of roads in Tippecanoe County, Indiana, USA, Glista et al. (2008) recorded 10,515 road kills in a 17 month period; over 9,100 of those deaths were anuran (frogs and toads) species. The high anuran mortality was attributed to individuals migrating to and from breeding sites. In contrast, bridges, underpasses, overpasses, and culverts serve as nesting and roosting sites for a number of species. For instance, both cliff swallows (*Hirundo pyrrhonota*) and cave swallows (*H. fulva*) have expanded their natural ranges by adopting bridges and culverts as nesting sites, and more than half of the 45 bat species in the United States use bridges as roosting sites (Adams and Lindsay 2011). Overall, the adaptability of a species to human infrastructure and landscape mosaic often determines its survivability in the urban landscape.

10.4.2 General Observations

Like plants, a general set of characteristics enable wildlife species to survive and possibly flourish in urban landscapes. They include: (1) physiological tolerance to extreme variation in the abiotic environment; (2) large zoogeographic distribution; (3) generalists rather specialists with respect to available food, shelter, and water resources; (4) high reproductive and survival rates; (5) habituation to human activities; (6) few competitors and/or predators; (7) adaptability to highly fragmented landscapes with abundant edges; and (8) high rates of recruitment through immigration (Adams and Lindsay 2009). In addition to these characteristics, habitat quality and availability play key roles in determining whether wildlife species will be present (Nilon 2009). Because humans control land uses and land covers in the urban matrix, habitat conservation must be coupled with urban planning and landscape design. Furthermore, because of the complexity of the urban landscape, conservation strategies must involve not only the habitat itself, but also the ecological context of the parcel, such as its connectivity, distance to other habitats, distance to water features and potential buffer zone to reduce anthropogenic influences if habitat quality is to be maintained too (Yli-Pelkonen and Niemeliä 2005). Planning tools, such biotoping, do exist to account for the intrinsic quality of landscape features and habitats (e.g., Douglas 2011; Löfvenhaft et al. 2002). A biotope is a mappable area with homogeneous environmental characteristics and biological communities. It also can be linked with social attributes such as income, home ownership, and ethnicity.

To assess amphibian species responses to urbanization, Hamer and McDonnell (2008) presented a hierarchical framework; this section extends this framework to generalize vertebrate responses to urbanization. Four critical components of the framework are: (1) habitat availability, (2) habitat quality, (3) species availability, and (4) species responses. The first two components identify key effects of urbanization, whereas, the latter two components are key responses and adaptations to urbanization (Hamer and McDonnell 2008). Each component has a subset of attributes that can influence vertebrate species richness and community structure. For instance, habitat loss, fragmentation, isolation and restoration affect habitat availability in an urban landscape. Likewise, habitat quality depends on vegetation structure and composition, patch configuration and context, hydrologic process and hydroperiods, presence or absence of native and non-native predators and competitors, water quality and pollution, diseases, and human disturbances and climate change. Important components for species availability include geographic range, dispersal, and demography. Life history and species attributes, response thresholds, and metapopulation dynamics play critical roles in species responses (Hamer and McDonnell 2008). The following subsequent sections use this framework to examine the effect of urbanization on mammals, birds, amphibians and reptiles, and arthropods.

10.4.3 Mammals

While there are a few studies on mammals in urban landscapes, there are a number of survival traits that have been identified for mammals in urban landscapes. Traits include commensalism, omnivory, and being habitat generalists and/or edge species (Riem et al. 2012). Examples of successful urban mammals include the raccoon (*Procyon loctor*), gray squirrels (*Sciurus carolinensis*), red fox (*Vulpes vulpes*), and Norway rat (*Rattus norvegicus*). As with native plants, native mammal richness and abundance generally declines with increasing levels of urbanization because of habitat loss, degradation, and isolation. There are, however, exceptions. For instance, with moderate levels of urbanization, abundance of native species may actually increase for different reasons including high habitat and spatial heterogeneity; altered habitat productivity, predator-prey associations, and disturbance regimes; and socio-economic factors such as supplemental feeding (Shochat et al. 2006).

To highlight the effect on urbanization on mammals, two urban settlements—Oxford, Ohio, USA, a small urban town in the United States with a human population of less than 22,000; and Buenos Aires, Argentina, a large city with a population of over 2.7 million—were compared. Riem et al. (2012) examined mammals along an urban-rural gradient in Oxford and observed that the greatest diversity and richness occurred with moderate levels of urbanization (a similar pattern observed for birds and butterflies). In fact, 7 of the 11 species inventoried occurred in the urban matrix. This finding implied that these species were adapting to the urban environment's supplemental food sources and additional cover from the built infrastructure. Overall, mammals responded to the juxtaposition of natural and human elements rather than the degree of urbanization (indicated by factors such as percent impervious surface) (Riem et al. 2012). In addition, Riem et al. (2012) reported that mammalian diversity and richness did not change rapidly but rather gradually with urban development.

Unlike Riem et al. (2012) and Cavia et al. (2009) observed a linear decline in species richness and diversity of rodents with increasing urbanization for Buenos Aires, Argentina. Seven species were sampled. Native species (4) were dominant on sites with natural vegetation, whereas non-native species (3) were dominant in shantytowns, industrial sites, and residential neighborhoods. The difference in distribution was attributed to spatial heterogeneity of the landscape and different urban environments. Native species richness declined as remnant habitats became more fragmented, isolated, or destroyed, whereas non-native species richness increased as new habitats were created with urbanization.

Although the comparison between Riem et al. (2012) and Cavia et al. (2009) is limited, the two studies illustrate the importance of spatial heterogeneity and landscape configuration of the urban matrix as they affect mammal species richness and diversity. In general, patch density (i.e., different types of land cover and land use per square kilometer) and edge density (i.e., total length of all edge segments per hectare) increase, whereas landscape connectivity decreases as the human population of a urban landscape increases (Luck and Wu 2002; Wu et al. 2011). Hence, towns and villages are less spatially heterogeneous than large cities, thus creating a more hospitable environment for native mammalian species, a pattern that is also observed for native plant species.

In Melbourne, Australia, van der Ree and McCarthy (2005) report that small, ground-dwelling mammals are extirpated from urban landscape not only because of habitat loss, but also simplification. In rural woodlands, fallen logs and branches are used by small-ground dwelling mammals as protection from predation. In urban woodlands, these habitat components are often removed for human safety and to reduce fire risk. This removal increases an individual's exposure to predation, thus reducing population density and species richness. Baker et al. (2003) also report on the effect of predation, principally by the domestic cat (*Felis catus*), on small mammal densities. In Bristol, UK, they observe that a cat kills 21 prey items per year. For the United States, Loss et al. (2013) estimate that free-ranging cats kill 1.4–3.7 billion birds and 6.9–20.7 billion mammals per year. Obviously not all of these losses occur in urban and urbanizing landscapes. Nonetheless, cats can kill a significant number of birds and mammals and significantly affect native species density and richness.

Native, carnivorous, mammalian species richness generally declines with urbanization; however, to fully assess the effect of urbanization on carnivores, this group needs to be divided into apex predators (e.g., large species such as wolves (*Canis lupus*)) and mesopredators (Prugh et al. 2009). As with many native species, apex predators quickly fall "prey" to habitat fragmentation and loss, reduction of landscape connectivity, and increase in road density caused by urbanization. By comparison, mammalian mesopredators, often omnivores, have adapted well to the highly fragmented urban landscape and have substantially increased in abundance in the absence of apex predators (a phenomenon known as mesopredator release) and due to an increase in food supply (Prugh et al. 2009). This increase of mammalian mesopredators has shifted trophic structures (Faeth et al. 2005) and has been detrimental to small prey species—especially ground nesting species—in urban landscapes.

10.4.4 Birds

Birds are the most studied vertebrate in the urban landscape. Marzluff et al. (2001) reviewed over 100 papers from 1990 to 2000 addressing birds in urban and urbanizing landscapes. Even with the high volume of studies that exist, urban effects on birds still need to be documented more extensively and more widely across regions in the world, especially in tropics. Furthermore, most studies focus on how avian community structure and composition changes with urbanization, but offer limited insights into the causal factors for those changes. This section highlights the salient patterns of avian species richness and diversity in urban landscapes and the mechanisms driving those patterns.

Patterns of bird species richness and diversity in urban landscapes result from individual responses as well as habitat quality and availability, and regional metapopulations. In his analysis of a land-use gradient in Santa Clara County, California, USA, Blair (1996) observed that native species richness declined and non-native species increased as sites became more urbanized. Because of the addition of non-native species, Blair also reported that overall species richness and diversity was highest in moderate (suburban) levels of urbanization (Fig. 10.5). Although non-native species contributed to species richness in moderately urbanized sites, their richness actually declined with increased development. Interestingly, even in the business district, native species flourished—e.g., the White-throated swift (*Aeronautes saxatalis*). A cliff dweller, the swifts are apparently using the tall buildings for nesting habitat. Similar observations have been reported for cliff dwelling raptors such as the Peregrine falcon (*Falco peregrinus*) and other raptors (e.g., Ospreys, *Pandion haliaetus*) using artificial structures for nest sites. In fact, urban landscapes can be superior habitats for raptors because they are often free of human persecution and have high availability of abundant food (Chace and Walsh 2006).

Blair and Johnson (2008) looked more closely at species richness under moderate level of urbanization to assess potential mechanisms. Studying three locations—Oxford, Ohio; Saint Paul, Minnesota; and Palo Alto, California, USA—they

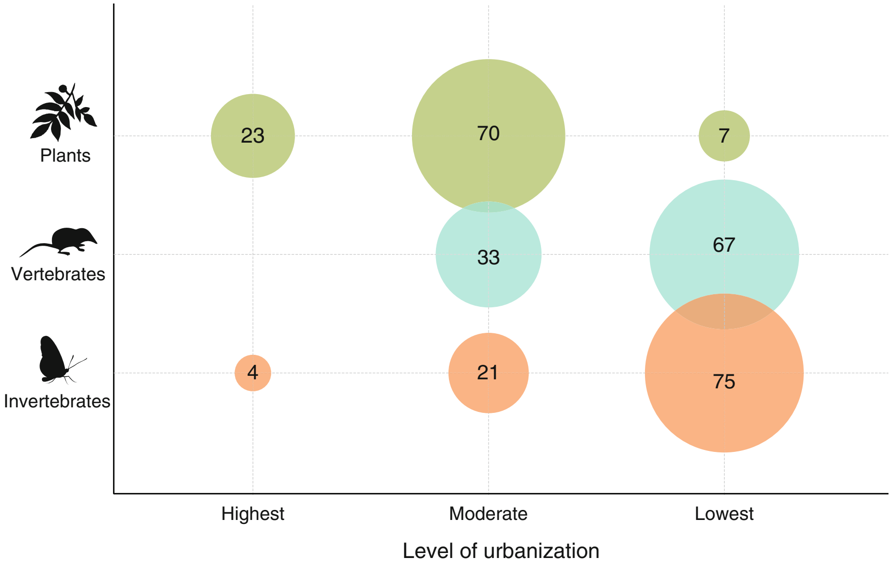

Fig. 10.5 Percentage of studies, by group, showing species richness peaks at three levels of urbanization (Modified from McKinney 2008, p. 166. Published with kind permission of © Springer 2008. All Rights Reserved)

observed that species richness, diversity, and evenness increased with moderate levels of urbanization and then decreased with more intensive urbanization. Interestingly, the moderate level of urbanization was also the inflection point for a shift in species community—the decline of native woodland species and the increase in ubiquitous, invasive, urban-exploiting species (Blair and Johnson 2008). As in the trend with plants, this shift to a dominance of invasive, non-native species represented a pattern of biotic homogenization of species accompanying increases in urbanization. Blair and Johnson (2008) also observed a shift in functional traits with urban species having multiple broods per year, nests on buildings, eating seeds, residing year-round, and tending to be non-territorial. By comparison, woodland species tended to have a single brood per year; nest in trees, shrubs, and snags; eat insects; migrate; and display territorial behavior. This pattern, however, changed with biomes (Chace and Walsh 2006). For instance, in desert landscapes, urban-avian communities were dominated by species that are seed eaters, ground foraging insectivores, water-dependent, and crevice-nesting.

In an analysis of habitat quality, structure and spatial pattern, Donnelly and Marzluff (2006) showed that for bird diversity, habitat quantity was more important than habitat pattern (e.g., patch shape and size and forest aggregation), and habitat structure was as important as habitat pattern. In general, retention of native bird species richness in Seattle, Washington, USA was achieved by limiting urban-land cover to levels <52 %, and by maintaining tree density (9.8 trees/ha), an evergreen presence (23 % of forest cover), and a forest not highly fragmented (>64 % aggregated) (Donnelly and Marzluff 2006).

10.4.5 Amphibians and Reptiles

Of the world's vertebrates, amphibians have the greatest proportion of species (21 %) on the verge of extinction (Stuart et al. 2004). By comparison, the proportion of endangered species for mammals and birds are 10 and 5 %, respectively. Although explanations for the world's decline in amphibians are limited, it is suggested urbanization is a major factor causing their decline (Hamer and McDonnell 2008). Unfortunately, the majority of studies examining the effect of urbanization on amphibians have been conducted primarily for temperate regions; similar studies need to be conducted for urban landscapes in tropical regions. Nonetheless, Hamer and McDonnell (2008) and Garden et al. (2007) report that the effect of urbanization on amphibians ultimately depends on life-history attributes, sensitivity to environmental changes, interspecies interactions, and dispersal requirements of the individual species composing regional populations.

Many amphibian species have a patchy distribution across landscapes creating a large network of metapopulations at the regional scale (see Pope et al. 2000). In addition, many amphibian species require complementary habitats at multiple scales to complete their complex life cycle. By disrupting dispersal through the construction of roads, buildings, fences and other barriers, urbanization reduces the functionality of these patchy networks of metapopulations (Pope et al. 2000). Consequently, amphibian species richness generally declines with increases in urbanization primarily due to changes in landscape structure and complexity (Garden et al. 2007). In their review of the literature on amphibians in urban and urbanizing landscapes, Hamer and McDonnell (2008) report that landscape changes include decreases in wetland area and density, increased wetland isolation, as well as decreases in wetland vegetation, forest cover, and other significant upland habitats.

Because of their broad habitat requirements, some amphibian species persist in urban landscapes. For instances, Carrier and Beebee (2003) report that the common frog (*Rana temporaria*) actually persist better in Britain's urban and suburban areas than in rural areas because of the greater abundance of garden ponds, which are used by frogs for breeding. Even if wetland habitats are present, water quality plays a critical role in their suitability for amphibians. Pesticides, fertilizers, road salt and oil, sediments, and heavy metals in stormwater runoff can drastically affect water quality (Rubbo and Kiesecker 2005). In fact, because of degraded water quality, urban wetlands may actually act as habitat sinks and possibly deplete regional metapopulations (McKinney 2002; Battin 2004).

Alteration of hydrologic processes, especially hydroperiod (i.e., the length of time a waterbody, wetland or stream continuously hold water) by urbanization can have profound effects on amphibian communities and species richness (Werner et al. 2007). For instance, Pearl et al. (2005) report that the shift from ephemeral wetlands to stable permanent wetlands in the Portland, Oregon, USA resulted in a shift from amphibians with rapid larval development (e.g., long-toed salamander, *Ambystoma macrodactylum*) to those species with longer larval development (e.g., bullfrog, *Rana catesbeiana*). Similarly, populations of stream salamanders in North Carolina, USA have drastically declined with urbanization because of the increase in magnitude of stream flow and sedimentation due to the increase in impervious surfaces in watersheds (Price et al. 2006). These and other studies show that the conservation of amphibians in urban and urbanizing landscapes requires the prevention of habitat loss and degradation (both aquatic and terrestrial), maintenance of regional metapopulations, and preservation of connectivity among habitats.

Unlike with amphibians, there is dearth of studies and reviews examining reptile species richness in urban landscapes. This section uses a global analysis of reptiles and site-specific studies to discern patterns in urban and urbanizing landscapes. In their global review of 1,500 reptile species, Böhm et al. (2013) identify a similar suite of anthropogenic threats associated with amphibians affecting reptiles. They include habitat loss, degradation, and fragmentation; invasive species; accidental mortality (e.g., road kills); altered trophic structures; and altered disturbance regimes. In addition, reptiles are frequently harvested for food and intentionally killed because of human aversions. However, because of the magnitude and scale of change, agriculture and logging pose even greater threats to reptiles than urbanization (Böhm et al. 2013).

To assess the effect of urbanization on amphibian and reptiles, Barrett and Guyer (2008) examined stream- and riparian- dwelling amphibians and reptiles in eight catchments in Chattahoochee Watershed of western Georgia, USA. They observed that amphibian species richness declined, but reptile species richness actually increased with urbanization of the watershed. Urbanization shifted conditions from a closed-canopy, shallow-water habitat, favored by salamanders and frogs, to a habitat characterized by open vegetation and deeper, warmer, and open water, conditions favored by turtles and snakes. A similar pattern was observed by Hunt et al. (2013) who reported that percent of urban land use had little effect on the occurrence of reptiles and individual species. Rather, habitat availability and quality determined species richness.

Using historic sighting records in wildlife databases, Hamer and McDonnell (2010) inferred the probability of persistence of amphibians and reptiles in Melbourne, Victoria, Australia for the period from 1850 to 2006. Their analyses showed a significant decline in both amphibians and reptiles, but urbanization had a greater effect on the persistence of reptiles than frogs. As indicated by van der Ree and McCarthy (2005) for small mammals, habitat simplification was attributed to the reduced persistence for reptiles. Hamer and McDonnell (2010) reported that there were fewer fallen logs and a loss of vegetation strata for reptiles in remnant forest patches to carry out their daily and seasonal activities. Garden et al. (2010) also report that local habitat composition and structure, as well as landscape composition and configuration of lowland remnant forests, had the greatest influence on reptile communities in Brisbane, Australia. They report that species richness discrepancies among studies were attributed to single-scaled studies as compared to multiple-scaled studies and to the physiological and behavioral characteristics of the species.

10.4.6 Arthropods

Arthropods are probably the least understood phylum in the urban landscapes, yet it is likely that they have the greatest effect on society. They provide critical ecosystems services such as pollination and pest control, while at the same time, they are considered a bane because of many factors including disease transmission, human discomfort (e.g., biting, stinging and sucking), and crop and horticultural damage. Because of the availability of studies, this section focuses principally on insects.

As one might expect with the diversity of insects, there is a range of responses to urbanization. McIntyre (2000) identified three groups of arthropods with respect to different levels of urbanization: (1) rural taxa (not present or low occurrence in urban landscapes), (2) urban taxa (principally found in urban landscapes or have a high abundance there), and (3) taxa found abundantly in both rural and urban landscapes. In her review she also identifies air, water, and thermal pollution as well as succession development as important drivers not only of arthropod occurrences, but also trophic structures. For instance, the urban environment may stress plants, which respond physiologically, and subsequently change their susceptibility to herbivores and sucking insects (Schmitz 1996). Similarly, the urban heat island may enable arthropods to occur at more northern latitudes than otherwise possible in rural landscapes. Gilbert (1989) reports that habitat age influences arthropod diversity. In a study of vacant lots, he observes that younger lots have fewer species and less diversity than older vacant lots, and species taxa and abundance shift from younger to older lots. Overall, terrestrial arthropod communities in urban environments (non-native species included) tend to be more diverse than those in rural environments (McIntyre 2000). In general, herbivores are more abundant in cities than rural sites. On the other hand, parasitoids tend to be more abundant in rural than urban sites. Bennett and Gratton (2012) observe that the occurrence of parasitoid wasps is directly related to flower density, but declines as impervious surface area increases (i.e., less space for gardens). Likewise, generalists tend to occur more frequently than specialists in cities (e.g., carabid beetles (Niemelä et al. 2002) and parasitoid wasps (Bennett and Gratton 2012)). For aquatic systems, the diversity of aquatic insects in streams often declines with increasing urbanization (Jones and Clark 1987).

Although differences in arthropod diversity occur between urban and rural landscapes, McIntyre (2000) points out the need to distinguish between numeric and proportionate changes in arthropod abundance with respect to urban effects. Numeric change refers to a change in absolute number, whereas proportionate changes refer to a change in a taxon's importance in respect to the overall assessment of diversity. With these differences, McIntyre (2000) hypothesizes the following patterns: (1) arthropod diversity decreases with increasing air and water pollution, (2) diversity increases with the age of urbanized area, (3) juxtaposition to native habitats plays an important role for recruitment and dispersal into new habitats, and (4) diversity of non-native species increases with the age of urban area.

10.5 Social-Ecological Perspective on Urban Biodiversity

Humans drive urban systems. As obvious as this statement is, only recently has there been a concerted effort by ecologists and sociologists to truly examine the complexity of socio-ecological interactions in urban landscapes (Cilliers 2010; Kinzig et al. 2005; Liu et al. 2007) (see also Chap. 33). This is not to say that ecologists have neglected to evaluate how urbanization affects ecological structure and function (such as biodiversity or how the natural environment is important to social systems). In fact the literature is replete with studies that examine the ecology *in* cities—how urbanization affects the abiotic environment and, in turn, the subsequent effects on biotic structure and function. A number these studies are highlighted in the previous sections. This section highlights how social and ecological systems interact to create patterns of biodiversity—the ecology *of* the city.

The emphasis on socioeconomic differences as drivers of biodiversity builds on social science theory that put forward the concept that social and spatial inequalities may drive patterns of similarity or difference within cities. In North America, work by Park (1915) and Park et al. (1925) stress that patterns of social, ethnic/racial, and economic inequality and immigration into cities create different zones that have unique characteristics with in a city. This focus on distinct zones provides the foundation for social areas analysis, an approach used by geographers to study different cultural groups and patterns of differentiation and inequality within and across cities (Shevky and Bell 1955; Drake and Cayton 1945). Contemporary studies using social areas analysis define socioeconomic status as a composite scale to indicate patterns of family income, education, occupation, and family structure (Maloney and Auffrey 2004). These approaches stress the role of economic and sociocultural changes that lead to distinct and new patterns of urbanization and result in changes in the spatial pattern of the built environment of cities (Cilliers 2010; Knox 1991). The concept of environmental justice, which became popularized at the beginning of the 1980s, starts with discussions about the unequal distribution of environmental harms like toxic waste, water and air pollution in relation to several socio-economic groups (Schlosberg 2007). Now, this concept includes biodiversity decline too, and with respect to urban areas, terms like "biological poverty" have been created (Melles 2005).

Ecologists have studied the relationship between urban biodiversity and socioeconomic patterns in cities since the 1970s. Schmid's (1975) study of vegetation in neighborhoods in Chicago, Illinois, USA, related patterns of tree species richness to census tract block data for the neighborhoods. Whitney and Adam's (1980) research on street and yard trees and Talarchek's (1990, 1985) study of street trees in New Orleans are examples of similar studies of street and yard trees that sought to identify census and other socioeconomic predictors of species richness. Hard (1985) produced and compared two urban maps of the city of Osnabrück (Germany). One map represents the socio-economic distribution of the human population and the other map demonstrates the distribution of plant communities. The comparison reveals that the both distributions are closely linked. Significantly, all these studies attempted to relate patterns of biodiversity to specific types of neighborhoods, thus building on ideas that were linked to theories about differentiation and spatial patterns in cities.

Since the mid-1980s ecologists and social scientists developed and tested theories about relationships between urban biodiversity and socioeconomic status. Palmer (1984) and Richards et al. (1984) studied the vegetation of residential lots in several Syracuse, New York, USA neighborhoods and developed the concept of neighborhoods as areas with discrete vegetation shaped by residents and their preferences, with those preferences shaped in part by socioeconomic status. Burch and Grove (1993) and Grove and Burch (1997) hypothesized that gender, property rights, technological change, and other variables might influence urban residents' decisions about managing urban vegetation and in turn create patterns of difference in urban vegetation within a city.

A number of models have been proposed to integrate social and ecological patterns and processes (see Alberti et al. 2003; Grimm et al. 2000; Pickett et al. 2001). These models build upon a system ecology approach to identify flows of energy, species, materials, and information across the urban landscape and discern how they are mediated by different social institutions, cultures, contexts, and human behavior (Alberti 2008). Nonetheless, they generally are biocentric, focusing primarily on the effect of social systems on ecological patterns and processes; only to a lesser extent do they explore how ecological systems influence social patterns and processes. Morse (2007) and Zipperer et al. (2011) build upon these models by integrating the concept of complex adaptive systems (Gunderson and Levenson 1997) and structuration theory (Scoones 1999). In doing so, they account for social and ecological systems that operate differently across spatial and temporal scales, and how actions and outcomes affect not only the respective systems but also the feedback loops between systems.

An important component of socio-ecological models is the scale (e.g., broad and fine) at which decision making processes are made and the subsequent effect on biodiversity. Kinzig et al. (2005) propose a social gradient similar to the ecological urban-rural gradient to capture changes in social patterns and processes, and recognized the importance of the scale of management: top-down and bottom-up. Top-down decisions reflect the broad scale of city-level management strategies and decisions affecting public lands such as parks, transportation corridors, and street-trees across a broad scale. In contrast, bottom-up decisions reflect the fine-scale decisions of private land owners and small-scale actions and outcomes. Although top-down decisions establish the rules and regulations for land usage and conversions, bottom-up decisions can collectively have a pronounced effect on local structure and connectivity in a neighborhood that varies by socioeconomic and cultural characteristics (Kinzig et al. 2005). The combined actions of top-down and bottom-up decisions across social and ecological gradients create the habitat mosaic and species distribution in urban landscapes.

In their analysis of socio-ecological drivers of plant and avian biodiversity in the metropolitan area of Phoenix, Arizona, USA, Hope et al. (2003) and Kinzig et al. (2005) found median income to be the most significant bottom-up influence on plant biodiversity in neighborhoods. Higher-income neighborhoods contained a greater biodiversity than lower-income neighborhoods in this desert city. Melles (2005) for Vancouver, Canada and Strohbach et al. (2009) for Leipzig, Germany observed similar patterns of vegetation and avian biodiversity. In contrast, Grove et al. (2006) observed lifestyle behavior, as derived from a marketing classification system called PRIZM, to be a better indicator of vegetation patterns for Baltimore, Maryland, USA, a temperate city. However, the use of PRIZM data to classify social systems and their corresponding relationship to ecological structure and function has not been fully documented and may not be appropriate for cross comparisons of social systems in different countries (McFarlane 2006).

Andersson et al. (2007) examined the importance of management scale on diversity and subsequently on ecosystem services in Stockholm, Sweden. They focused on three types of green spaces in Stockholm, Sweden: parks, managed by the city; cemeteries, generally managed by the Church of Sweden; and allotment gardens, managed by individuals. Those systems managed by individuals—bottom up management—had the greatest diversity and abundance of pollinators and a different suite of seed dispersers and insectivores than systems managed by the city and the Church of Sweden—top-down management. Scale of management translated into contrasting ecosystem services for local residents (Andersson et al. 2007).

The effect of bottom-up influence on biodiversity can be considerable and important in identifying social feedback loops in socio-ecological systems across institutional scales (Ernstson 2013). For instances, Cilliers et al. (2011) used urban domestic gardens effectively in the North-West Province, South Africa, a province with one of the lowest level of quality of life in South Africa, to maximize community involvement, increase food production, and conserve adjacent natural grasslands by providing an alternative to clearing natural habitats for farming. To enhance participation by residents, the gardens were placed around houses. After a period of time, researchers returned to the community and found the gardens removed. Through discussions with homeowners, researchers learned that planting around a home conflicted with a cultural belief that the area around houses should be open and devoid of vegetation. Even though residents benefited from these gardens, the strong cultural belief of removing vegetation adjacent homes created a negative feedback to improving the lives of residents and conserving adjacent natural areas. To address cultural beliefs and other challenges, social and environmental educational programs were developed to increase residents' awareness of the costs and benefits of urban domestic gardens (Cilliers 2010). This example illustrates the complex relationship between social and ecological interactions in our urban landscapes.

Knowledge of the interplay between social and ecological systems in urban landscapes becomes increasingly important as the world population becomes increasingly urban. In fact, the socio-economic systems of an urban landscape influence not only species richness, but also how species are distributed and how species coexist (Swan et al. 2011). In urban landscapes, the social factors that directly or indirectly control and influence biodiversity include (1) ownership and its organizational structure, (2) access to and control of the land and its resources that species require; (3) the financial resources and social dynamics (or lack thereof) that affect management, and (4) the knowledge—traditional and/or academic—used to design and manage land cover. Acknowledging the interplay between social and ecological patterns and processes, and the influence of urban environmental filters (sensu Williams et al. 2009), Swan et al. (2011) propose the use of a metacommunity approach to explore how local versus regional processes may shape community structure and composition by organizing species distributions into two extreme assemblages—self and facilitated. Self-assemblages are species patterns responding to disturbances and environments created by humans but species occurrence is not directly manipulated by humans. Species composition is the consequence of human activities and decisions about how urban landscapes are physically structured (Swan et al. 2011). Examples of created or modified habitats by humans include vacant lots, abandoned properties, roadside verges, railroad beds, and retention ponds (see Sect. 10.3.2). Both the ecosystem-stress hypothesis (Menge and Southerland 1987) and the intermediate disturbance hypothesis (Connell 1978) have been used to explain species occurrence, richness and pattern in these habitats. Nonetheless, it is the socio-economic context of the decision-making processes, which ultimately drive the patterns of environmental and social constraints, that creates these assemblages (Swan et al. 2011).

Facilitated-assemblages result directly from human placement and manipulation through landscape design. Socio-economic factors decide and control what species are present (i.e., desirable) or absent (i.e., undesirable) on a site. The most obvious habitats are private gardens and lawns (see Sects. 10.3.2 and 10.6.2).

Through maintenance and management, desirable species can survive outside of their natural ranges and habitats (Swan et al. 2011). Similarly, the environmental filters, which created self-assemblages, are mediated by humans to create an environment conducive for desirable species composition and structure. With facilitated assemblages, there is a strong social desire for particular species to persist and undesirable species to be removed (Swan et al. 2011).

10.6 Influence of Landscape Design on Urban Biodiversity

This section reviews the literature on globalized trends in landscape architecture since the second part of the twentieth century and the consequent effect on biodiversity. Emphasis is given to existing case studies of modern, alternative-ecological design, which reinforces the reintroduction of native plants into green areas, the support of native biodiversity and the development of a sense of place.

10.6.1 The Global Extension of European Landscape Design Styles

The most influential landscape architecture styles, recognized globally, are simplified versions of English landscape and Gardenesque styles. Primarily during the Victorian age, these two styles were brought by Europeans to the New World to

Fig. 10.6 (**a**) Chatsworth Park in England provides an example of the original English landscape style. (**b**) A public park in Adelaide, Australia illustrates the simplification of that style (Photographed by and published with kind permission of © Maria Ignatieva 2013. All Rights Reserved)

change the landscape into something familiar to the colonists. The most dramatic influence was the introduction of numerous non-native plants, birds and mammals, which often altered the local biodiversity. In fact, the Victorian era was a time of large-scale exchanges of plants from new lands and introduction of these plants to private and public gardens (Thacker 1979). Elements such as lawn and carpet flower beds (as a special display for numerous exotic plants) were popular not only in European countries but also in all British colonies (such South Africa, Asia, New Zealand, and Australia).

The English landscape style of the eighteenth century followed the fundamental designs of the Picturesque Movement, a landscape design approach formulated and based on the variety and irregularity of nature. By the end of the eighteenth century and the beginning of nineteenth century, this movement reshaped not only the English landscape but also much of Europe and colonial countries, regardless of climatic conditions. This landscape style was characterized by curvilinear shapes, gentle rises and hills, bright green grass and scattered groves, woodlands or single deciduous trees, and romantic bridges and pavilions with scenic views. Frederick Law Olmsted, the famous American landscape architect who is often referred to as the "father of landscape architecture", literally created parks around the world that adhered to the English style. His designs became a widespread, western approach for designing urban parks (Schenker 2007). Unfortunately, many modern parks have lost the original intent of the English landscape style and its symbolism and spirituality, and are represented by a very simplified structure—lawn with scattered groups of trees and single trees, a pond or lake, and curvilinear pathways (Fig. 10.6).

The Gardenesque style, which succeeded the Picturesque Movement, had even a greater influence on Western landscape architecture style than the English landscape style. The Gardenesque style evolved during the industrial revolution in Europe and the Victorian era, and was characterized by artificiality and extravagancy, which was directly opposite of the English landscape style, which celebrated naturalness (Zuylen 1995). The Gardenesque style, typified by eclecticism in

Fig. 10.7 Modern examples of flowerbeds across the world that illustrate the Gardenesque landscape style. Photographs shown are from (**a**) Mumbai, India; (**b**) Shanghai, China; and (**c**) Brisbane, Australia (Photographed by and published with kind permission of © Maria Ignatieva 2013. All Rights Reserved)

architectural and landscape styles, preferred the use of non-native plants, the development of botanical-garden displays, and the occurrence of glasshouses with unusual palms, ferns, cacti, and other tropical and subtropical plants. Eclecticism in landscape style means the integration of different traditions of formal gardens with their straight lines and topiaries, and the introduction of unusual or exotic buildings and plants. Current examples of Gardenesque gardens across the world are a simplified version of those from the Victorian time. Most of these gardens have lost the original style and innovative character of their historical cousins. Today, these gardens are characterized by 'pretty', 'tidy', 'colorful' and 'beautiful' homogeneous landscapes based on non-native plants. Examples can found in temperate as well as tropical climates (Fig. 10.7).

10.6.2 Globalization of Plant Material

The ubiquity of Gardenesque style gardens throughout the world has actually created a market of available plant material that is quite similar worldwide. Ignatieva (2011) analyzed nursery catalogs from temperate zones in the United States, New Zealand, Russia, Germany, and found a high degree of similarity among available plant material regardless of location. This homogeneity, deemed "unification" of plant material on a global scale, results from planting designs creating a pool of "chosen" plants. Favorable "chosen" plants in temperate zones were European deciduous trees and shrubs and some "fashion" conifers. These global plants can be linked to English landscape and Gardenesque garden styles at the end of the nineteenth and beginning of the twentieth centuries. Popular plants included pines (*Pinus* spp.), spruce (*Picea* spp.), Lawson cypress (*Chamaecyparis lawsoniana* cultivars), junipers (*Juniperus* spp.), cedars (*Thuja* spp.), birches (*Betula* spp.), cherries (*Prunus* spp.), willows (*Salix* spp.), poplars (*Populus* spp.), oaks (*Quercus* spp.), elms (*Ulmus* spp.), maples (*Acer* spp.), ashes (*Fraxinus* spp.), and rhododendrons (*Rhododendron* spp.). For annual flowerbed displays of the global Gardenesque gardens, favorites included marigolds (*Tagetes* spp.), petunias (*Petunia* spp.), violets (*Viola* spp.), and geraniums (*Pelargonium* spp.). Likewise, common grass cultivars of the European lawn included English ryegrass, Kentucky bluegrass, common bent, and red fescue (Ignatieva 2011).

Unlike the temperate zone, there is a lack of data on what kinds of decorative ornamental plants are being used in urban green areas in tropical countries; only recently are inventories are being collected for these green spaces. For instance, Abendroth at al. (2012) report that over 80 % of woody plants in parks of Bandung, Indonesia are non-native species. In the southern Indian city of Bangalore, Nagendra and Gopal (2011) report 77 % of urban park trees are non-native. A similar pattern has also been reported for Rio de Janeiro, Brazil (Santos et al. 2010). Common plants across the tropics include palms, South American bougainvilleas (*Bougainvillea* spp.), Chinese hibiscus (*Hibiscus rosa-sinensis*), South-East Asian orchids, African bird of paradise (*Strelitzia reginae*), South American frangipanis (*Plumeria* spp.), and Australian Casuarina (*Casuarina* spp.) (McCracken 1997; Soderstrom 2001). Regardless of climate, temperate or tropical, studies reveal a common pattern of using non-native over native species in landscape designs because of ornamental qualities rather than ecological function (Quigley 2011). Nevertheless, there is an ecological movement within the nursery business to grow more native species.

10.6.3 Trends Towards Landscape Design Supporting Biodiversity

Most European urban parks, gardens and other landscape architecture types are based on indigenous flora and alien ornamentals introduced since the sixteenth century. Of the ornamentals only a small percentage (approximately 11 %) became invasive and competed with native species. This pattern of using indigenous species in parks differed on other continents, especially in the Southern Hemisphere because of European colonization. In European colonies, non-native species, imported from the colonizing country, were used rather than indigenous flora when creating parks and gardens. Conducive climate, absence of natural control agents and (in many cases) broad species niches facilitated the spread of non-native species, which dramatically changed native landscapes and ecosystems. New Zealand, especially, exhibits dramatic examples of native ecosystems loss. Today, the number of naturalized, non-native plants is the same as the number of indigenous vascular plants (2,500). Over 20,000 non-native species have been introduced since colonization. The speed with which the New Zealand native biota has been suppressed is unprecedented (Meurk 2007). Even the use of the term "native biodiversity" is problematic because of the large number of non-natives occupying native ecosystems (Meurk and Swaffield 2007). The native flora is particularly decimated in urban environments.

A consequence of globalization of landscape design is the process of homogenization of cultures, environments, and biodiversity. Today's urban environments with similar urban planning structure; architectural buildings; public parks and gardens; plants; networks of shops, hotels, and restaurants; and standardized food form one of the most important parts of a homogenized global culture. Likewise, the use of unified products from commercial nurseries results in a homogenization of the urban environment and a suppression of local biodiversity in both temperate and tropical climatic zones (Ignatieva 2011).

Comprehending the role of urban biodiversity as a crucial element of the urban ecosystems and an important component of a region's ecological and cultural identity, landscape designers and planners are incorporating more native species into landscape and park designs. Likewise, ecologists are realizing that gardens (and not just large conservation areas) may play a critical role for native species refuge in the advent of climate change by facilitating migration and seed dispersal (Goddard et al. 2009; Rudd et al. 2002). Nonetheless, because of developmental history and colonization patterns, approaches to urban biodiversity design differ between Europe and the rest of the world at the beginning of the twenty-first century. The European approach can be summarized as following: reintroduce native biodiversity, design with natural processes, and plant as many spaces as possible to increase biodiversity (using even very small biotopes) within the urban environment. By comparison, because of its colonization history, the Southern Hemisphere approach can be summarized as following: redevelop designs based on local climatic and historical traditions with an emphasis of revegetation with indigenous plants; manage sites intensively (even vacant lots and derelict lands) to control non-native species and pests; and increase native biodiversity whenever possible (Müller and Werner 2010).

The incorporation of native biodiversity into new and existing parks and landscape designs is an important element of an integrated holistic approach to create sustainable urban infrastructure. For instance, green corridors along highways, railways, bikeways or riparian zones and park infrastructure fulfill multiple functions in addition to enhancing biodiversity. Connecting green areas not only creates recreational networks by linking different social elements, but also ecological networks by linking remnant patches of vegetation and native ecosystems (Florgård 2009; Swaffield et al. 2009). Table 10.1 shows a compilation of activity examples using approaches of urban design for biodiversity across the world.

Table 10.1 Global examples of landscape design to enhance native biodiversity

Country	Activity examples	Source
Argentina	Indigenous plantings and restoration	Burgueño et al. (2005); Bernata (2007)
	Public green areas and modern private gardens	Faggi and Madanes (2008); Faggi and Ignatieva (2009)
Australia	Indigenous species gardens	Urquhart (1999)
Brazil	Landscape ecological planning	Herzog (2008)
	Green infrastructure and sustainability	Frischenbruder and Pellegrino (2006)
	Indigenous plantings and restoration	Vaccarino (2000); Chacel (2001)
Germany	Urban biotope mapping	Sukopp and Weiler (1988)
	Go Spontaneous	Kuhn (2006)
New Zealand	Low Impact Urban Design and Development	Ignatieva et al. (2008)
	Plant signatures	
	Going native: indigenous biodiversity	Spellerberg and Given (2004)
South Africa	Native gardens	Cilliers et al. (2011)
Sweden	Conservation of remnant vegetation	Florgård (2007, 2009); Swaffield et al. (2009)
	Perennial beds vs. annual beds	Ignatieva (2011)
	Pictorial meadows	
United Kingdom	London Biodiversity Partnership	Beatley (2000)
	"Naturalistic" plant communities	Hitchmough (2004); Dunnett (2008)
	Pictorial meadows	
United States	Low Impact Development: Portland, Oregon, Chicago, Illinois	Eason et al. (2003); Weinstein and English (2008)
	Prairie Restoration	Nassauer (1995)
	Backyard Conservation; Going Native	USDA NRCC (1998)
	Xericscaping	Knopf et al. (2002)

Most new and innovative design concepts—such as developing a new landscape architecture style, Biodiversinesque—can be used as a powerful visual tools for reinforcing urban biodiversity and making urban biodiversity more visible and recognizable for the general public in everyday life (Ignatieva and Ahrné 2013). In fact, the most recent trend in landscape design is to include not only native plant species but also insects, invertebrates and birds to mimic native ecosystems (Barnett 2008).

10.7 Biological Hotspots and Urban Landscapes

Because of the confluence of habitats and geomorphology, urban settlements often occur in biological hotspots—sites with high biological diversity. A compiled database (Aronson et al. 2012, and hereafter referred to as the NCEAS database) provides an opportunity to look at patterns of native and non-native species in biodiversity hotspots (for further discussion on the global confluence of urbanization and biodiversity hotspots, see Chap. 3). Myers et al. (2000) identified 25 global biodiversity hotspots, defined as regions that had greater than 1,500 endemic species of vascular flora and where more than 70 % of habitat had been lost. There has been considerable debate in the conservation community as to the ecological and management-based justifications for designating hotspots, however the recognition that certain areas in the world support high levels of biodiversity and that many of these areas are under threat is accepted as valid (Jepson and Canney 2001). Cincotta et al. (2000) and Cincotta and Engleman (2000) reported that there are 146 cities in or directly adjacent to biodiversity hotspots, and 62 of these cities have over one million people. The large number of cities located in or adjacent to global hotspots and the potential for rapid urbanization in global hotspots and associated threats to biodiversity are both justifications for understanding patterns of biodiversity global hotspots. For a discussion of projected expansion of urban areas in relation to biodiversity hotspots, see Chap. 22.

Much of the literature on cities in biodiversity hotspots focuses on impacts of urbanization on protected areas, emphasizing the potential decline in species richness and extirpation of some species as urban areas expand (McDonald et al. 2008) (Chap. 3). However, only a small number of studies have looked at specific case studies of individual cities within hotspots. For instance, the NCEAS database on birds and plants for 25 cities occurring in biodiversity hotspots as defined by Conservation International identified that nine hotspot regions within the Mediterranean Basin contained the largest number of cities (Table 10.2) (Aronson et al. 2012).

Native species dominated the avifauna of the cities in biodiversity hotspots in the NCEAS database, with native species comprising greater than 85 % of all species in 13 of 15 cities where bird data were available. Only cities in New Zealand had fewer than 55 % native bird species. A similar pattern was observed among the 12 cities with plant data that occurred in biological hotspots. Greater than 75 % of species were native, with the exception of the East Afromontane city (Bujumbura, Burundi) and the New Zealand cities (Auckland and Hamilton) (Table 10.2) (Fig. 10.8). The NCEAS database contains only a few cities from Africa, Latin America and the Caribbean, and Southeast Asia and the Pacific Islands. The Garcillán et al. (2009) study of Ensenada, Mexico provides insights into patterns of plant diversity in Central America. They report that 61 % of the vascular plant species found in arroyo (dried river beds) and vacant lot habitats are non-native species. Ensenada has experienced rapid growth and expansion typical of cities in the global south and had a higher percentage of non-native species than reported cities in the same biogeographic realm in the United States. Garcillán et al. (2009) suggest that rapid urbanization from recent population growth has resulted in a loss of remnant habitats and an associated increase in the proportion of non-native plant species. Similar changes may occur in rapidly developing cities (see Chap. 3).

Table 10.2 Plant and bird species richness, percent native species, and percent IUCN Red-List species for cities occurring in global biodiversity hotspots

Hotspot and city	Area (km^2)	Bird species total	% Native	IUCN Red data species	Vascular plant species total	% Native	IUCN Red data species
California Floristic Province							
Fresno (Schelder 2010)	286	73	91.7	0			
Los Angeles (Müller and Mayr 2002; Schwartz et al. 2006)	1,214				865	77.7	1
San Diego (2010)	842				893	79.2	4
Mesoamerica							
Morelia (López-López 2011)	106	65	98.5	0			
Querétaro (Pineda-López 2011)	317	98	89.8	1			
Tropical Andes							
LaPaz (Villegas and Garitano-Zavala 2010)	187	64	98.4	0			
Mediterranean Basin							
Lisbon (Geraldes and Costa 2005)	85	94	98.9	0			
Valencia (Murgui 2005)	469	211	89.1	1			
Montpellier (Caula et al. 2008)	57	65	98.5	0			
Florence (Dinetti 2005)	102	82	97.6	0			
Rome (Cignini and Zapparoli 2005; Celesti-Grapow 1995)	803	98	89.8	1	1,259	82.2	1
Patras (Chronopoulos and Christodoulakis 1996, 2000)	333				765	87.2	0
Thessaloniki (Krigas and Kokkini 2004, 2005)	112				963	85.3	1
Alexandroupolis (Chronopoulos and Christodoulakis 2006)	1,219				439	91.4	0
Istanbul (Osma et al. 2010)	34				311	86.2	0
Jerusalem Bird Observatory (2008)	125	24	87.5	0			
East Afromontane							
Bujumbura (Bigirimana et al. 2011)	87				397	57.2	0

(continued)

Table 10.2 (continued)

Hotspot and city	Area (km^2)	Bird species total	% Native	IUCN Red data species	Vascular plant species total	% Native	IUCN Red data species
IndoBurma							
Hong Kong (Lock 2000; Thrower 1975)	1,104	111	94.6	1	1,883	87.1	12
Sundaland							
Singapore (Chong et al. 2000; Wang and Hails 2007)	710	368	94.3	19	1,787	87.4	41
Japan							
Sendai (Imai and Nakashizuka 2010)	784	31	96.8	0			
New Zealand							
Auckland (Duncan and Young 2000; Esler 2001)	664				1,350	21.6	0
Dunedin (van Heezik et al. 2008)							
Hamilton (Clarkson et al. 2007; Coleman and Clarkson 2010; Comes et al. 2000; Comes and Clarkson 2010; Innes et al. 2008)							

Richness data are grouped by biogeographical realm (From the NCEAS database Aronson et al. 2012)

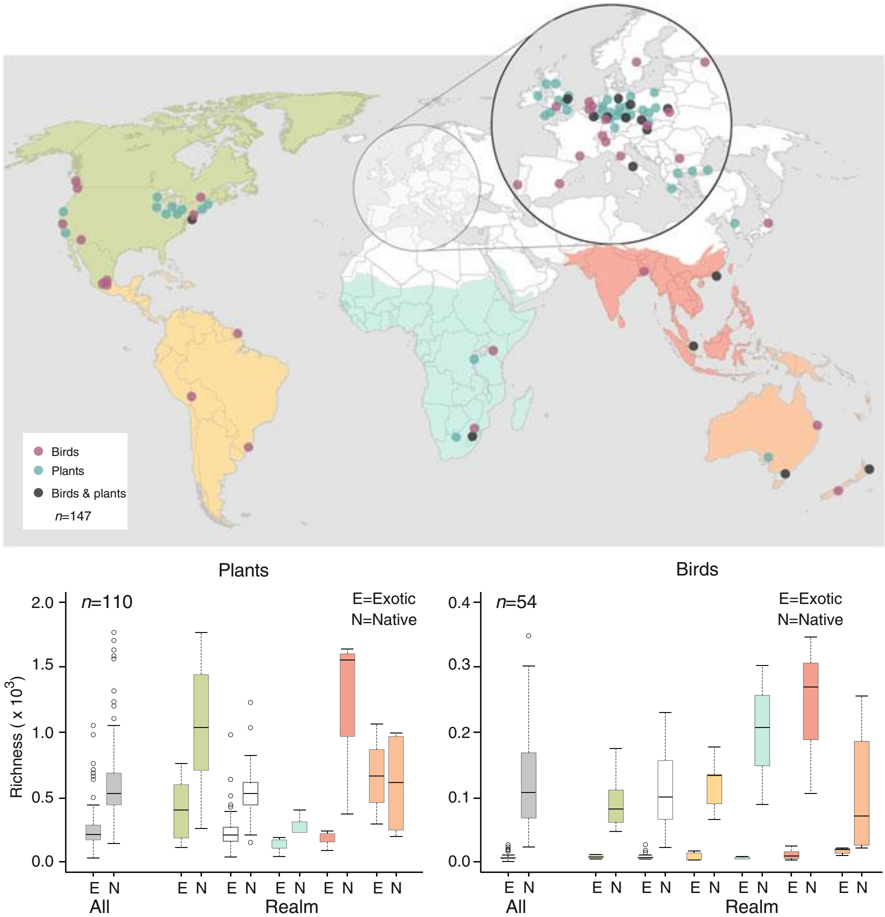

Fig. 10.8 Cities with data on plant species richness, bird species richness, and number of native and non-native plant and bird species. Richness data are grouped by biogeographical realm (From Aronson et al. 2012)

10.8 Conclusions

Species patterns and assemblages presented here reveal that social and ecological systems of the urban landscape are interconnected and form the observed patterns of biodiversity. Changes in the social context in urban landscapes often result in changes in ecological structure and function, and ultimately, urban biodiversity. Although generalizations about the effect of urbanization on biodiversity are often made, actual patterns can vary by region, biomes, and city history. Similarly, a species occurrence may vary among cities within a biome because of habitat availability, habitat quality, species availability, species adaptability, and site history. Nonetheless, urbanization does cause a loss of native biodiversity. This loss of biodiversity increases human vulnerability to natural calamities and reduces our resilience to those events. Likewise, the benefits of this biodiversity have only recently been linked to human health and well-being (see Chap. 11).

Even though we know that biodiversity is essential for human health and well-being, vital ecosystems are lost or destroyed and species are extirpated as cities continue to expand because of a burgeoning human population. These losses, however, occur unnecessarily. Current knowledge of ecosystem patterns and processes linked with landscape design, as detailed in this chapter, enables not only planners and managers but also individuals to build sustainable landscapes for humans as well as flora and fauna. Sustainable designs can be implemented at fine-scales through bottom-up planning as well as broad-scale through top-down planning (see Chap. 23 for discussion of urban governance for biodiversity and ecosystem services). Nonetheless, rapid human population growth as well as a basic lack of education resources available to a large portion of the world's population are major barriers to sustainability and implementation of these designs and practices. If the link between humans and nature is continuously re-established through actions across scales, the urban matrix can be sustained as a livable landscape for all species.

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Chapter 11
Urban Ecosystem Services

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Abstract We explore the potential of urban ecosystem services for improving resilience and quality of life in cities. First, we classify and categorize important ecosystem services and disservices in urban areas. Second, we describe a range of valuation approaches (cultural values, health benefits, economic costs, and resilience) for capturing the importance of urban ecosystem service multiple values. Finally, we analyze how ecosystem service assessment may inform urban planning and governance and provide practical examples from cities in Africa, Europe, and America. From our review, we find that many urban ecosystem services have already been

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identified, characterized and valued, and have been found to be of great value and importance for human well-being and urban resilience. We conclude that the use of the concept of urban ecosystem services can play a critical role in reconnecting cities to the biosphere, and reducing the ecological footprint and ecological debt of cities while enhancing resilience, health, and quality of life of their inhabitants.

11.1 Reconnecting Cities to the Biosphere

Cities are interconnected globally through political, economic, and technical systems, and also through the Earth's biophysical life-support systems (Jansson 2013). Cities also have disproportionate environmental impacts at the local, regional, and global scales well beyond their borders (Grimm et al. 2000, 2008; Seto et al. 2012), yet they provide critical leadership in the global sustainability agenda (Folke et al. 2011). Although urbanized areas cover only a small portion of the surface of the planet, they account for a vast share of anthropogenic impacts on the biosphere. Still, the impacts of urbanization on biodiversity and ecosystems as well as the potential benefits from ecosystem restoration in urban areas remain poorly understood (see e.g., McDonald and Marcotullio 2011). For further discussion on urban restoration ecology, also see Chap. 31.

11.1.1 Ecology of vs. Ecology in Cities

Cities appropriate vast areas of functioning ecosystems for their consumption and waste assimilation (see Chaps. 2 and 26). Most of the ecosystem services consumed in cities are generated by ecosystems located outside of the cities themselves, often half a world away (Rees 1992; Folke et al. 1996; Rees and Wackernagel 1996;

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Deutsch and Folke 2005, see Chap. 2). Folke et al. (1997) estimated that the 29 largest cities in the Baltic Sea Drainage Basin, taking into account only the most basic ecosystem services such as food production and assimilation of nitrogen and carbon, appropriate ecosystem areas equivalent to the size of the entire drainage basin, several hundred times the area of the cities themselves (Chap. 26). Thus, our analysis needs to go beyond what is sometimes referred to as "the ecology *in* cities" (Niemelä et al. 2011), which often focuses on single scales and on designing energy-efficient buildings, sustainable logistics, and providing inhabitants with functioning green urban environments, to put more focus on "the ecology *of* cities" characterized by interdisciplinary and multiscale studies with a social-ecological systems approach (Grimm et al. 2000; Pickett et al. 2001, see also Chap. 3). This framework acknowledges the total dependence of cities on the surrounding landscape and the links between urban and rural, viewing the city as an ecosystem itself (Grimm et al. 2008). We need to be concerned with the generation potential, not only to uphold and safeguard the well-being of city inhabitants, but also to effectively manage the potential of cities as arenas for learning (this aspect is discussed in detail in Chap. 30), development, and transformation.

11.1.2 Urban Ecosystems and Ecological Infrastructure

Definitions of urban areas and their boundaries vary between countries and regions (for a discussion on "What is urban?" see Chap. 1). The focus of this chapter is on the services and benefits provided by urban ecosystems, defined here as those areas where the built infrastructure covers a large proportion of the land surface, or as those in which people live at high densities (Pickett et al. 2001). In the context of urban planning, urban ecosystems are often portrayed as embedding both the built infrastructure and the ecological infrastructure. The concept of ecological infrastructure captures the role that water and vegetation in or near the built environment play in delivering ecosystem services at different spatial scales (building, street, neighborhood, and region). It includes all 'green and blue spaces' that may be found in urban and peri-urban areas, including parks, cemeteries, gardens and yards, urban allotments, urban forests, single trees, green roofs, wetlands, streams, rivers, lakes, and ponds (EEA 2011). Defining clear boundaries for urban ecosystems often proves difficult because many of the relevant fluxes and interactions necessary to understand the functioning of urban ecosystems extend far beyond the urban boundaries defined by political or biophysical reasons. Thus, the relevant scope of urban ecosystem analysis reaches beyond the city area itself; it comprises not only the ecological infrastructure within cities, but also the hinterlands that are directly affected by the energy and material flows from the urban core and suburban lands (Pickett et al. 2001, p. 129), including city catchments, and peri-urban forests and cultivated fields (La Rosa and Privitera 2013). Whilst virtually any ecosystem is relevant to meet urban ecosystem service demands, the focus here is on services provided within urban areas.

11.2 Classifying Urban Ecosystem Services

In recent years a mounting body of literature advanced our understanding of urban ecosystem services in their biophysical, economic, and socio-cultural dimensions. Furthermore, urban ecosystem services were addressed by major initiatives like the Millennium Ecosystem Assessment (Chapter 27 in MA 2005) and The Economics of Ecosystems and Biodiversity (TEEB 2011), and also have received increasing attention as part of the policy debate on ecological infrastructure. Yet, despite the fact that more than half of the world's population today lives in cities, the attention given to urban ecosystems in the ecosystem services literature has yet been relatively modest as compared to other ecosystems like wetlands or forests. This section aims at classifying and describing ecosystem services provided in urban areas and how these may contribute to increase quality of life in cities.

Building on previous categorizations of ecosystem services (Daily 1997; de Groot et al. 2002), the Millennium Ecosystem Assessment (MA 2005) and The Economics of Ecosystem Services and Biodiversity (TEEB 2010) grouped ecosystem services in four major categories: provisioning, regulating, habitat, and cultural and amenity services (TEEB 2010) (Fig. 11.1). Provisioning services include all the material products obtained from ecosystems, including genetic resources, food and fiber, and fresh water. Regulating services include all the benefits obtained from the regulation by ecosystem processes, including the regulation of climate, water, and some human diseases. Cultural services are the non-material benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experience as well as their role in supporting knowledge systems, social relations, and aesthetic values. Finally, supporting or habitat services are those that are necessary for the production of all other ecosystem services. Examples include biomass production, nutrient cycling, water cycling, provisioning of habitat for species, and maintenance of genetic pools and evolutionary processes.

Because different habitats provide different types of ecosystem services, general classifications need to be adapted to specific types of ecosystems. Urban ecosystems are especially important in providing services with direct impact on human health and security such as air purification, noise reduction, urban cooling, and runoff mitigation. Yet, which ecosystem services in a given scale are most relevant varies greatly depending on the environmental and socio-economic characteristics of each geographic location. Below we provide a classification and description of important ecosystem services provided in urban areas using the Millennium Ecosystem Assessment and the TEEB initiative as major classification frameworks, and drawing on previous research on the topic (e.g., Bolund and Hunhammar 1999; Gómez-Baggethun and Barton 2013).

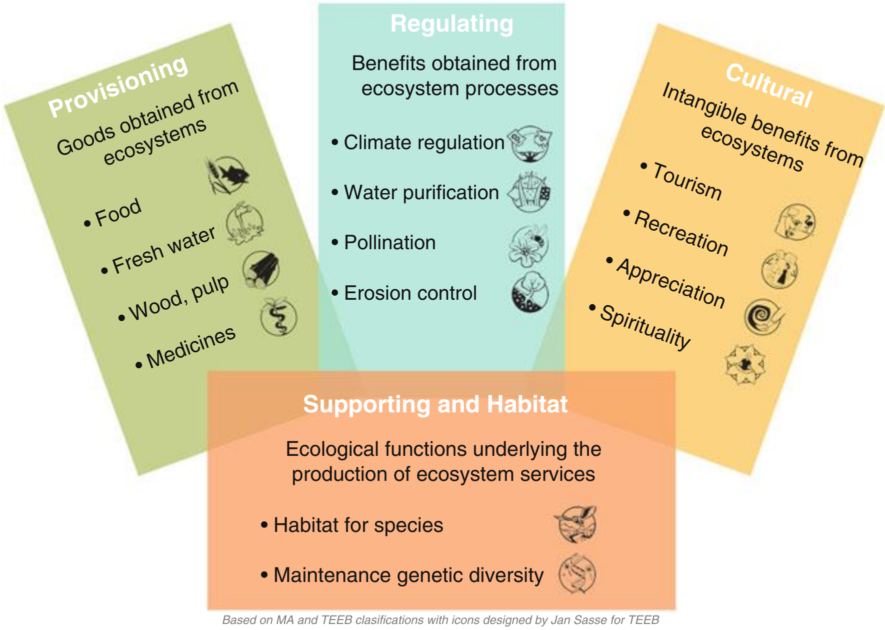

Fig. 11.1 Classification of ecosystem services based on the Millennium Ecosystem Assessment (MA 2005) and the Economics of Ecosystems and Biodiversity initiative (TEEB 2012) (Produced by Gómez-Baggethun 2013 with icons designed by Jan Sasse for TEEB. Icons reproduced from Jan Sasse for TEEB. Published with kind permission of © Jan Sasse and TEEB 2013. All Rights Reserved)

11.2.1 Provisioning Services

11.2.1.1 Food Supply

Urban food production takes place in peri-urban farm fields, on rooftops, in backyards, and in community gardens (Andersson et al. 2007; Barthel et al. 2010). In most geographical contexts, cities only produce a small share of the food they consume, depending largely on other areas to meet their demands (Folke et al. 1997; Ernstson et al. 2010). In some geographical areas and in particular periods, however, food production from urban agriculture can play an important role for food security, especially during economic and political crises (Smit and Nasr 1992; Moskow 1999; Page 2002; Buchmann 2009; Barthel et al. 2011; Barthel and Isendahl 2013). Altieri et al. (1999) estimated that in 1996 food production in urban gardens of Havana included 8,500 t of agricultural products, 7.5 million eggs and

3,650 t of meat. Moustier (2007) provides an extensive summary of the importance of urban agriculture in 14 African and Asian cities. Among the results they found that 90 % of all vegetables consumed in Dar es Salaam (Jacobi et al. 2000) and 60 % of vegetables consumed in Dakar (Mbaye and Moustier 2000) originate from urban agriculture. With regards to staple foods such as rice, plantain banana, and maize, the situation is highly variable among cities. In Asia, the share of rice supplied by the city to urban residents ranges from 7 % (in Phnom Penh) to 100 % (in Vientiane, where pressure on land is low); Hanoi is an intermediary case with 58 % (Anh 2004; Ali et al. 2005). For a detailed examination of the connection between urbanization and food systems, see Chap. 26.

11.2.1.2 Water Supply

The growth of cities throughout the world presents new challenges for securing water to meet societal needs (Fitzhugh and Richter 2004). Ecosystems provide cities with fresh water for drinking and other human uses and by securing storage and controlled release of water flows. Vegetation cover and forests in the city catchment influences the quantity of available water (for a global overview of cities' relationships with freshwater ecosystem services, see Chap. 3). One of the most widely cited examples of the importance of functioning ecosystems for city water supply is the New York City Watershed. This watershed is one of New York State's most important natural resources, providing approximately 1.3 billion gallons of clean drinking water to roughly nine million people every day. This is the largest unfiltered water supply in the United States (Chichilnisky and Heal 1998). Another example is the Omerli Watershed outside Istanbul, Turkey. The Omerli Watershed is the most important among the seven Mediterranean watersheds that provides drinking water to Istanbul, a megacity with over ten million people. The watershed, however, is threatened by urban development in and around its drinking water sources, and it faces acute, unplanned pressures of urbanization with potentially serious impacts on water quality and biodiversity (Wagner et al. 2007). For a detailed assessment on Istanbul, including further discussion on the Omerli Watershed, see Chap. 16.

11.2.2 Regulating Services

11.2.2.1 Urban Temperature Regulation

Ecological infrastructure in cities regulates local temperatures and buffers the effects of urban heat islands (Moreno-Garcia 1994). For example, water areas buffer temperature extremes by absorbing heat in summertime and by releasing it in wintertime (Chaparro and Terradas 2009). Likewise, vegetation reduces temperature in the hottest months through shading and through absorbing heat from the air by evapotranspiration, particularly when humidity is low (Bolund and Hunhammar 1999; Hardin and Jensen 2007). Water from the plants absorbs heat as it evaporates, thus cooling the air in the process (Nowak and Crane 2000). Trees can also regulate local surface and air temperatures by reflecting solar radiation and shading surfaces, such as streets and sidewalks that would otherwise absorb heat. Decreasing the heat loading of the city is among the most important regulating ecosystem services trees provide to cities (McPhearson 2011).

11.2.2.2 Noise Reduction

Traffic, construction, and other human activities make noise a major pollution problem in cities, affecting health through stress. Urban soil and plants can attenuate noise pollution through absorption, deviation, reflection, and refraction of sound waves (Aylor 1972; Kragh 1981; Fang and Ling 2003). In row plantings of trees, sound waves are reflected and refracted, dispersing the sound energy through the branches and trees. It has also been shown that different plant species mitigate noise differently (see e.g., Ishii 1994; Pathak et al. 2007). Empirical research has found that vegetation factors important for noise reduction include density, width, height and length of the tree belts as well as leaf size and branching characteristics. For example, the wider the vegetation belt, the higher the density, and the more foliage and branches to reduce sound energy, the greater the noise reduction effect (Fang and Ling 2003). Noise reduction is also affected by factors beyond the characteristics of vegetation. For example, climate influences the velocity of sound propagation (Embleton 1963) and noise attenuation increases with distance between the source point and the receiver due to friction between atmospheric molecules when sound progresses (Herrington 1976).

11.2.2.3 Air Purification

Air pollution from transportation, industry, domestic heating, and solid urban waste incineration is a major problem for environmental quality and human health in the urban environment; it leads to increases in respiratory and cardiovascular diseases. Vegetation in urban systems can improve air quality by removing pollutants from the atmosphere, including ozone (O_3), sulfur dioxide (SO_2), nitrogen dioxide (NO_2), carbon monoxide (CO) and particulate matter less than 10 μm (PM10) (Nowak 1994a; Escobedo et al. 2008). While significant differences in performance have been found between plant species (e.g., between deciduous and evergreen species), urban trees have been shown to be especially important in intercepting air pollutants (Aylor et al. 2003). The distribution of different particle size fractions can differ both between and within species and also between leaf surfaces and in waxes (Dzierzanowski et al. 2011). Removal of pollution takes place as trees and shrubs filter out airborne particulates through their leaves (Nowak 1996). Performance of pollution removal also follows daily variation because during the night the plant stomata are closed and do not absorb pollutants, and monthly variation because of the changes in light hours and because of the shedding of the leaves by deciduous forest during the winter.

11.2.2.4 Moderation of Climate Extremes

Climate change is increasing the frequency and intensity of environmental extremes; this poses increasing adaptation challenges for cities, especially for those located in coastal areas (Meehl and Tebaldi 2004; Zahran et al. 2008). In Europe, heat waves have been the most prominent hazard with regards to human fatalities in the last decade. The European 2003 heat wave, for example, accounted for more than 70,000 excess deaths (EEA 2010). Ecological infrastructure formed by mangroves, deltas and coral reefs can act as natural barriers that buffer cities from extreme climate events and hazards, including storms, heat waves, floods, hurricanes, and tsunamis; this infrastructure can drastically reduce the damage caused to coastal cities (Farber 1987; Danielsen et al. 2005; Kerr and Baird 2007). Vegetation also stabilizes the ground and reduces the likelihood of landslides. Devastating effects caused by events like the Indian Ocean Tsunami in 2004 and Hurricane Katrina in 2005 have led a number of scientists to call for a new vision in risk management and vulnerability reduction in cities, based on wise combinations in the use of built infrastructure (e.g., levees) and ecological infrastructure (e.g., protective role of vegetation) (Danielsen et al. 2005; Depietri et al. 2012).

11.2.2.5 Runoff Mitigation

Increasing the impermeable surface area in cities leads to increased volumes of surface water runoff, and thus increases the vulnerability to water flooding. Vegetation reduces surface runoff following precipitation events by intercepting water through the leaves and stems (Villarreal and Bengtsson 2005). The underlying soil also reduces infiltration rates by acting as a sponge by storing water in the pore spaces until it percolates as through-flow and base-flow. Urban landscapes with 50–90 % impervious cover can lose 40–83 % of rainfall to surface runoff compared to 13 % in forested landscapes (Bonan 2002). Interception of rainfall by tree canopies slows down flooding effects and green areas reduce the pressure on urban drainage systems by percolating water (Bolund and Hunhammar 1999; Pataki et al. 2011). Street trees in New York, for instance, intercept 890 million gallons of stormwater annually (Peper et al. 2007). Other means of reducing urban stormwater runoff include linear features (bioswales), green roofs, and rain gardens (Clausen 2007; Shuster et al. 2008). For example, green roofs can retain 25–100 % of rainfall, depending on rooting depth, roof slope, and the amount of rainfall (Oberndorfer et al. 2007). Also, green roofs may delay the timing of peak runoff, thus lessening the stress on storm-sewer systems. Rain gardens and bioretention filters can also reduce surface runoff (Clausen 2007; Villarreal and Bengtsson 2005; Shuster et al. 2008).

11.2.2.6 Waste Treatment

Ecosystems filter out and decompose organic wastes from urban effluents by storing and recycling waste through dilution, assimilation and chemical re-composition (TEEB 2011). Wetlands and other aquatic systems, for example, filter wastes from human activities; this process reduces the level of nutrients and pollution in urban wastewater (Karathanasis et al. 2003). Likewise, plant communities in urban soils can play an important role in the decomposition of many labile and recalcitrant litter types (Vauramo and Setälä 2010). In urban streams, nutrient retention can be increased by adding coarse woody debris, constructing in-channel gravel beds, and increasing the width of vegetation buffer zones and tree cover (Booth 2005).

11.2.2.7 Pollination, Pest Regulation and Seed Dispersal

Pollination, pest regulation and seed dispersal are important processes in the functional diversity of urban ecosystems and can play a critical role in their long term durability (Andersson et al. 2007). However, pollinators, pest regulators and seed dispersers are threatened by habitat loss and fragmentation due to urban development and expansion. In this context, allotment gardens (called community gardens in North America, i.e. a plot of land made available for individual, non-commercial gardening), private gardens and other urban green spaces have been shown to be important source areas (Ahrné et al. 2009). Also, research in urban ecosystem services shows that a number of formal and informal management practices in allotment gardens, cemeteries and city parks promote functional groups of insects that enhance pollination and bird communities, which in turn enhance seed dispersal (Andersson et al. 2007). To manage these services sustainably over time, a deeper understanding of how they operate and depend on biodiversity is crucial (Nelson et al. 2009). Jansson and Polasky (2010) have developed a method for quantifying the impact of change in pollination potential in the regional urban landscape. Their results indicate that while the impact of urban development on the pollination service can be modest, the erosion of the resilience of the service, measured through change in response diversity, could be potentially high. For discussion on response diversity see Elmqvist et al. (2003).

11.2.2.8 Global Climate Regulation

Because urban areas exhibit multiple artificial surfaces and high levels of fossil fuel combustion, climate change impacts may be exacerbated in cities (Meehl and Tebaldi 2004). Emissions of greenhouse gases in cities include carbon dioxide (CO_2), methane (CH_4), nitrous oxide (NO_2), chlorofluorocarbons (CFCs), and tropospheric ozone (O_3). Urban trees act as a sinks of CO_2 by storing excess carbon as biomass during photosynthesis (Birdsey 1992; Jo and McPherson 1995; McPherson and Simpson 1999). Because the amount of CO_2 stored is proportional to the biomass of the trees, increasing the number of trees can potentially slow the accumulation of atmospheric carbon in urban areas. Thus an attractive option for climate change mitigation in cites is tree-planting programs. The amount of carbon stored and sequestered by urban vegetation has often been found to be quite substantial, for instance, 6,187 t/year in Barcelona (Chaparro and Terradass 2009) and 16,000 t/year in Philadelphia (Nowak et al. 2007b). Urban soils also act as carbon pools (Nowak and Crane 2000; Pouyat et al. 2006; Churkina et al. 2010). Yet, the amount of carbon a city can offset locally through ecological infrastructure is modest compared to overall city emissions (Pataki et al. 2011).

11.2.3 Cultural Services

11.2.3.1 Recreation

Because city environments may be stressful for inhabitants, the recreational aspects of urban ecosystems are among the highest valued ecosystem service in cities (Kaplan and Kaplan 1989; Bolund and Hunhamar 1999; Chiesura 2004; Konijnendijk et al. 2013). Parks, forests, lakes and rivers provide manifold possibilities for recreation, thereby enhancing human health and well-being (Konijnendijk et al. 2013). For example, a park experience may reduce stress, enhance contemplativeness, rejuvenate the city dweller, and provide a sense of peacefulness and tranquility (Kaplan 1983). The recreational value of parks depends on ecological characteristics such as biological and structural diversity, but also on built infrastructure such as availability of benches and sport facilities. The recreational opportunities of urban ecosystems also vary with social criteria, including accessibility, penetrability, safety, privacy and comfort, as well as with factors that may cause sensory disturbance (i.e., recreational value decreases if green areas are perceived to be ugly, trashy or too loud) (Rall and Haase 2011). Urban ecosystems like community gardens also offer multiple opportunities for decommodified leisure and nowadays represent important remnants of the shrinking urban commons.

11.2.3.2 Aesthetic Benefits

Urban ecosystems play an important role as providers of aesthetic and psychological benefits that enrich human life with meanings and emotions (Kaplan 1983). Aesthetic benefits from urban green spaces have been associated with reduced stress (Ulrich 1981) and with increased physical and mental health (e.g., Maas 2006; van den Berg et al. 2010a). Ulrich (1984) found that a view through a window looking out at greenspaces could accelerate recovery from surgeries, and van den Berg et al. (2010b) found that proximity of an individual's home to green spaces was correlated with fewer stress-related health problems and a higher general health perception. People often choose where to live in cities based in part on the characteristics of the natural landscapes (Tyrväinen and Miettinen 2000). Several studies have shown an increased value of properties (as measured by hedonic pricing) with greater proximity to green areas (Tyrväinen 1997; Cho et al. 2008; Troy and Grove 2008; Tyrväinen and Miettinen 2000; Jim and Chen 2006).

11.2.3.3 Cognitive Development

Exposure to nature and green space provide multiple opportunities for cognitive development which increases the potential for stewardship of the environment and for a stronger recognition of ecosystem services (Krasny and Tidball 2009; Tidball and Krasny 2010). As an example, urban forests and allotment gardens are often used for environmental education purposes (Groening 1995; Tyrväinen et al. 2005) and facilitate cognitive coupling to seasons and ecological dynamics in technological and urbanized landscapes. Likewise, urban allotments, community gardens, cemeteries and other green spaces have been found to retain important bodies of local ecological knowledge (Barthel et al. 2010), and embed the potential to compensate observed losses of ecological knowledge in wealthier communities (Pilgrim et al. 2008). The benefits of preserving local ecological knowledge have been highlighted in terms of increased resilience and adaptive capacities in urban systems (Buchmann 2009), and the potential to sustain and increase other ecosystem services (Colding et al. 2006; Barthel et al. 2010). For further discussion on how urban landscapes can serve as learning arenas for biodiversity and ecosystem services management, see Chap. 30.

11.2.3.4 Place Values and Social Cohesion

Place values refer to the affectively charged attachments to places (Feldmann 1990; Altman and Low 1992). Research conducted in Stockholm, for example, found sense of place to be a major driver for environmental stewardship, with interviewees showing strong emotional bonds to their plots and the surrounding garden areas (Andersson et al. 2007). Attachment to green spaces in cities can also give rise to other important societal benefits, such as social cohesion, promotion of shared interests, and neighborhood participation (Gotham and Brumley 2002). Examples include studies conducted in Chicago, Illinois, United States, and Cheffield, United Kingdom (Bennett 1997). Environmental authorities in the European Union have emphasized the role of urban green space in providing opportunities for interaction between individuals and groups that promote social cohesion and reduce criminality (European Environmental Agency 2011; Kázmierczak 2013). Likewise, urban ecosystems have been found to play a role in defining identity and sense of community (Chavis and Pretty 1999; Gotham and Brumley 2002). Research on sense of community in the urban environment indicates that an understanding of how communities are formed enable us to design housing that will be better maintained and will provide for better use of surrounding green areas (Newman 1981).

11.2.4 Habitat Services

11.2.4.1 Habitat for Biodiversity

Urban systems can play a significant role as refuge for many species of birds, amphibians, bees, and butterflies (Melles et al. 2003; Müller et al. 2010). Well-designed green roofs can provide habitat for species affected by urban land-use changes (Oberndorfer et al. 2007; Brenneisen 2003). In cold and rainy areas, golf courses in urban setting can have the potential to contribute to wetland fauna support (Colding and Folke 2009; Colding et al. 2009). Old hardwood deciduous trees in the National City Park of Stockholm, Sweden are seen as an important resource for the whole region for species with high dispersal capacity (Zetterberg 2011). Diversity of species may peak at intermediate levels of urbanization, at which many native and non-native species thrive, but it typically declines as urbanization intensifies (Blair 1996).

A synthesis of the above classification of urban ecosystem services is provided in Table 11.1

11.2.5 Ecosystem Disservices

Urban ecosystems not only produce ecosystem services, but also ecosystem disservices, defined as "functions of ecosystems that are perceived as negative for human well-being" (Lyytimäki and Sipilä 2009, p. 311). For example, some common city tree and bush species emit volatile organic compounds (VOCs) such as isoprene, monoterpenes, ethane, propene, butane, acetaldehyde, formaldehyde, acetic acid and formic acid, all of which can indirectly contribute to urban smog and ozone problems through CO and O_3 emissions (Geron et al. 1994; Chaparro and Terradas 2009). Urban biodiversity can also cause damages to physical infrastructures; microbial activity can result in decomposition of wood structures and bird excrements can cause corrosion of stone buildings and statues. The root systems of vegetation often cause substantial damages by breaking up pavements and some animals are often perceived as a nuisance as they dig nesting holes (de Stefano and Deblinger 2005; Lyytimäki and Sipila 2009).

Green-roof runoff may contain higher concentrations of nutrient pollutants, such as nitrogen and phosphorus, than are present in precipitation inputs (Oberndorfer et al. 2007). Further disservices from urban ecosystems may include health problems from wind-pollinated plants causing allergic reactions (D'Amato 2000), fear from dark green areas that are perceived as unsafe, especially by women at nighttime (Bixler and Floyd 1997; Koskela and Pain 2000; Jorgensen and Anthopoulou 2007), diseases transmitted by animals (e.g., migratory birds carrying avian influenza, dogs carrying rabies), and blockage of views by trees (Lyytimäki et al. 2008). Likewise, just as some plants and animals are perceived by people as services, as

Table 11.1 Classification of important ecosystem services in urban areas and underlying ecosystem functions and components

Ecosystem functions	Ecosystem service type	Examples	Key references
Energy conversion into edible plants through photosynthesis	Food supply	Vegetables produced by urban allotments and peri-urban areas	Altieri et al. (1999)
Percolation and regulation of runoff and river discharge	Runoff mitigation	Soil and vegetation percolate water during heavy and/or prolonged precipitation events	Villarreal and Bengtsson (2005)
Photosynthesis, shading, and evapotranspiration	Urban temperature regulation	Trees and other urban vegetation provide shade, create humidity and block wind	Bolund and Hunhammar (1999)
Absorption of sound waves by vegetation and water	Noise reduction	Absorption of sound waves by vegetation barriers, specially thick vegetation	Aylor (1972); Ishii (1994); Kragh (1981)
Dry deposition of gases and particulate matter	Air purification	Absorption of pollutants by urban vegetation in leaves, stems and roots	Escobedo and Nowak (2009); Jim and Chen (2009); Chaparro and Terradas (2009); Escobedo et al. (2011)
Physical barrier and absorption of kinetic energy	Moderation of environmental extremes	Storm, flood, and wave buffering by vegetation barriers; heat absorption during severe heat waves; intact wetland areas buffer river flooding	Danielsen et al. (2005); Costanza et al. (2006b)
Removal or breakdown of xenic nutrients	Waste treatment	Effluent filtering and nutrient fixation by urban wetlands	Vauramo and Setälä (2010)
Carbon sequestration and storage by fixation in photosynthesis	Global climate regulation	Carbon sequestration and storage by the biomass of urban shrubs and threes	Nowak (1994b); McPherson (1998)
Movement of floral gametes by biota	Pollination and seed dispersal	Urban ecosystem provides habitat for birds, insects, and pollinators	Hougner et al. (2006); Andersson et al. (2007)

(continued)

Table 11.1 (continued)

Ecosystem functions	Ecosystem service type	Examples	Key references
Ecosystems with recreational values	Recreation	Urban green areas provide opportunities for recreation, meditation, and relaxation	Chiesura (2004); Maas et al. (2006)
Human experience of ecosystems	Cognitive development	Allotment gardening as preservation of socio-ecological knowledge	Barthel et al. (2010); Groening (1995); Tyrväinen et al. (2005)
Ecosystems with aesthetic values	Aesthetic benefits	Urban parks in sight from houses	Tyrväinen (1997); Cho et al. (2008); Troy and Grove (2008)
Habitat provision	Habitat for biodiversity	Urban green spaces provide habitat for birds and other animals that people like watching	Blair (1996); Blair and Launer (1997)

Modified from Gómez-Baggethun and Barton (2013) based on a literature review

Note: The suitability of indicators for biophysical measurement is scale dependent. Most indicators and proxies provided here correspond to assessment at the plot level

Table 11.2 Ecosystem disservices in cities (Modified from Gómez-Baggethun and Barton 2013)

Ecosystem functions	Disservice	Examples	Key references
Photosynthesis	Air quality problems	City tree and bush species emit volatile organic compounds (VOCs)	Chaparro and Terradas (2009); Geron et al. (1994)
Tree growth through biomass fixation	View blockage	Blockage of views by trees standing close to buildings	Lyytimäki et al. (2008)
Movement of floral gametes	Allergies	wind-pollinated plants causing allergic reactions	D'Amato (2000)
Aging of vegetation	Accidents	Break up of branches falling in roads and trees	Lyytimäki et al. (2008)
Dense vegetation development	Fear and stress	Dark green areas perceived as unsafe in night-time	Bixler and Floyd (1997)
Biomass fixation in roots; decomposition	Damages to infrastructure	Breaking up of pavements by roots; microbial activity	Lyytimäki and Sipila (2009)
Habitat provision for animal species	Habitat competition with humans	Animals/insects perceived as scary, unpleasant, disgusting	Bixler and Floyd (1997)

Modified from Gómez-Baggethun and Barton (2013)

discussed above, animals such as rats, wasps and mosquitoes, and plants such as stinging nettles, are perceived by many as disservices. A summary of disservices from urban ecosystems is provided in Table 11.2.

11.3 Valuing Urban Ecosystem Services

11.3.1 Ecosystem Services Values

Valuation of ecosystem services involves dealing with multiple, and often conflicting value dimensions (Martinez Alier et al. 1998; Chan et al. 2012; Martín-López et al. 2013). In this section, we broaden the traditional focus of the ecosystem services literature on biophysical measurement and monetary values to explore a range of value domains, including biophysical, monetary, socio-cultural, health, and insurance values, and discuss concepts and methods through which they may be measured and captured.

11.3.1.1 Biophysical Values

Quantifying ecosystem service performance involves the use of biophysical measures and indicators. The difficulty of measuring ecosystem services in biophysical terms increases as the focus shifts from provisioning, to regulating to habitat, to cultural services. Thus, while most provisioning and some regulating ecosystem services can be quantified through direct measures, such as tons of food per hectare per year, or tons of carbon sequestered per hectare per year, in most cases measurement in biophysical terms involves the use of proxies and indicators.

Biophysical measures of ecosystem services are often presented as a prerequisite for sound economic valuations. While this may hold true, biophysical measures themselves often provide powerful information to guide urban planning. Thus, various biophysical indexes of urban green areas have been used for guiding planning procedures in cities (revised in Farrugia et al. 2013). An early attempt was made in Berlin, Germany with the Biotope Area Factor (BAF), which scored land surface types in development sites according to their ecological potential and formulated target BAFs for specific urban functions which developers were obliged to meet in order to obtain approval for any development proposal. Malmö City Council in Sweden adopted a similar system to incorporate green and blue infrastructure in land use planning, while aiming to reduce the extent of impervious surfaces in any development plans (Kruuse 2011). Another attempt to quantify the value of green areas was made in Kent Thameside in the United Kingdom (Defra 2008), which scored ecosystem services such as biodiversity, recreation and flood regulation using surrogates. The Southampton City Council in the United Kingdom developed a version of the Green Space Factor (GSF) tool to evaluate the contribution of green areas to water regulation flood control (Finlay 2010).

A summary with examples of indicators and proxies to measure ecosystem services and disservices is provided in Table 11.3.

11.3.1.2 Economic Values

Conventional economic valuations are restricted to priced goods and services, which represent only a limited subset of ecosystem services (i.e., those which are exchanged in markets). As price formation is conditioned to the existence of supply and demand relations, every change in human well-being lacking a market is invisible to conventional economic accounts. The economic literature refers to these effects as environmental externalities, which can be either negative (e.g., pollution) or positive (e.g., ecosystem services). The public good nature of most ecosystem services implies that their economic value is often not adequately reflected in management decisions that are mainly based on economic information (e.g., cost–benefit analysis). Consequently, it is argued, ecosystem services with no explicit economic value tend to be depleted.

Table 11.3 Examples of indicators and proxies for measuring urban ecosystem services and disservices in biophysical terms

Ecosystem services	Examples of biophysical indicators and proxies
Provisioning services	
Food supply	Production of food (t/year)
Freshwater supply	Water flow (m^3/year)
Regulating services	
Water flow regulation and runoff mitigation	Soil infiltration capacity; % sealed relative to permeable surface (ha)
Urban temperature regulation	Leaf Area Index
Noise reduction	Leaf area (m^2) and distance to roads (m); noise reduction [dB(A)]/vegetation unit (m)
Air purification	O_3, SO_2, NO_2, CO, and PM_{10} μm pollutant flux (g/cm^2/s) multiplied by tree cover (m^2)
Moderation of environmental extremes	Cover density of vegetation barriers separating built areas from the sea
Waste treatment	P, K, Mg and Ca in mg/kg compared to given soil and water quality standards
Climate regulation	CO_2 sequestration by trees (carbon multiplied by 3.67 to convert to CO_2)
Pollination and seed dispersal	Species diversity and abundance of birds and bumble bees
Cultural services	
Recreation and health	Area of green public spaces (ha)/inhabitant (or every 1,000 inhabitants); self-reported general health
Cognitive development and knowledge preservation	Participation, reification, and external sources of social-ecological memory
Habitat for biodiversity	
Habitat for biodiversity	Abundance of birds, butterflies and other animals valued for their aesthetic attributes
Ecosystem disservices	*Examples of indicators proxies*
Air quality problems	Emission of VOCs (t/year)/vegetation unit
View blockage	Tall trees close to buildings
Allergies	Allergenicity (e.g., OPALS ranking)
Accidents	Number of aged trees
Fear and stress	Area of non-illuminated parks
Damages on infrastructure	Affected pavement (m^2) wood (m^3)
Habitat competition with humans	Abundance of insects, rats, etc.

Modified from Gómez-Baggethun and Barton (2013), based on various sources

Because biodiversity loss generally involves long-term economic costs that are not adequately reflected in conventional economic accounts (Boyer and Polasky 2004; Tyrväinen et al. 2005; TEEB 2010; EEA 2011; Escobedo et al. 2011; Elmqvist et al. forthcoming) economic valuation of ecosystem services attempts to make visible the 'hidden' economic costs from the conversion of ecological infrastructure to built infrastructure (or from natural capital to human-made capital). These may include sanitary costs related to health damages from air pollution (Escobedo et al. 2008, 2011; Escobedo and Nowak 2009) and costs from increased property damages with loss of natural barriers to climate extremes (Costanza et al. 2006a).

Over the last few decades, a range of methods have been developed to calculate economic costs resulting from loss of ecological infrastructure. Avoided cost methods, for example, show that loss of urban vegetation can lead to increased energy costs in cooling during the summer season (McPherson et al. 1997; Chaparro and Terradas 2009). Likewise, decline of water regulation services from land-use change and loss of vegetation in the city catchments increase the dependence on water purification technologies, which are generally very costly (Daily and Ellison 2003). Economic costs may also derive the loss of ecosystem services such as air purification (McPherson et al. 1997; Nowak and Crane 2002), noise reduction by vegetation walls (Bolund and Hunhammar 1999), carbon sequestration by urban vegetation (McPherson et al. 1999; Jim and Chen 2009), buffering of climate extremes by natural barriers (Costanza et al. 2006a), and regulation of water flows (Xiao et al. 1998). These costs are not merely hypothetical. In most cases they are real economic costs derived from the partial substitution of ecological infrastructure and ecosystem services by built infrastructure and different economic services. Table 11.4 shows examples of quantitative measures of economic values directly or indirectly attached to ecosystems services in the urban context.

When pollutants are not specified, calculations include NO_2, SO_2, PM_{10}, O_3 and CO). Results from Jim and Chen (2009) concerted from RMB to $US after Elmqvist et al. forthcoming. Not all figures were normalized to net present values and therefore they should be taken as illustration only.

Using combinations of valuation methods is often necessary to address multiple ecosystem services (Boyer and Polasky 2004; Costanza et al. 2006b; Escobedo et al. 2011). The choice of valuation methods is determined by factors including the scale and resolution of the policy to be evaluated, the constituencies that can be contacted to obtain data, and supporting data constraints, all subject to a study budget (Table 11.5).

Avoided expenditure or replacement cost methods are often used to address values of regulating services such as air pollution mitigation and climate regulation (Sander et al. 2010). Meta-analyses on economic valuations of ecosystem services show that hedonic pricing (HP) and stated preference (SP) methods (and contingent valuation in particular), have been the methods most frequently used in urban contexts (Boyer and Polasky 2004; Tyrväinen et al. 2005; Costanza et al. 2006b; Kroll and Cray 2010; Sander et al. 2010; Brander and Koetse 2011). Economic valuation using hedonic pricing has often been used to capture recreational and amenity benefits (Tyrväinen and Miettinen 2000), views and aesthetic benefits (Anderson and Cordell 1985; Sander et al. 2010), noise reduction (Kim et al. 2007), air quality (Smith and Huang 1995; Bible et al. 2002; Chattopadhyay 1999), and water quality (Leggett and Bockstael 2000). A review by Kroll and Cray (2010) shows that hedonic pricing methods have been used mainly to value property features at neighborhood scales, especially in relation to open space, vegetation, and wetlands (Table 11.6).

Table 11.7 suggests potential valuation methods that can inform urban planning issues at different scales.

Table 11.4 Examples of economic valuations of five urban ecosystem services. Examples from empirical studies conducted in Europe, USA, and China

Ecosystem service	City	Ecological infrastructure	Biophysical accounts	Economic valuation	Reference
Air purification	Barcelona, Spain	Urban forest	305.6 t/y	€1,115,908	Chaparro and Terradas (2009)
	Chicago, USA	Urban trees	5,500 t/y	US$ 9 million	McPherson et al. (1997)
	Washington, USA	Urban trees	540 t/y 0.12 t/ha/y	–	Nowak and Crane (2000)
	Modesto, USA	Urban forest	154 t/y 3.7 lb/tree	US$ 1.48 million US$ 16/tree	McPherson et al. (1999)
	Sacramento, USA	Urban forest	189 t/y	US$ 28.7 million US$ 1,500/ha	Scott et al. (1998)
	Lanzhou, China	Urban plants	28,890 t pm/y 0.17 t pm/ha/y 1.8 million t SO_2/y 10.9 t SO_2/ha/y	US$ 102 US$ 6.3/ha –	Jim and Chen 2009
	Beijing, China	Urban forest	2,192 t SO_2/y 1,518 t pm/y 2,192 t SO_2/y (132 t SO_2/ha/y)	US$ 4.7 million US$ 283/ha	Jim and Chen (2009) Elmqvist et al. (Forthcoming)
Microclimate regulation	Chicago	City trees	Saved heating 2.1 GJ/tree Saved cooling 0.48 GJ/tree	US$ 10/tree US$ 15/tree	McPherson et al. (1997) McPherson (1992)
	Modesto, USA	Street and park trees	Saved 110,133 Mbtu/y	US$ 870,000 122kWh/tree US$ 10/tree)	McPherson et al. (1999)
	Sacramento, USA	Urban vegetation	Saved 9.8 MW/ha/y	US$ 1,774/ha/y	Simpson (1998)
	Beijing, China	Urban forest	1.4kWH/ha/day	US$ 12.3 million US$ 1,352/ha/y	Jim and Chen (2009)

(continued)

Table 11.4 (continued)

Ecosystem service	City	Ecological infrastructure	Biophysical accounts	Economic valuation	Reference
Carbon sequestration	Barcelona, Spain	Urban forest	113,437 t (gross) 5,422 t (net)		Chaparro and Terradas (2009)
	Modesto, USA	Urban forest	13,900 t or 336 lb/tree	US$ 460,000 or US$ 5/tree	McPherson et al. (1999)
	Washington DC, USA	Urban forest	16,200 t 3,500 t/h/y	US$ 299,000/y US$ 653/ha/y	Elmqvist et al. (Forthcoming)
	Philadelphia, USA	Urban forest	530,000 t (gross) 96 t/ha 16,100 t (net) 2.9 t/ha/y	US$ 9.8 million (gross) US$ 297,000 (net)	Nowak et al. (2007b)
	Beijing, China	Urban forest	4, 200,000 t 256 t/ha/y	US$ 20,827/ha/y	Jim and Chen (2009)
Regulation of water flows	Modesto, USA	Urban forest	Reduced runoff 292,000 m^3 or 845 gal/tree	US$ 616,000 or US$ 7/tree	McPherson et al. (1999)
	Sacramento	Urban trees	Annual rainfall reduced by 10 %	US$ 572/ha	Xiao et al. (1998)
Aesthetic information	Modesto, USA	Urban forest	88,235 trees	US$ 1.5 million US$ 17/tree)	McPherson et al. (1999)
	Guangzou, China	Urban green space	7,360 ha	US$ 17,822/ha/y	Jim and Chen (2009)

Modified from Gómez-Baggethun and Barton (2013)

Legend: *PM* particulate matter, *t* ton, *y* year, *lb* pound, *ha* hectare, *GJ* gigajoule, *Mbtu* million British Thermal Units, *MW* megawatt, m^3 meters cubed, *gal* gallon, *kWh* kilowatt hours

Table 11.5 Economic valuation of ecosystem services in urban planning

Scale	Urban planning issue	Role of economic valuation	Methodological challenges
Region	Prioritizing urban growth alternatives between different areas	Valuing benefits and costs of (i) urban revitalization (ii) urban infill (iii) urban extension (iv) suburban retrofit (v) suburban extension (vi) new neighbor-hoods with (vii) existing infrastructure (ix) new infrastructure (x) in environmentally sensitive areas	Comprehensive benefit-cost analysis at multiple scales and resolutions at multiple locations is expensive
	Fair and rational location of undesirable land uses (LULUs)	Value of the impacts and disservices of e.g., power plants and landfills and foregone ecosystem service values of ecological infrastructure	Using benefit-cost analysis to allocate infrastructure with local costs versus regional benefits may not achieve fair outcomes
	Preservation of productive peri-urban farm belt	Willingness to pay for preservation of open space and 'short distance' food	Large import substitution possibilities for locally produced food
	Water availability to support urban growth	Valuation to support full cost pricing of water supply. Incentive effects of removing water subsidies	Can require inter-regional geographical scope of valuation
	Using transferable development rights (TDR) to concentrate growth and achieve zoning	Determine farmer opportunity costs and benefits of foregoing urban development as a basis for predicting the size of a TDR market	
Neigh-borhood	Preserving views, open spaces, and parks in neighborhoods	Willingness to pay of households for quality and proximity of recreational spaces	Accounting for substitute sites and recreational activities
	Conserving soil drainage conditions and wetlands	Valuation of replacement costs of man-made drainage and storage infrastructure	Hydrological and hydraulic modeling required
	Conserving water	Costs of household water harvesting, recycling and xeriscapes	Cost-benefit evaluation requires comparison with full costs of water supply
	Natural corridors	Quantify opportunity costs of preserving corridors	Difficulty in specifying habitat connectivity requirements of corridors
	Local farm produce	Willingness to pay for local, fresh produce	Large import substitution possibilities for locally produced food
	Edible gardens	Recreational value of home gardens	

(continued)

Table 11.5 (continued)

Scale	Urban planning issue	Role of economic valuation	Methodological challenges
Street-scape	Street trees	Value pedestrian safety through slowing traffic; disamenities of heat islands; absorption of stormwater, and airborne pollutants	Associating ecosystem service values at neighborhood and street level to individual trees
	Green pavements for stormwater management	Willingness to pay of households for green streetscape; additional costs of larger dimension storm-water	
Building	Green rooftops Yard trees Lawns vs. xeriscapes	Additional costs of traditional stormwater management; mitigation of heat island	Associating ecosystem service values at neighborhood and street level to individual roofs, trees and lawns

Produced by Barton et al. (2012), based on a listing by Duany et al. (2010)

Table 11.6 Overview of hedonic pricing studies in cities

Scale	Property feature	# of studies
National/regional	Policies affecting property rights	5
Regional/neighborhood	Open space	28
	Water & wetlands	24
Neighborhood/streetscape	Open space vegetation & trees	20
Streetscape	Pavement type	7
Streetscape/property	Climate & temperature	5
Building	Energy efficiency	7

Produced by Barton et al. (2012), adapted from Kroll and Cray (2010)

11.3.1.3 Social and Cultural Values

People bring various material, moral, spiritual, aesthetic, and other values to bear on the urban environment; their values can affect their attitudes and actions toward ecosystems and the services they provide. These include emotional, affective and symbolic views attached to urban nature that in most cases cannot be adequately captured by commodity metaphors and monetary metrics (Norton and Hannon 1997; Martinez Alier et al. 1998; Gómez-Baggethun and Ruiz-Pérez 2011; Daniel et al. 2012). Here, we refer to these values broadly as social and cultural values. The ecosystem services literature has defined cultural values as "aesthetic, artistic, educational, spiritual and/or scientific values of ecosystems" (Costanza et al. 1997, p. 254) or as "non-material benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experience" (Millennium Ecosystem Assessment 2005, p. 894).

Social and cultural values are included in all prominent ecosystem service typologies (Daily et al. 1997; de Groot et al. 2002; Millennium Ecosystem Assessment 2005). Yet, compared with economic and biophysical values, social, cultural, and other non-material values of ecosystems and biodiversity have generally been neglected in much of the ecosystem services literature. Moreover, social and cultural values may be difficult to measure, often necessitating the use of more holistic approaches and methods that may include qualitative measures, constructed scales, and narration (Patton 2001; Chan et al. 2012). In some cases, tools have been developed to measure these values using constructed scales, as in the case of sense of place (Williams and Roggenbuck 1989; Shamai 1991) and local ecological knowledge (Gómez-Baggethun et al. 2010a). In other cases translating cultural values into quantitative metrics may be too difficult or produce results that are nonsensical or meaningless.

Recent research has made substantial progress in the quest to better integrate social perspectives and valuation techniques into the ecosystem services framework, and to enable a fuller representation of socio-cultural values in research and practice (e.g., Chan et al. 2012). Articulation of social and cultural values in decision-making processes may require, in most cases, some sort of deliberative

Table 11.7 Potential valuation methods for urban ecosystem service valuation

Valuation method	Types of value, ecosystem services	Scale	Constituencies	Constraints
Hedonic pricing (Revealed Preferences)	Use values (option value)	Building, streetscape and neighborhood characteristics	Home and property owners	Observable quality variables. Spatially explicit
	Cultural services (amenities)			Autocorrelation and latent variables
Travel cost (Revealed Preferences)	Use values	Regional park/recreational destinations	Recreational visitors	No/low travel costs to neighborhood open spaces. Spatially explicit.
	Cultural services (amenities)			Locational self-selection.
Contingent valuation (Stated Preferences)	Use and non-use values	All infrastructure scales, easier for location specific policy scenario	Households or individuals, often as voters	Hypothetical, question framing issues, information burden
	All ecosystem services, but often amenities			Usually not spatially explicit
	Service bundles			
Choice experiments (Stated Preferences)	Use and non-use values	All infrastructure scales, but easier for location specific policy choice alternatives	Households or individuals, often as consumers	Hypothetical, question framing issues,
	All ecosystem services, but often amenities. Incremental service levels, controlling for bundles			Information burden
				Usually not spatially explicit
Production	Use values	Neighborhood and regional scale	Natural scientists, experts	Requires spatially explicit biophysical modeling.
Function/Damage cost Replacement cost	Regulating services Use values	Building, streetscape, neighborhood level municipal infrastructure	Engineers, experts	Determining service equivalence for man-made replacement; depends on health and safety standards
	All services, but often regulating services			

Produced by Barton et al. (2012)

Table 11.8 Socio-cultural values of ecosystems and biodiversity

Socio-cultural values	Explanation	References
Spiritual values	In many places, especially among peoples with animistic religions, ecosystems and biodiversity are deeply intertwined with spiritual values	Stokols (1990)
Sense of place	Emotional and affective bonds between people and ecological sites	Altman and Low (1992), Feldman (1990), Williams et al. (1992), Norton and Hannon (1997)
Sense of community	Feelings towards a group and strength of attachment to communities	Doolittle and McDonald (1978), Chavis and Pretty (1999)
Social cohesion	Attachment as source of social cohesion, shared interests, and neighborhood participation	Bennett (1997), Gotham and Brumley (2002), Kázmierczak (2013)

Produced by Gómez-Baggethun (2013)

process, use of locally defined metrics, and valuation methods based on qualitative description and narration. A set of values that may be labeled as socio-cultural and associated descriptions is provided in Table 11.8.

11.3.1.4 Health Values

Multiple connections between urban vegetation and human health have been identified (Tzoulas et al. 2007; Bowler et al. 2010a), and the study of the links between green areas, human health and recovery rates is a rapidly expanding field of research (Grahn and Stigsdotter 2003). For example, access to green space in cities was shown to correlate with longevity (Takano et al. 2002), with recovery from surgeries (Ulrich 1984) as well as with self-reported perception of health (Maas 2006; van den Berg et al. 2010a). Proximity to green space reduced stress in individuals (Korpela and Ylén 2007), and children with attention deficit disorder have showed improved alertness (Taylor and Kuo 2009). Evidence also exists of other health benefits that correspond to green space availability (Hu et al. 2008; Bedimo-Rung et al. 2005; Ohta et al. 2007). Kaczynski and Henderson (2007) reviewed 50 quantitative studies that looked at the relationship between parks and physical activity and found that proximity to parks was associated with increased physical activity.

Green spaces have also been shown to influence social cohesion by providing a meeting place for users to develop and maintain neighborhood ties (Maas et al. 2009; Kázmierczak 2013). Other studies suggest that urban ecosystem services like air pollution reduction (Lovasi et al. 2008; Pérez et al. 2009) and urban cooling (Bowler et al. 2010b) have multiple long term health benefits. However, although the evidence of most studies suggests that green spaces have beneficial health effects, it should be noted that establishing a causal relationship has proven very difficult (Lee and Maheswaran 2010).

11.3.1.5 Environmental Justice Values

Social practices not only affect which ecosystem services are produced through the management of urban ecosystems (Andersson et al. 2007), but also who in society benefits from them (Ernstson 2012). Urban political ecology is the study of ecological distribution conflicts (i.e., conflicts on the access to ecosystem services and on the burdens of pollution). Environmental justice (Hofrichter 1993) represents the perspective within political ecology that conceives of balanced access to ecosystem services and balanced exposure to pollution across groups as a fundamental right. The notion was first used in relation to environmental conflicts in cities of the United States, where minority groups including African Americans, Latinos, and Native Americans bore disproportionate burdens of urban pollution and exposure to toxic waste (Martínez Alier 2005). While the bulk of the literature has focused on unequal exposure to pollution, the study of environmental conflicts related to unequal access to the benefits of ecosystem services are likely to become an important field of research for political ecology in the coming years. A recent study by Ernstson (2012) draws on empirical studies from Stockholm, Cape Town, and other cities to inform a framework to relate ecosystem services to environmental justice in urban areas.

Ecological distribution conflicts not only emerge from unequal access to ecosystem services within cities but also from asymmetries in the appropriation of ecosystem services by cities vis-à-vis their surrounding environment and more distant regions (Hornborg 1998). Extensive research has shown that urban growth depends on the appropriation of vast areas of ecosystem services provision beyond the city boundaries (Folke et al. 1997; Rees 1992; Rees and Wackernagel 1996). Thus, an important associated value of urban ecosystem services resides in their potential to reduce the ecological footprint of cities, and thus, cities' ecological debt to the non-urban environment. Building on the ecosystem services concept, Gutman (2007) calls for a new rural–urban compact, where cities channel more employment opportunities and more income to the rural areas in exchange for a sustainable supply of products and ecosystem services provided by restored rural environments.

11.3.1.6 Insurance Values

Urban ecological infrastructure and ecosystem services can play a major role in increasing the resilience of cities through enhancing their ability to cope with disturbance and adapt to climate and other global change. The contribution of ecological infrastructure and ecosystem services to increased resilience and reduced vulnerability of cities to shocks has been referred to as a form of insurance value (Gómez-Baggethun and de Groot 2010). Ecosystem services that are critical to the resilience of cities in response to specific disturbances include urban temperature regulation, water supply, runoff mitigation, and food production. For example, urban temperature regulation can be critical to buffer the effects of heat waves

Table 11.9 Sources of resilience and carriers of social-ecological memory to deal with disturbance and change in urban allotments

Category	Examples found in allotment gardens
Habits/rituals (*participation*)	Imitation of practices, exchange of seeds, embodied habits
Oral tradition (*participation*)	Ongoing negotiations, mentor programs, daily small talk
Rules-in-use (*reification*)	Norms of social conduct, norms towards the environment, property rights
Physical forms/artifacts (*reification*)	Written material, pictures, the gardens, tools, stories
External memory sources	Media and organizations external to individual allotment gardens

Produced by Jansson (2012), modified from Barthel et al. (2010)

(Lafortezza et al. 2009; EEA 2010; Depietri et al. 2012), ecological infrastructure that enhances water supply can increase resilience to drought, and runoff mitigation provided by urban vegetation can reduce the likelihood of damages by flooding and storms (Villarreal and Bengtsson 2005).

Special attention has been given to the role that food production in urban allotments can play in increasing food security and building resilience to shocks, especially in times of economic and political crisis (Smit and Nasr 1992; Moskow 1999; Page 2002; MA 2005; UNEP 1996). The Millennium Ecosystem Assessment notes that "for many of today's urban dwellers, urban agriculture provides an important source of food and supplementary income" (MA 2005, p. 810). In Cuba, urban agriculture that emerged in response to the decline of Soviet aid and trade and the persistence of the trade embargo came to play a major role in food security (Altieri et al. 1999; Moskow 1999). Likewise, urban agriculture has provided an important safety net for landless peoples in sub-Saharan Africa (Maxwell 1999). At present, urban social movements associated with allotments gardens are emerging all around Europe (Barthel et al. 2010). Table 11.9 provides examples of how urban allotments can contribute to increasing resilience and storing social-ecological memory to deal with shocks.

Recent contributions have also noted the role of urban ecosystems in maintaining living bodies of local ecological knowledge (Andersson et al. 2007). Because local and traditional knowledge systems embed accumulated knowledge and practices to cope with environmental change, maintaining these bodies of knowledge can be essential for resilience to shocks (Barthel et al. 2010; Gómez-Baggethun et al. 2012).

Measuring the insurance value of resilience remains a challenging task. For example, there is growing evidence that increased resilience can bring multiple indirect economic benefits (Walker et al. 2010). Yet, translating the value of resilience into monetary metrics can be complicated and in some cases also useless. Because the economic value of ecosystem services is affected by the distance to ecological thresholds, trying to capture the value of resilience with economic valuation at the margin can be risky and even misleading (Limburg et al. 2002);

when thresholds are close, small changes can trigger abrupt shifts in ecosystem services and related values (Scheffer et al. 2001; Walker and Meyers 2004; Pascual et al. 2010).

11.4 Ecosystem Services and Urban Governance

11.4.1 Connecting Ecosystem Service Values to Urban Policy and Governance

Local authorities in many cities throughout the world are looking for innovative ways to maintain and increase ecological infrastructure as a part of urban planning and design (Rosenzweig et al. 2009; see also Chap. 27). Yet, many studies have suggested that the ability of local authorities to implement ecological infrastructure is not sufficiently recognized and hence lacks further integration into spatial planning systems (Kruuse 2011). Economic and non-economic valuation of ecosystem services is often demanded by policy makers and practitioners as supporting information to guide decisions in urban planning and governance. Ways in which valuation can inform urban planning include awareness raising, economic accounting, priority-setting, incentive design, and litigation, thus broadly reflecting the objectives of "recognizing, demonstrating, and capturing value" as suggested in the TEEB report (TEEB 2010) (Fig. 11.2).

The demand for accuracy and reliability of valuation methods increase successively when moving from a policy setting, requiring simply awareness raising (e.g. regarding costs of ecosystem service loss); to including ecological infrastructure in accounting of municipal assets; to priority-setting (e.g. for location of new neighborhoods); to instrument design (e.g. user fees to finance public utilities); or finally to calculation of claims for damage compensation in a litigation (e.g. siting of locally undesirable land-uses (LULUs)). While several monetary valuation methods are potentially applicable at different spatial scales, valuation studies in urban areas for support in any given decision-making context are more demanding because of requirements for higher spatial resolution and multiple scales of analysis. Using valuation of urban ecosystem services for decisions about ecological infrastructure requires attributing service values to the particular assets at specific locations. For regulating services this requires some form of spatially explicit biophysical modeling which increases valuation costs with increasing geographical scale and resolution (Fig. 11.2).

11.4.2 Ecosystem Services in Urban Planning and Design

A better understanding of ecosystem services, their spatial characteristics and interrelations is very much needed in order to move ecosystem services from an

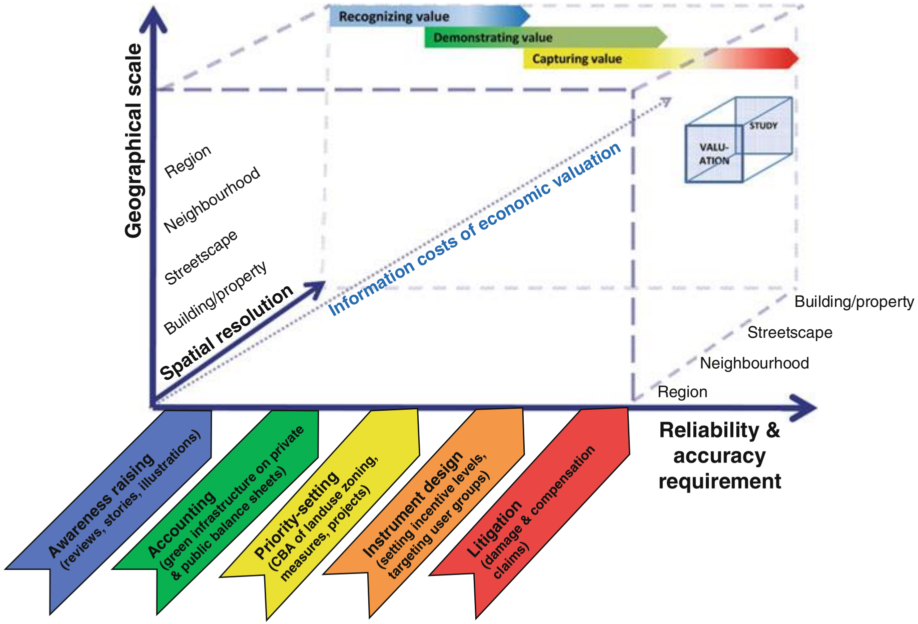

Fig. 11.2 Trade-offs between scale, resolution, and accuracy in recognizing, demonstrating and capturing values in different decision-support contexts of valuation (Source: Adapted from Gómez and Baggethun 2012; Modified from Gómez-Baggethun and Barton 2013, p. 241. Published with kind permission of

assessment tool to a practical instrument for planning and design (Troy and Wilson 2006). For a discussion of patterns and trends in urban biodiversity and design, with applications to ecosystem services, see Chap. 10. Ecosystem service research is slowly merging with landscape ecology and spatial planning to address the issue of the scales and structures related to the generation and utilization of ecosystem services (see e.g., Fisher et al. 2009). There are several possible spatial relationships between the scale at which one ecosystem service is generated and the scale at which people may benefit from it. Some services can only be enjoyed at the source (e.g., shading from vegetation or many recreational uses of green areas), whereas others spill over into adjacent areas (e.g., noise reduction, wind breaks and pollination). Such spill-over may be unidirectional or directional, the latter partly due to physical geography (e.g., of waterways, topography, and location of roads) and the location of the beneficiaries. The connection between ecosystem service source areas and end-users is mediated by social structures such as built infrastructure and institutions defining access to land. There are a wide range of solutions for providing the people in different cities with similar ecosystem services and city-specific scales of relevance for addressing each ecosystem service.

Spatial scales and landscape structure affect the possibilities and constraints for ecosystem service planning. Efforts to address bundles of services to create or maintain multifunctional landscapes have seen considerable progress in the last decade. On larger scales, access to multiple ecosystem services can be achieved by ensuring generation of different ecosystem services in different parts of the landscape—as long as they are accessible to the users (see Brandt and Vejre 2003). However, the scale in these studies is often coarse and is not well suited to pick up the small-scale heterogeneity of the urban landscape. When the potential service-providing areas are few and situated in a matrix of many and diverse users, the number of services expected from each of these areas is likely to increase. Multiple interests coupled with limited size will highlight trade-offs between services and potentially lead to conflicts.

The urban mosaic is often complex and characterized by multiple spatial boundaries between different land-uses. With such heterogeneity, relative location and context can be expected to be especially important. Some ecosystem services will rely on species that require easy access to two or more habitat types (Andersson et al. 2007). For example, Lundberg et al. (2008) described how long-term maintenance of an oak dominated landscape with highly valued cultural and aesthetical qualities in Sweden depends also on patches of coniferous forest, the latter providing the main seed disperser, Eurasian Jay (*Garrulus glandarius*), with breeding habitat. Other ecosystem services such as pest control or pollination rely on close proximity to a source area (e.g. Blitzer et al. 2012).

Many ecosystem services are directly mediated or provided by different organisms (Kremen 2005) and can thus be addressed through a focus on these organisms. From a temporal perspective, long-term provisioning of ecosystem services within cities raises concerns about population dynamics, including the risks of extinction (at least on the local scale) and potential for re-colonization. For many species, habitat within cities may be perceived as quite fragmented, suggesting not only that future urban development should try to avoid further fragmentation but also that increased connectivity should be one of the prime objectives for restoration efforts (Hanski and Mononen 2011). It seems reasonable that the general character of urban green structures should be as similar as possible to that of the hinterlands in order to benefit the most from potential near-city source areas of ecosystem-service-providing organisms. To draw on these source areas, cities need a connected green structure that reaches all the way through urban and peri-urban areas into the rural.

From a spatial perspective, at least two distinct strategies for ensuring ecosystem service generation can be identified (see Forman 1995). The first draws on traditional conservation planning and is foremost concerned with enhancing and securing internal values within a bounded area, for example biodiversity or recreational opportunities within a protected area. This approach advocates large areas, and if spatial issues are considered at all it is usually in terms of green area networks where "green areas" are not necessarily the same as ecosystem service generating areas. The second strategy adopts more of a landscape management perspective in which the focus is on enhancing the performance of all parts of the landscape (see Fahrig et al. 2011), not just the few large areas suggested in the first approach. Instead, this perspective highlights the potential of smaller units interspersed throughout an area (for example, small clumps of trees mixed with residential development may enhance overall biodiversity or aesthetic values). The two approaches are by no means incompatible or always opposing, but their focus, prioritizations, and trade-offs differ. Both are needed and address different aspects of ecosystem services.

11.5 Ecosystem Services in Three Cities

Since appropriate management strategies for ecosystems outside and within cities may differ due to, for example, the difference in social, ecological and economic pressures, it is essential to acquire a fairly detailed outline of a city's ecosystem service needs, both within and outside the city boundaries. The information on where different ecosystem services are being produced (i.e., the location of the production unit), whether inside the city itself or elsewhere, is also significant in determining how vulnerable or resilient a city and its inhabitants are to potential disruptions in the generation of ecosystem services when exposed to change. Assessing restoration/transformation potential in the urban landscape is important for mitigating disruptions in service generation and can be a powerful tool for urban planning. Furthermore, since the generation of ecosystem services in a specific ecosystem often affects the generation potential in other ecosystems, it is also crucial to identify spill-over effects. In the following tables a review of ecosystem services for three different cities are presented: Cape Town, New York, and Barcelona (in-depth assessments on Cape Town and New York are presented in Chaps. 24 and 19, respectively).

11.5.1 Cape Town

The city of Cape Town is home to approximately 3.7 million people. It is characterized by apartheid city planning with racially distinct urban residential areas and a massive disparity in development between these areas. Key socio-economic challenges within the city include the provision of housing, education, transport infrastructure, nutrition and healthcare. Current development strategies acknowledge these issues and also recognize that population growth and migration to this city will increase the magnitude of these challenges.

The Cape Floristic region in which Cape Town is located is a globally recognized biodiversity hotspot. The city is home to 19 of the 440 national vegetation types, and hosts 52 % of the nationally critically endangered vegetation types (Rebelo et al. 2011). Cape Town is also a major tourism destination in Africa, a function of the heterogeneous natural environment, which provides multiple other ecosystem services. The Table Mountain National Park, which is surrounded by the city, is a key conservation area for retaining both the biodiversity as well as the ecosystem services that support local residents (Anderson and O'Farrell 2012). The lowland areas within the City area are not well protected and are under extreme and constant pressure of transformation, particularly for much-needed housing (see Chap. 24). In a recent assessment of the ecosystem services found within Cape Town, O'Farrell et al. (2012) examined the effect of transformation on a number of services by contrasting historical landscape structure (500 years prior) with current conditions, and in addition explored potential future transformation effects (using a scenario where all undeveloped land not under formal conservation protection was transformed to formal housing) (Fig. 11.3). Their study indicated that all services had decreased from their potential level; provisioning services were particularly affected, with reductions between 30 and 50 % depending on the service. The study highlights the significance of the loss of regulating services, which while less threatened than other services in the study, are potentially more problematic when lost, as these services must be delivered in situ. Whereas provisioning services can be outsourced to areas beyond the city boundaries (such as the provision of food), this is not possible with most regulating services (such as flood mitigation and coastal zone protection) (see Table 11.10).

Recognized important ecosystem services to the City of Cape Town are the provision of water supply, flood mitigation, coastal zone protection and tourism (see Table 11.10). Many of these services, and the biodiversity and ecological infrastructure on which they depend, have been degraded. There are a number of examples where there are programs and projects in place aimed at attempting to restore these and thereby enhance ecosystem service benefits.

Invasive alien plants have become a dominant feature in the catchments that supply Cape Town with water. These plants use significantly more water than the indigenous vegetation, and thereby decrease surface run-off and ultimately water supply and security (Le Maitre et al. 1996). The Working for Water program was established in 1995 as a direct response to the loss of this critical resource

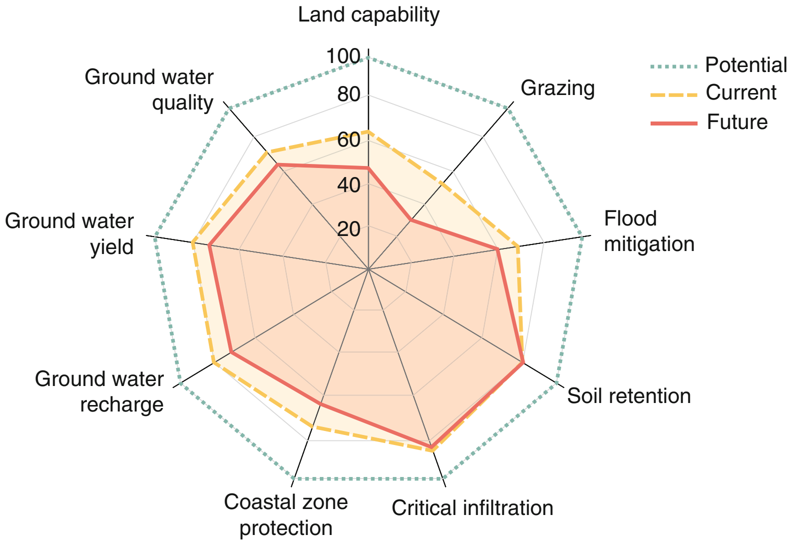

Fig. 11.3 Present and potential changes in ecosystem service supply for Cape Town shown as a percentage of the potential service produced (Modified from O'Farrell et al. 2012, p. 6. Published with kind permission of © Ecology and Society 2012. All Rights Reserved)

(Van Wilgen et al. 1998) (see Chap. 24). Clearing teams are continuing to remove invasive plants from these catchments in an attempt to restore optimal water flows, which are critical to the growth and development of the city.

Within this restoration space, interventions are emerging at many tiers of society. Smaller initiatives driven by local communities or smaller government agencies aimed at restoring natural vegetation have been shown to have considerable ecosystem service benefit (Avlonitis 2011). While these often emerge in a cultural space, or towards recreational ends, there are evident ecological spin-offs. A study by Avlonitis (2011) has shown the potential of communities to work in conjunction with larger government initiatives such as Working for Wetlands, where community initiative and labor are used to promote the development of indigenous vegetation gardens. Here, cultural agendas are forwarding the restoration of regulating services. This study points to the value of targeting sites where multiple agendas can be met through intervention. Restoration initiatives should take advantage of community interest and energy and align interventions with local cultural needs. An examination of the relevance of urban green space to the local population shows multiple opportunities to find these nodes of congruent opportunity (Pitt and Boulle 2010).

The opportunity for restoring the regulating services of coastal zone protection are largely lost where there has been considerable historic development close to the coastal zones. These areas tend to be associated with erosion problems and are a major financial sink for City management who strive to protect settlements, often

Table 11.10 Ecosystem services in Cape Town

Ecosystem services	Location of production: local, regional, global	Production unit	Spill over effects	Unit affected by spill over	ES affected	Restoration, transformation potential	References
Provisioning							
Food (broken down below)							
(Vegetables)	Local	Private gardens, community gardens	Recreation, food security, reduced biodiversity, community cohesion, pollution and N & P into ground water	Biodiversity remnant patches, open space, ground water sources affected locally	Positive recreation benefits, livestock production and water purification negatively impacted		Crush et al. (2010); Battersby (2011)
(Crop production)	Local	Urban agricultural areas	Biodiversity loss, green house gas (GHG) emissions.				O'Farrell et al. (2012)
(Livestock)	Local, regional	Vegetated areas, urban open space	GHG emissions, biodiversity loss, social well-being, cultural identity	Biodiversity remnant patches, open space	Cultivation, biodiversity conservation, negatively impacted	Land manage-ment programs to reverse degradation are possible	O'Farrell et al. (2012); Lannas and Turpie (2009); Battersby (2011)

(Freshwater fish)	Local	Dams & wetlands	Food security and conservation	Dams & wetlands	Recreation	Possible with reduced extraction	
(Seafood)	Local, Regional, global	Ocean, lagoons, estuaries	Ecological functioning (supporting services)	Ocean, lagoons, estuaries	Recreation	Possible with reduced extraction	Turpie et al. (2003)
Natural medicinal, ornamental and food resources	Local, regional	Natural vegetation	Possible population level impacts and loss of biodiversity	Natural vegetation	Biodiversity conservation	Possible with reduced extraction	Petersen et al. (2012)
Drinking water supply	Berg, Breede Disa, Palmiet and Steenbras river catchments	Surrounding mountain catchment and watersheds	Local climate regulation; Recreation, biodiversity conservation in mountain catchments, invasion by alien plants	Catchments	Agriculture	Clearing catchments of invasive alien plants	Brown and Magoba (2009); Van Wilgen et al. (1998)
Fuel wood	Local	Natural vegetation (degraded)	Positive biodiversity effects – clearing of invasive alien vegetation	Open spaces, conservation areas, catchments	Natural vegetation remnants (positive effect)	Harvesting fuel wood typically a restoration benefit as harvested species are invasive alien plant species	Lannas and Turpie (2009)

(continued)

Table 11.10 (continued)

Ecosystem services	Location of production: local, regional, global	Production unit	Spill over effects	Unit affected by spill over	ES affected	Restoration, transformation potential	References
Fiber harvest	Local and regional	Wetlands, coastal plains	Habitat impacted	Wetland edges, coastal plains (restios)	Water quality regulation (negatively affected) Soil retention		
Regulating							
Water purification	Berg, Breede Disa, Palmiet and Steenbras river catchments	Wetlands, rivers, vleis	Positive biodiversity and conservation impacts, agricultural practices curtailed	Mountain catchments, river systems and their buffer areas	Agriculture		Brown and Magoba (2009)
Water infiltration/groundwater recharge	Local	Natural vegetation, gardens, open space	Reduced flooding potential, increased pollution of ground water sources if water polluted	Local level		Vegetation and biodiversity restoration	O'Farrell et al. (2012)
Soil retention	Local	Natural vegetation, gardens, open space	Restricted agricultural practices on steep slopes with high rainfall			Vegetation and biodiversity restoration	O'Farrell et al. (2012)

Carbon sequestration	Local and regionally	All natural vegetation remnants, plantations, parkland street trees	Climate regulation, shade provision, high water consumption effects possible, and biodiversity impacts	Catchments and water bodies	Water provision	Vegetation restoration	
Flood control and mitigation	Local and regional	Natural vegetation remnants	Filtration and absorption of water and waterborne pollutants, agricultural land use food provision restricted, recreational use of rivers areas enhanced	Open space, remaining natural remnants, river buffer areas	Cultivation areas restricted (negative effect), ground water recharge enhanced	Wetland and vegetation restoration	O'Farrell et al. (2012); Musungu et al. (2012); Bouchard et al. (2007)
Coastal storm surge protection	Local	Natural vegetation remnants	Restricted use of coastal environments	Coastal dune systems (extending from water's edge to 1 km inland)		Coastal vegetation restoration	O'Farrell et al. (2012)

(continued)

Table 11.10 (continued)

Ecosystem services	Location of production: local, regional, global	Production unit	Spill over effects	Unit affected by spill over	ES affected	Restoration, transformation potential	References
Noise reduction	Locally	Street trees, plantations,		Heat island reduction, carbon sequestration		City planting, recreation	
N retention	Regionally	Wetlands				Restoration of wetlands, creation of new wetlands	
Pollination	Local, regionally	Gardens, parks, golf courses, natural vegetation remnants and nature reserves	Retaining pollinators requires maintaining ES production units, transformation to certain crops and timber is restricted	Local open space effects	Production of timber, crops	Increase % semi-natural areas, connectivity	O'Farrell et al. (2012)

Cultural							
Health	Local, regional, national	Green open space			Pollination, Biological control, infiltration, flood mitigation, coastal protection are all positively affected	Macro-scale urban planning, creative design of open space	
Recreation	Local, regional	Beaches, nature reserves, parks, urban green space, gardens	Sense of place; education; health; increase in property value and tax revenue	Individuals, communities, Neighbourhoods			De Wit et al. (2012)
Tourism	Local, regional	All natural assets					De Wit et al. (2012); O'Farrell et al. (2012)

(continued)

Table 11.10 (continued)

Ecosystem services	Location of production: local, regional, global	Production unit	Spill over effects	Unit affected by spill over	ES affected	Restoration, transformation potential	References
Education opportunities	Local, regional, national	Conservation areas, wetlands, rivers, estuaries, beaches	Environmental values; improved ecosystem function	Individuals, communities, urban vegetation remnants and reserves, urban waterways, breaches	All locally-produced services		O'Farrell et al. (2012)
Sense of place cultural ties	Local, national	Rural areas, usually distant, national parks	Increased well-being				De Wit et al. (2012)
Cultural rites/initiation	Local	Open green natural space, isolated area	Affects social cohesion, human-nature interactions,	Individuals, communities	Sense of well-being, sense of place		

Produced by O'Farrell (2013)

with expensive engineering interventions. Opportunities need to be sought for the effective incorporation of existing regulating services into ongoing and future developments. Large buffer zones protecting coastal dune systems with an associated functioning ecology are a critical service and one likely to become more so with projected sea rise and increased storm surge. A spatial plan needs to be developed assessing where restoration might be an option, and where engineering interventions must be considered. Remnant areas need the strictest protection as the city continues to grow within these areas (see Chap. 24 for additional discussion on this challenge).

There are numerous cases where ecosystem services may be effectively delivered outside of the natural indigenous biodiversity framework. For example, certain urban agricultural areas may be effective sites of groundwater recharge serving as a site of effective regulation, and forest plantations provide much enjoyed recreation sites serving an important cultural service. What is apparent is a suite of emerging novel ecosystems that speak to ecosystem service delivery, but do not necessarily meet biodiversity conservation goals. The high endemic biodiversity and global conservation significance of the vegetation of South Africa's Western Cape means that conservation agendas tend to predominate in this discourse. This is where ecosystem services and biodiversity conservation agendas may diverge. Future spatial planning and development as well as restoration activities must pay due attention to both conservation priorities and the ecosystem service needs and delivery potential of the remaining open spaces within the city.

11.5.2 New York

New York City is a classic example of a complex social-ecological system (SES) (McGrath and Pickett 2011; McPhearson 2011) situated in a large urban region along the northeast coast of the United States. The metropolitan region encompasses a dense urban core, surrounded by sprawling suburban and exurban development housing over 20 million people with unparalleled ethnic and social diversity. New York is both the largest city in the U.S. and the densest. Though people may often think of the city as a network of tightly-knit architectural forms and elaborately paved infrastructure, New York has a higher percentage of open space than any other major city in the U.S. (The Trust for Public Land 2011).

Throughout the five municipal boroughs of Manhattan, Brooklyn, Queens, Bronx, and Staten Island, there are approximately 11,300 ha of city parkland—nearly 40 % of which (4,450 ha) is still natural—harboring freshwater wetlands, salt marshes, rocky shorelines, beaches, meadows and forests. Ensconced within these ecosystems are more than 40 % of New York State's rare and endangered plant species. As a result, scientists are beginning to view New York City as an ecological hot spot—more diverse and richer in nature than the suburbs and rural counties that surround it. Regional ecosystems beyond the city boundaries also provide critical ecosystem services to New Yorkers including drinking water, climate regulation, food production, recreation, and more, some of which have yet to be documented and described (Table 11.11).

Nonetheless, valuation of ecosystem services in New York has moved from economic valuation assessment of wetlands and forests to planning and legislation aimed at expanding and improving the management of ecosystems in the city for the purpose of improving the health and well-being of urban residents. The most prominent example is the recent 20-year economic and environmental sustainability plan, PlaNYC, which includes 132 initiatives. These ambitious initiatives range from revamping aging infrastructure to cutting greenhouse gas emissions 30 % by 2030 (New York City 2011). Since its inception, PlaNYC has gained tremendous attention both nationally and internationally and has been acknowledged around the world as one of the most ambitious and pragmatic sustainability plans anywhere (see Chap. 19, Local Assessment of New York).

One of the many ecosystem service-focused initiatives of PlaNYC is MillionTreesNYC, a public-private partnership between the NYC Department of Parks & Recreation and the New York Restoration Project, with the goal of planting and caring for one million trees across the city's five boroughs over the next decade. By planting one million trees, New York City intends to increase the size of its urban forest by 20 %. Since MillionTreesNYC began in 2007, over 600,000 trees have already been planted on city streets, private land, and public parkland. The impetus for such a significant investment in trees is the ecosystem services that the urban forest provides to city residents. One recent study by the U.S. Forest Service put the compensatory value of the city's urban forest at over $5 billion (Nowak at el. 2007a). Nowak and colleagues estimated that the urban forest stores 1.35 million tons of carbon, a service valued at $24.9 million. The forest sequesters an additional 42,300 t of carbon per year (valued at $779,000 per year) and about 2,202 t of air pollution per year (valued at $10.6 million per year; Nowak et al. 2007a). Urban trees provide a direct ecological service to cities by reducing urban surface and air temperatures through shading and evapotranspiration, yet the indirect effects of trees are just as important. For example, a cooler city leads to substantial reductions in energy use for air-conditioning. The U.S. Forest Service found that New York City's street trees provide an estimated $27 million a year in energy savings through shading buildings. Trees can also regulate local surface and air temperatures by reflecting solar radiation and shading surfaces, such as streets and sidewalks that would otherwise absorb heat. Decreasing the heat loading of the city and thereby mitigating the urban heat island effect may be the most important ecological service trees provide to cities (McPhearson 2011). If an urban area like New York City eventually adds one million additional trees to its urban forest, the total cooling effect could decrease the heat of the city by a full degree or more (Rosenzweig et al. 2009).

Urban trees also capture rainfall on their leaves and branches and take up water through their roots, acting as natural stormwater capture and retention devices. Capturing stormwater to prevent pollution loading to local streams, rivers, and estuaries is a major goal of PlaNYC. Street trees in NYC intercept almost 900 million gallons of stormwater annually, or 1,500 gallons per tree on average. The total value

Table 11.11 Ecosystem services in New York

Ecosystem service	Location of production: local, regional, global (%)	Production unit	Spill over effects	Unit affected by spill over	ES affected	Restoration, transformation potential	References
Provisioning							
Food (broken down below)							
(Produce and crops)	**Local** < 1%[a]	Private gardens, community gardens	Food security, stormwater retention, energy efficiency, and waste reduction; increased habitat and biodiversity; urban landscape beautification, increased property values and tax base	Floral and faunal species, individuals, communities	Recreation, sense of place, education, social-ecological memory	Expanding local food movement and urban farming	Ackerman (2011); Farming Concrete (2010); Gittleman et al. (2010); Voicu and Been (2008); McPhearson and Tidball (2012)

(continued)

Table 11.11 (continued)

Ecosystem service	Location of production: local, regional, global (%)	Production unit	Spill over effects	Unit affected by spill over	ES affected	Restoration, transformation potential	References
	Regional – New York Metropolitan Area	Agriculture fields	Food security, decreased water quality, N, P leakage, Biodiversity loss, GHG emissions	Birds, bees, wildlife, individuals, communities, wetlands, lakes, rivers and streams, estuary	Seafood production, water quality, recreation, sense of place, educational opportunities, carbon storage, carbon sequestration	2.2 % NYC; 34 % for New York State; value of sales of organic production \$54 million; 131,796 acres; many regional groups work on sustainable and organic agriculture practices (e.g., NYSWAG, NOFA-NY)	USDA (2007); Peters et al. (2009, 2007)
	Global[b] – overwhelming majority (common knowledge)	Agriculture fields	Biodiversity loss, GHG emissions, N, P leakage	Birds, bees, wildlife, individuals, communities, wetlands, lakes, rivers and streams, estuaries, coral reefs	Soil building, seafood production, carbon storage, carbon sequestration		

(Livestock)	**Regional –** New York Metropolitan Area	Agriculture fields	Food Security, GHG emissions, biodiversity loss	Birds, bees, wildlife, wetlands, lakes, rivers and streams, wildlife, individuals, communities, airshed[c]	Seafood production, water quality, carbon storage, carbon sequestration		USDA (2007)
(Seafood)	**Regional –** 13 % of seafood purchased by Fulton Market is from New York fishermen and other NY suppliers	Lakes, rivers, wetlands, estuaries, ocean	Sustainability of fisheries, food security	Regional fisheries	Recreation, educational opportunities, sense of place		New York Sea Grant (2001)
	Global – 20 % of seafood purchases by Fulton Market are from foreign sources and 67 % are from other US states	Lakes, rivers, wetlands, estuaries, ocean	Decreased biodiversity, sustainability of fisheries	Global fisheries	Recreation		New York Sea Grant (2001)

(continued)

Table 11.11 (continued)

Ecosystem service	Location of production: local, regional, global (%)	Production unit	Spill over effects	Unit affected by spill over	ES affected	Restoration, transformation potential	References
Drinking water supply	**Regional**: 100 % Catskill-Delaware Watershed. In previous years, 10 % came from Croton watershed	Watershed[d]	NYC Dept. of Environmental Protection funds the Watershed Agricultural Council to implement water quality enhancement programs including purchasing conservation easements and paying farmers to manage farmland for water quality. While some of these programs support agriculture, they may also remove land from agricultural production and limit logging	Agricultural land, forests, wetlands, lakes, rivers and streams	Recreation, sense of place, social-ecological memory, education opportunities, food, wood and fiber	The Watershed Agricultural Council has an ongoing commitment to supporting Whole Farm Planning, in which it incentivizes farmers to manage risks to the water supply, protecting watershed land through conservation easements, and incentivizing landowners to engage in forest management planning.	Pires (2004); NYC Environmental Protection (2010a); New York State Department of Environmental Conservation (2010); Watershed Agricultural Council (2011, 2012)

Wood and fiber	**Regional**	Forest	Carbon emissions from burning biomass fuel (firewood) and timber harvesting and processing, changing forest community structure and function	Airshed, forests, individuals, communities	Air quality (particulate matter), carbon storage, carbon sequestration, sense of place, social-ecological memory, recreation	NY State DEC forest resource assessment and strategy Keeping NY's forests as forests New York State Plan to preserve forest ecology 2010–2015 Forest resource assessment	New York State Department of Environmental Conservation (2010)
Regulating							
Drinking water quality enhancement	100 % Regional	Watershed forest	Enhanced water quality supports aquatic life and recreation	Wetlands, lakes, rivers and streams	Recreation, sense of place, social-ecological memory, seafood	Watershed Agricultural Council's ongoing watershed protection programs reduce pollutants in NYC's drinking water supply	NYC Environmental Protection (2010a)

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Table 11.11 (continued)

Ecosystem service	Location of production: local, regional, global (%)	Production unit	Spill over effects	Unit affected by spill over	ES affected	Restoration, transformation potential	References
Flood control	Local	Urban forest	Filtration and absorption of water and waterborne pollutants	Wetlands, lakes, rivers and streams	Stormwater quality	Using ecological infrastructure to capture 1st inch of rainfall on 10 % of impervious areas in combined sewer watersheds would result in reduced combined sewer overflows of 1,514 million gallons yearly	USDA Forest Service (2007); NYC Environmental Protection (2010b)
Stormwater quality enhancement (N, P, coliform, Total Suspended Solids)	Local, Regional	Watershed, Forest	Absorption of water	Wetlands, lakes, rivers and streams	Flood control	Could vegetate between 1,085 and3,255 acres of impervious surface to absorb pollutants	NYC Environmental Protection (2010b)

Air purification/air quality regulation	Local, Regional	Forests and Other Green Spaces	Reduced atmospheric deposition of NO_x into waterways (US Environmental Protection Agency 2001), plants can increase allergens in outdoor air, tree maintenance results in increased CO_2, trees emit biogenic volatile organic compounds	Wetlands, lakes, rivers and streams	Seafood, drinking water quality, recreation, sense of well-being	Local: Increased tree cover of 11,836 acres (6 % increase to = 30 % total canopy cover) would add 91.3 metric t/year additional pollution removal; MillionTreesNYC restoration effort will increase air purification	Grove et al. (2006); McPhearson (2011); Nowak et al. (2007a); U.S. Environmental Protection Agency (2001)

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Table 11.11 (continued)

Ecosystem service	Location of production: local, regional, global (%)	Production unit	Spill over effects	Unit affected by spill over	ES affected	Restoration, transformation potential	References
C sequestration	Local, Regional	Forests and other green spaces		Airshed		Local: Canopy cover could be increased by 11,836 acres (6 % increase to = 30 % total canopy cover) = 2,486 t/year additional C sequestration MillionTreesNYC restoration effort will increase C sequestration	Grove et al. (2006); McPhearson (2011); Nowak et al. (2007a)
C storage	Local, Regional	Forests and other green spaces		Airshed		Canopy cover could be increased by 11,836 acres (6 % increase to = 30 % total canopy cover) = 80,485 t additional C storage MillionTreesNYC restoration effort will increase C storage	Grove et al. (2006); McPhearson (2011); Nowak et al. (2007a)

Temperature regulation	Local, Regional	Forests and other green spaces	Shading and evapotranspiration lowers air temperature and results in less use of air conditioning, reduced O_3 formation, and avoided CO_2 emissions (Nowak et al. 2007a)	Individuals, communities, airshed	Air purification, sense of well-being	Greening half of NYC roofs (7,698 acres) would reduce temperature by 0.8 °F. MillionTreesNYC restoration effort will help regulate temperature	McPhearson (2011); Nowak et al. (2007a); NYC Environmental Protection (2010b); Rosenzweig et al. (2009)
Noise mitigation	Local	Forests and other green spaces	Psychological benefits, people engage in more outdoor activity	Individuals, communities	Recreation, sense of well-being, sense of place		
Cultural							
Aesthetic value	Local	Forests and other green spaces	Increased property value, gentrification, people engage in more outdoor activity	Individuals, communities	Recreation, sense of well-being, sense of place	Trees increase nearby property values by $90/tree Community gardens in NYC add 9.4 % to the value of properties around them	USDA Forest Service (2007); Voicu and Been (2008)

(continued)

Table 11.11 (continued)

Ecosystem service	Location of production: local, regional, global (%)	Production unit	Spill over effects	Unit affected by spill over	ES affected	Restoration, transformation potential	References
Recreation	Local	Pocket parks, Neighborhood parks, destination parks, regional parks	Sense of place; education; health; increase in property value and tax revenue (Appleseed 2009)	Individuals, communities		The city adopted a standard of 1.5 acres per 1,000 population in addition to specific PlaNYC goals: park within 10 min walk for all population, expansion of park land by additional 2,700 acres, increased hours, increased usage functions Planned investment: 400 million to be invested in new regional parks in the city[e]	New York City (2007, 2011)
Educational opportunities	Local, Regional	Forests, other green space, aquatic ecosystems, urban gardens, urban farms	Increased civic engagement, social connectedness, environmental values; improved ecosystem function	Individuals, communities, urban forest, urban waterways, airshed	All locally-produced services		Tidball and Krasny (2010); McPhearson and Tidball (2012)

Sense of place	Local, Regional	Forests, other green space, aquatic ecosystems, urban gardens, urban farms	People engage in more outdoor activities in their communities	Individuals, communities	Recreation, sense of well-being, education
Sense of well-being	Local	Forests, other green space, aquatic ecosystems, urban gardens, urban farms	People engage in more outdoor activities in their communities	Individuals, communities	Recreation, sense of place
Social-ecological memory	Local, Regional	Forests, other green space, aquatic ecosystems, urban gardens, urban farms	Can affect social cohesion, human-nature interactions, increased affinity for ecosystem stewardship	Individuals, communities	Sense of well-being, sense of place, recreation, education

Produced by McPhearson et al. (2013)

[a] Proposed calculation: based on average production per sq ft for different urban agricultural types: community gardens, urban farms, home gardens

[b] The notion that the majority of food arrives at NYC from great distances was already substantiated in 1913 (Miller et al. 1913)

[c] An airshed is an area of the atmosphere that shares an air supply and can become uniformly polluted

[d] A watershed is a land area in which water that is under and drains off it all flow to the same place

[e] There are survey results for what people want to see in the revamped regional parks: http://www.nycgovparks.org/email_forms/planyc_surveys/McCarren_Results.pdf

of this benefit to New York City is over $35 million each year. A comprehensive accounting of the ecosystem services of New York's urban forest and other green spaces is part of research in progress, but it is clear that urban ecological infrastructure is providing additional social and ecological benefits to the city including increased wildlife habitat, forestry products, materials for community projects, neighborhood beautification, places for social bonding, increased safety, neighborhood stability, and social-ecological resilience (Grove et al. 2006).

For example, ecological infrastructure in New York provides a number of cultural services to city. New York City's park system offers numerous recreational opportunities to residents from large urban parks such as Central Park in Manhattan and Prospect Park in Brooklyn, to playgrounds, sport fields and small pocket and neighborhood parks. While the city's park system is one of the largest in the world, PlaNYC acknowledges that many communities still lack sufficient access to park and open space. Therefore, the City has set a target of 1.5 acres of open space per 1,000 people, coupled with the goal of having a park within a 10-min walk for all city residents. To achieve these goals, the City has committed to expanding the park system by 2,700 acres, improving existing facilities and offering extended hours in various park facilities. US$400 million are slated for investment in the creation of new regional parks within the city boundaries (New York City 2007, 2011).

Ecological infrastructure is also important for the provisioning of food for New York residents (Table 11.11). Though only a small fraction of food consumed is produced locally, the vibrant and growing local food movement is one of the promising trends in urban ecosystem services. Urban gardens in private homes, community gardens, rooftop gardens and urban farms contribute to urban ecosystems by providing habitat to support biodiversity and increased resilience. In addition they provide varied ecosystem services such as runoff retention, recreation and education opportunities, and support sense of place and are sites for social-ecological memory. The New York local food movement is diverse, comprised of NGOs, research and education institutions, government organizations and many individuals. Programs such as the City's GreenThumb (http://www.greenthumbnyc.org/), Farming Concrete (http://farmingconcrete.org/), 596 acres (http://596acres.org/), Five Borough Farm (http://www.designtrust.org/projects/project_09farm.html) and many others are working tirelessly to convert built acres into ecologically sound, productive spaces. With over 1,000 gardens, 30 urban farms and 2,000 acres of still vacant land, the trend is only beginning to fulfill its potential.

That the human components of the social-ecological system are intimately tied to the ecological components through ecosystem services is becoming better understood in policy and planning in New York City. The last decade has shown significant progress towards resilience and sustainability planning, most recently through PlaNYC. Still, it will continue to be essential for city planners, managers, and policy makers to better understand trade-offs and synergies in the provisioning of ecosystem services in order to generate best practices for managing and enhancing biodiversity and ecosystem services in the New York metropolitan region.

11.5.3 Barcelona

Barcelona is a compact city located at the Mediterranean shore in North-Eastern Spain. The Barcelona Metropolitan Region (BMR) has been described as a circular structure, comprised of two extensive outer metropolitan rings, a dense middle ring and the municipality of Barcelona as the compact inner core (Catalán et al. 2008). The BMR, with around five million inhabitants—including the municipality of Barcelona with 1.62 million inhabitants—is the second largest urban area in Spain. Population density is relatively low in the outer rings and increases to over 16,000 inhabitants per km^2 in the inner core (Census 2012, IDESCAT), which makes Barcelona one of the densest cities in Europe. While the population size of the BMR showed stability within the last decades, its distribution pattern changed considerably. The horizontal expansion of the city—in form of a migratory movement from the dense core to outer rings of the BMR—more than doubled the size of the urbanized area since 1975 (Domene and Saurí 2007; Catalán et al. 2008). This urban sprawl movement has been described as beneficial to the population of the BMR, considering trade-offs between the loss of rural landscape in the outer parts and an increase of green space per capita in the inner city (Garcia and Riera 2003).

Currently, the total green space within the municipality of Barcelona amounts to 28.93 km^2, representing 28.59 % of the total municipal area and 17.91 m^2 of green space per inhabitant (Barcelona City Council, Statistical Yearbook 2012). However, most of this green space corresponds to the peri-urban forest of Collserola (Boada et al. 2000). In the core of Barcelona—excluding Collserola forest—green space per capita amounts to no more than 6.80 m^2 per inhabitant, which is a very low ratio in comparison with other European cities (Fuller and Gaston 2009). On the contrary, the number of single street trees—with almost 160,000 units and a ratio of almost 1 tree per 10 inhabitants—is comparatively high (Pauleit et al. 2002).

The urban street trees and the urban forests of Barcelona have been documented to provide a wide range of benefits to the city dwellers by generating a variety of regulating ecosystem services such as urban temperature regulation, noise reduction, and water flow regulation (Table 11.12). Chaparro and Terradas (2009) estimate that urban forests in Barcelona contribute to GHG emission offsets by carbon storage amounting to 113,437 t (11.2 t/ha) and by carbon sequestration amounting to net 5,422 t/year (0.54 t/ha/year). Urban forests also contribute to air purification, an important policy issue in Barcelona due to elevated air pollution levels (Toll and Baldasano 2000; Pérez et al. 2009). Air purification by urban forest, shrubs, and street trees in Barcelona has been estimated in 305.6 t/year, including 166 t/year PM10 removal, 72.6 t/year of O_3, 54.6 t/year of NO_2, 6.8 t/year of SO_2, and 5.6 t/year of CO removal (Chaparro and Terradas 2009). Decreases in air pollution levels can provide considerable health benefits. For example, previous research has suggested that urban vegetation of Barcelona could decrease current PM10 levels from 50 to 20 mg/m^3, thereby increasing the average life expectancy of its inhabitants by 14 months (Pérez et al. 2009).

Table 11.12 Ecosystem services in Barcelona

Ecosystem service	Location of production: local, regional, global	Production unit	Spill over effects	Unit affected by spill over	ES affected	Restoration, transformation potential	References
Provisioning							
Food provisioning	Local	community gardens (Anguelovski 2012), vegetable gardens (Domene and Saurí 2007)	People engage in (outdoor) activities in their communities, human-nature interchange, potential disruption perception of disorder within planned infrastructure (Domene and Saurí 2007)	Individuals, planning entities	Increased sense of place, education, social-ecological memory and social cohesion, potentially reduced aesthetics (Domene and Saurí 2007)	Long term environmental revitalization and neighborhood rehabilitation (Anguelovski 2013)	Domene and Saurí (2007); Anguelovski (2012)
Drinking water supply	Regional	Freshwater wetland, open freshwater, riparian buffer (Brenner et al. 2010)	Conservation/restoration of catchment area	Catchment area	Habitat, aesthetic, and spiritual experiences (Brenner et al. 2010)	.	Brenner et al (2010)

Regulating							
Water flow regulation and runoff mitigation	Local, Regional	Forests and other green spaces (Chaparro and Terradas 2009), urban green space (Brenner et al. 2010)	Potential of rainwater retention and use (Núñez et al. 2010)	Lower parts, areas with high slopes, aquifers	Flood control, erosion control, Drinking Water Supply		Chaparro and Terradas (2009); Brenner et al (2010)
Air purification/ air quality regulation	Local, Regional	Urban Forests (Chaparro and Terradas 2009), urban green space (Brenner et al. 2010)	Decrease of air quality (increase of O_3-levels) due to VOC-emissions (Chaparro and Terradas 2009; Toll and Baldasano 2000)	Individuals, whole city	Aesthetic and recreation (Brenner et al. 2010)	Increased tree cover would lower health risk and mortality, e.g. lowering PM10 levels to 20 mg/m^3 reduces deaths by 3,500 (Pérez et al. 2009), Maximization of species emitting few VOCs to reduce the formation of O_3 and CO (Chaparro and Terradas 2009)	Pérez, Sunyer and Künzli (2009); Chaparro and Terradas (2009); Toll and Baldasano (2000); Brenner et al. (2010)

(continued)

Table 11.12 (continued)

Ecosystem service	Location of production: local, regional, global	Production unit	Spill over effects	Unit affected by spill over	ES affected	Restoration, transformation potential	References
(Global) Climate regulation	Local, Regional	Forests and other green spaces (Chaparro and Terradas 2009)				Expansion of tree cover to increase C-storage, recycling of timber from pruning or dead trees to increase C-storage time (Chaparro and Terradas 2009)	Chaparro and Terradas (2009)

Temperature regulation	Local, Regional	Forests and other green spaces (Chaparro and Terradas 2009)	Reduction of GHG emissions and monetary costs due to lower heating and air-conditioning requirements (Chaparro and Terradas 2009)	Housing, individuals	Sense of well-being, (Global) climate regulation	Reduction of heat island effect which can reach an intensity of up to 8 °C (Moreno-Garcia 1994), Planting of species with high leaf areas and transpiration rates, promotes cooling (only where water sources are given) (Chaparro and Terradas 2009)	Chaparro and Terradas (2009); Moreno-Garcia (1994)
Noise reduction	Local	Forests and other green spaces (Chaparro and Terradas 2009)	Stress reduction	Individuals	Recreation, sense of well-being		Chaparro and Terradas (2009)

(continued)

Table 11.12 (continued)

Ecosystem service	Location of production: local, regional, global	Production unit	Spill over effects	Unit affected by spill over	ES affected	Restoration, transformation potential	References
Cultural							
Amenity and aesthetic	Local	Urban green space (Brenner et al. 2010)					Brenner et al. (2010); Domene and Saurí (2007)
Recreation (physical and mental)	Local	Urban green space (Brenner et al. 2010)					Brenner et al. (2010)
Environmental education and cognitive development	Local	Community gardens (Anguelovski 2012), vegetable gardens (Domene and Saurí 2007)	Enrichment due to "caring activity" (Domene and Saurí 2007)	Individuals, communities, urban parks (Anguelovski 2012)	Food security, knowledge preservation (Domene and Saurí 2007)	Social cohesion potential, decreased health-care costs	(Anguelovski 2012); Domene and Saurí (2007)
Spiritual experience and sense of place	Local						
Sense of well-being	Local	Allotment gardens, vegetable gardens (Domene and Saurí 2007)	People engage in (outdoor) activities in their communities	Individuals, communities	Recreation, sense of place, social cohesion		Domene and Saurí (2007)

Knowledge preservation	Local	community gardens (Anguelovski 2012), vegetable gardens (Domene and Saurí 2007)	Human-nature interactions, unplanned change/ degradation of ecosystems	Riverbanks, brown fields	Food security, cognitive development, habitat loss (Domene and Saurí 2007)	Increased resilience, maintenance of cultural identity by immigrant (Domene and Saurí 2007)	(Anguelovski (2012); Domene and Saurí (2007)
Social cohesion	Local	Community gardens (Anguelovski 2012)	People engage in (outdoor) activities in their communities, integration of marginalized (immigrant) societal groups (Anguelovski 2012)	Surrounding neighborhoods	Recreation, sense of place	Development potential for waste/brown fields	(Anguelovski 2012)

Produced by Langemeyer (2013)

However, the importance of green space for biodiversity and the generation of ecosystem services has only gained stronger recognition in urban policy making recently, as manifested in Barcelona's *Pla del Verd i la Biodiversitat* (Plan of Green Space and Biodiversity), a strategic plan with the goal to enhance Barcelona's ecological infrastructure. Because Barcelona is a highly compact city and available space for the restoration of green space is relatively low, urban planning needs to account for trade-offs between different ecosystem services as favored under different policy and land-use scenarios. The perceived scarcity of available green space in Barcelona and a disregard of the need for specific ecosystem services by urban planning has led to many individual and community-based informal greening initiatives (Domene and Saurí 2007; Arbaci and Tapada-Berteli 2012). An outstanding example is the creation of the "Pou de la Figuera," a green space located in the old town of the city. This area, which was previously intended for the construction of parking spaces and high-end apartments, is today a popular green space created by the initiative of neighbors and environmental activists. It embeds planted areas, sports areas, and a community garden, all of which provide support for a variety of ecosystem services including recreational activities, social cohesion, environmental education, and food production (see Anguelovski 2012).

The provision of cultural ecosystem services is also crucial in urban parks, which have been in the focus of urban planning in Barcelona since the end of the nineteenth century (Roca 2000, p.405). For example, the Park Montjuïc, which—with more 300 ha—is the biggest inner city park in Barcelona, provides a broad range of cultural ecosystem services and receives about 16 million visitors per year (Ajuntament de Barcelona, Modificació del Pla General Metropolità de la Muntanya de Montjuïc 2010). Simultaneously Montjuïc embeds the city's highest levels of biodiversity and serves as habitat for multiple species (Boada et al. 2000). The limited amount of green space in the dense city of Barcelona necessitates a broader knowledge about trade-offs and synergies between the supply of different ecosystem services. It further requires a broader acknowledgement of citizens' needs in the planning of urban green spaces. Waste and brown-fields, even if they are very limited in their extension, have a high potential to provide ecosystem services when used—for example—as community gardens.

11.6 Conclusions

Urbanization and technological progress has fostered the conception of an urban society that is increasingly disconnected and independent from ecosystems. However, demands on natural capital and ecosystem services keep increasing steadily in our urbanized planet (Gómez-Baggethun and de Groot 2010; Guo et al. 2010). Decoupling of cities from ecosystems can only occur locally and partially, thanks to the appropriation of vast areas of ecosystem services provision beyond the city boundaries. Just as any other social-ecological system, cities depend on ecosystems and their components to sustain long-term conditions for life, health, good social relations and other important aspects of human well-being. If taken seriously in urban policy, ecosystem services can play an important role in reconnecting cities to the biosphere (Jansson 2013).

The present review synthesizes research that outlines the potential role of urban ecological infrastructure in enhancing resilience and quality of life in cities. Ecosystem services that can be especially relevant in urban contexts include noise reduction, urban temperature regulation, moderation of climate extremes, outdoor recreation, cognitive development, and social cohesion. Besides their contribution to quality of life, urban ecosystem services can be a major source of resilience for cities, thereby enhancing capacity to deal with environmental and socio-economic shocks. For example, temperature regulation by vegetation reduces health impacts from heat waves, and natural barriers such as mangroves and coral reefs in coastal cities reduce the potential damages from storms and waves. Likewise, urban allotment gardens can improve food security in times of crises.

The importance of urban ecosystem services can be approached from multiple, sometimes conflicting, value perspectives, each of which may capture a relevant dimension of urban environmental policy (Martínez Alier 2002). Ethics and aesthetics, health, environmental justice, economic costs, and resilience are all relevant languages in the valuation of urban ecosystem services. They each emphasize different forms of value that cannot simultaneously maximized or reduced to single measurements. Loss of green space may simultaneously involve health impacts and increased vulnerability to shocks but may (or may not) also provide additional economic benefits. Clearing a patch of forest to create a park enhances recreational values but generally reduces biodiversity. Thus, trade-offs arise not only across ecosystem services but also across the different dimensions of value of those services (Martín-López et al. 2013). Furthermore, specific ecosystem processes and components that may be perceived as services by some, may be perceived as disservices by others. Green areas in cities can be simultaneously perceived by different people as pleasant sites for recreation (Chiesura 2004) or as dangerous places to walk at night (Bixler and Floyd 1997). Likewise, large street trees may be positively seen as providing shade and aesthetic benefits by pedestrians, while people living in the buildings close to them may perceive them as a nuisance because they reduce sunlight and block views out of their windows. Reaching a comprehensive picture of the multiple potential benefits and nuisances of restoring or losing urban ecosystems therefore involves endorsing integrated valuation approaches capable of combining multiple value dimensions, stakeholder perspectives, knowledge systems and fields of expertise.

Framing and achieving a new vision to enhance the sustainability of cities based on the restoration of ecological infrastructure and ecosystem services means moving away from conventional approaches to economics and engineering and towards the application of ideas from broader, more transdisciplinary fields (Costanza et al. 2006a; Lundy and Wade 2011). Although the ecosystem services perspective has led to great progress in our understanding of specific forms of human-nature relations, it should be noted that awareness of the links between urban ecosystems and human well-being is not a novel finding of the ecosystem service approach

(Gómez-Baggethun et al. 2010b). Meteorologists, urban architects, urban planners, urban ecologists, and urban sociologists, among others, also have studied the effects of urban vegetation in cooling, pollutant reduction, noise attenuation, aesthetics, and also the role of green space for human enjoyment and quality of life in cities—though not necessarily under the terms of what we today call urban ecosystem services. An important contribution of the ecosystem service approach has been to provide a framework to integrate information from various fields of knowledge concerned with the urban environment and to facilitate an arena for interdisciplinary dialogue.

Despite mounting evidence of the links between urban ecosystems and quality of life, direct relations of specific ecosystem services to well-being components should not be taken for granted or extrapolated in simplistic ways into urban planning processes. Commonly cited benefits of urban ecosystems are still poorly supported by empirical evidence, and our understanding of their links to well-being is uneven. For example, a recent study by Pataki et al. (2011) found that to date, there is little data showing that urban green space can reduce urban greenhouse-gas emissions or air and water pollutant concentrations but that there is wide evidence supporting substantial reductions in urban runoff and effects in local temperature regulation. These authors also suggest that improvements in human health do not seem to be related in simple ways to absorption of air pollutants by urban forest. The effectiveness of solutions to urban problems based on ecological infrastructure should also be compared against other strategies, and often considered as a complement to them. For example, whereas restoration of urban forests is likely to be an effective measure to enhance biodiversity and opportunities for recreation, caps on car use or taxes on fuels may be a more effective measure to reduce urban greenhouse-gas emissions and to improve air quality in cities.

The same cautionary note holds for over simplistic narratives on the economic benefits of restoring urban ecological infrastructure. Including the economic value of ecosystem services in cost-benefit analysis does not guarantee that solutions based on ecological infrastructure will be cheaper when compared to solutions based on built infrastructure and technology. Moreover, when using the approach of economic values, serious economic analysis should not only take into account benefits from ecosystem services but also costs from ecosystem disservices. Multiple valuation languages come at play in our interaction with urban nature and perspectives relying on single values are unlikely to capture the complexity of ecosystem services. Urban development projects that make economic sense may not be acceptable if they affect important cultural values, human health, habitats for rare species, or if they violate basic principles of environmental justice.

Urban ecosystem services and ecological infrastructure can play a key role in reconnecting cities to the biosphere, restoring local commons, reducing ecological footprints, orchestrating disciplinary fields and stakeholder perspectives, and guiding policies to improve quality of life in cities. Strategies aimed at restoring and enhancing urban ecosystem services should play a major role in the transition towards more healthy, resilient, and sustainable cities.

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Chapter 12
Shrinking Cities, Biodiversity and Ecosystem Services

Dagmar Haase

Abstract Urban shrinkage is a new challenge for both land-use and biodiversity research. Currently, more than 370 cities worldwide, mainly but by no means exclusively in the developed western world, are experiencing population decline. Consequently, visionary urban biodiversity policy has to deal with the opposite phenomenon of growth: processes of de-densification, depletion and land abandonment. This chapter will show that urban shrinkage appears in many different shapes and forms. It addresses the complex relationships between socio-demography, infrastructure, land-use, ecosystem services and biodiversity *in* and *of* shrinking cities. The chapter gives examples of how to ensure both urban quality of life and healthy urban ecosystems under conditions of shrinkage. It places emphasis on how opportunities provided by shrinkage can be used to make cities greener and more diverse, while developing them in greater harmony with nature.

The chapter shows how city governments might face shrinkage in terms of re-thinking their visions, planning strategies and governance. Because shrinkage in cities and the relationship between shrinkage and ecosystems has been rarely discussed in scientific literature thus far, most of the empirical material for this chapter comes from the city of Leipzig, Germany, currently one of the best-investigated shrinking cities in terms of urban ecological processes and patterns. Due to the fact that shrinkage has been documented predominantly in the developed world, both discussion and conclusions of Chap. 12 relate to such types of cities and not to developing world-type megacities, though it may well be that urban shrinkage also needs to be considered as a challenge in the developing world.

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12.1 Introduction

When looking at urban areas and cities, one of the main questions worldwide is how to keep nature intact and environmental resources and biodiversity available for future generations (Seto et al. 2012; Haase 2012, 2013). Over decades, land-use and spatial planning in urban areas has been focused on how to minimize negative effects of settlement, transport and industrial growth, and consequent land consumption, loss and degradation of habitats, and air pollution (Nuissl et al. 2008; Johnson 2001). Set against this context, urban shrinkage and cities with declining population represent a comparatively new challenge. Currently, more than 370 cities worldwide – most of them located in the developed western world – are recorded as shrinking, meaning they are losing population (≥1 % per year or up to 10 % overall). Shrinking cities are predominantly situated in Europe, Russia, Japan and the United States (Rieniets 2009). However, cities in the developing world can also be sites of shrinkage. It should be noted that for countries at war, urban shrinkage is a common outcome. Mogadishu, for example, reputedly lost over a million people because of the Somali conflict (Webersik 2006; Marchal 2006). Harare in Zimbabwe has seen net outmigration due to political and economic instability under President Robert Mugabe (Potts 2009). For regions undergoing rapid urbanization, it is rare for all cities to grow at the same rate and many urban places (especially smaller cities and cities in less affluent parts of a country or region) are likely to experience population decline associated with net outmigration. South Africa, for instance, exhibits this mixed pattern of urban growth and decline even though the country as a whole is experiencing net urban growth and rising rates of urbanization (Boraine et al. 2006). In China, the issue is not shrinking cities, but rather the "hollowing out" of villages because of urbanization. There are hundreds of examples of villages and rural communities in China that show small villages lack development planning and are emptying out as villagers move to new cities for jobs (Long et al. 2012). Because for most developing countries the overarching picture is one of urban growth (Chap. 21), the issue of the shrinkage of selected cities, and especially how this might impact the ecological integrity of the urban system, has received disproportionately little attention. The rapid demographic transition anticipated in China and in middle-income countries suggest, however, that this lacuna will require urgent attention. In this regard, the experiences of managing biodiversity in Europe's shrinking cities is of considerable wider interest.

In order to address shrinkage, visionary urban policy has to deal with population aging, and accompanying processes of under-use of buildings and infrastructure, de-densification, and an increase in derelict land and brownfields as a consequence of land abandonment. This presents new questions for the urban policy agenda such as: How can land-use development be directed in order to ensure both urban quality of life and urban healthy ecosystems under conditions of shrinkage? How can high-quality and sustainable urban livelihoods be maintained in a shrinking city? Are there new opportunities provided by processes and patterns of shrinkage that can make our cities greener and more diverse? Is shrinkage a way to reduce the urban footprint? Until now, urban policy making and planning has mostly been concerned with directing urban growth; there are no prescriptions for how to comprehensively develop or plan a shrinking city. This chapter addresses the complex relationships between socio-demography, infrastructure, land-use patterns and ecosystems in shrinking cities. It further discusses the consequences of shrinkage in terms of sustainability, ecosystem services, urban footprint and biodiversity. The chapter focuses on empirical and model data elaborated for two shrinking cities in Eastern Germany: Leipzig and Halle. They have been undergoing the process of shrinkage for more than 30 years and therefore serve as rich case examples from which to study patterns of shrinkage and its socio-environmental and ecological consequences.

12.2 What Is Urban Shrinkage?

Referencing Haase et al. (2012), Rink (2009), urban shrinkage is defined as a phenomenon of massive population loss in cities that results from a specific interplay of the economic, financial, demographic or settlement systems, environmental hazards, and changes in political or administrative systems. A prominent example of these shifts were the systemic changes that occurred in Germany and Eastern Europe after 1990 and were coupled with the introduction of a market economy (Rink et al. 2009; Moss 2008). Urban shrinkage has been examined through the lens of uneven economic development (Harvey 2006) and the underlying dynamics of the territorial division of labor (Amin and Thrift 1994). Shrinkage might also result from tremendous environmental disasters, such as Hurricane Katrina, which devastated the city of New Orleans in 2005. Another reason for urban shrinkage is demographic change, namely low fertility and massive out-migration (Müller 2004). The current processes determining urban shrinkage in Eastern Germany – specifically important for the case studies of Leipzig and Halle – emerge in the form of the post-transition decline of traditional heavy industries, a decline that induces general economic crises, unemployment, out-migration to other prospering regions and a subsequent decline in fertility and an increase in population aging (Haase et al. 2012). Furthermore, rampant suburbanization in the peri-urban zones around shrinking cities leads to residents abandoning the city. Both processes often rapidly cause an increase in the age of the remaining population as the elderly stay in the city and their relative percentage increases. This, as kind of a vicious cycle, results in further demographic decline (Nuissl and Rink 2005; Kabisch et al. 2006). Such development was found in many shrinking cities across Europe and the U.S. (Couch et al. 2005). In any case, shrinkage is a socio-economic process but also refers to spatial and land-use patterns (Berg et al. 1982; Lever 1993; Garreau 1991).

12.3 International Relevance and Prospects of Urban Shrinkage

In the last 50 years, about 370 cities with more than 100,000 residents have undergone population losses of more than 10 % (Fig. 12.1). They are distributed across the globe, predominantly in its early-industrialized and developed part, though the

Fig. 12.1 Shrinking cities faced with population losses worldwide (Reproduced from Oswalt and Rieniets 2006. Published with kind permission of © Hatje Cantz Publishers 2006. All Rights Reserved)

lack of data in poorer countries almost certainly masks an undercount of urban shrinkage, just as it precludes the precise tracking of urban growth. In Europe, there are currently more than 70 shrinking cities (Kabisch et al. 2012), and 92 of the worldwide recorded depopulating cities are located in the United States. However, urban shrinkage is already or will be on the political agenda in countries like Japan, Russia or even China.

Population decline not only impacts business and employment in the city it also carries repercussions for housing, social and technical infrastructure, municipal finances, social cohesion, segregation and other aspects of urban life (Oswalt and Rieniets 2006; Großmann et al. 2008; Haase et al. 2010; Lauf et al. 2012a, 2012b). Urban shrinkage results in a mismatched supply and demand of space and infrastructure: more space is available for fewer inhabitants. It also reduces the tax base. In this regard, urban shrinkage results in a reconfiguration or reshaping of urban land-use patterns. On the one hand, shrinkage leads to vacancies and derelict land in the affected neighborhoods; on the other, it permits a redistribution of households according to their housing preferences because of low housing costs in favored inner-urban locations and a high number of affordable apartments and houses. Clearly then, shrinkage can greatly affect the quality of urban life (Haase 2008) and support urban resurgence (Kabisch et al. 2009).

Urban shrinkage reshapes social settings for a variety of actors in the city: residents, planners, policy makers, entrepreneurs, and service suppliers (Haase et al. 2007; Jessen 2006). It is difficult to steer or govern urban shrinkage, because under

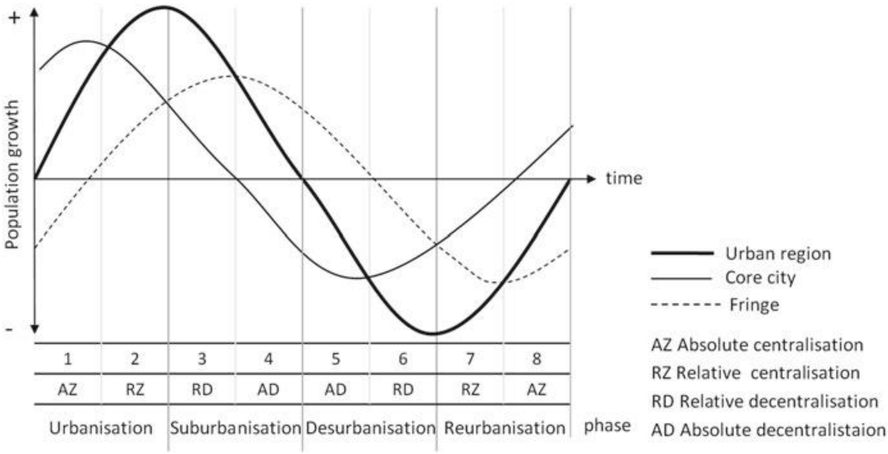

Fig. 12.2 Model of the stages of urban development (Reproduced from Berg et al. 1982; Kabisch and Haase 2011. Published with kind permission of © Population, Space and Place 2011. All Rights Reserved)

the conditions it produces, governance arrangements risk becoming unstable and fragmented due to a high dependency on external funding for initiatives and activities to address shrinkage, a funding-dependent restriction on initiatives, and unstable coalitions among weak actors (Couch et al. 2011). Several comparative studies of European urban development trends exist for the second half of the twentieth century. One of them has been carried out by van den Berg et al. (1982) in the early 1980s. Based on their results over the period 1950–1975, they developed a four-stage sequential model of urban development that was consistent with the urbanization process in Europe from the early nineteenth century onwards (Fig. 12.2). Using the proxy of population growth of an urban region, four main stages were outlined: urbanization, suburbanization, desurbanization, and, more hypothetically, reurbanization (Kabisch and Haase 2011). Suburbanization processes lead to a decline in the core city and to a desurbanization phase, in which population decline appears everywhere in the core city and the fringe area, and finally leads to negative population growth rates in the entire urban region. This stage is characterized by a dispersal of activities to rural areas and satellite towns, or simply a total decline of activities in the entire urban region. Although Berg et al. (1982) did not refer at all to urban shrinkage, the above described phase of desurbanization helps to theoretically sort the phenomenon of urban shrinkage into the phases of urban development.

To provide some visual examples of what shrinking cities may look like, Fig. 12.3 shows some selected typical land-use patterns of urban shrinkage using examples from Eastern Germany, Poland and the United States. Here, under-use and vacancy of built houses and infrastructure eventually leads to a patchwork-pattern demolition of buildings and the creation of derelict land and brownfields (Lorance Rall and Haase 2011), and an emergence of what Lütke-Daldrup (2001) described as "land-use perforation" (cf. Sect. 12.4.2).

Fig. 12.3 Typical features of urban shrinkage affecting built space: under-use and vacancy of built infrastructure, demolition of buildings, creation of residential and commercial brownfields, and emergence of land-use perforation. Examples are shown from cities in Eastern Germany (*upper photos*), Poland (*lower left*) and the United States (*lower right*) (Photographs published with kind permission of © Google Earth 2013. All Rights Reserved)

12.4 Processes and Patterns of Urban Shrinkage

12.4.1 Demographic Change and Aging

Currently, many cities across Europe are undergoing demographic change (Kabisch and Haase 2011; theoretical background see Cloet 2003; Lutz 2001). Demographic change is becoming increasingly important in discussions about planning policy and governance, as it is considered to be an important factor for future land use development and urbanization throughout the whole of Europe's cities and urban regions (UN 2007; UNPF 2007). In a range of European countries such as Germany, Italy, Greece, Poland, Portugal, Russia, and Spain, the demographic development is characterized by a predominantly declining and aging population. This demographic trend is due to a fertility that is below replacement and an increasing life expectancy (Edmonston 2006). Another important aspect of demographic change is the decline of the average household size in line with what demographers describe as the "Second Demographic Transition" (Lesthaeghe and Neels 2002; Steinführer and Haase 2007). In addition to a decrease in fertility, migration can be an even stronger determining factor that influences population size and age structure. Shrinking cities particularly suffer from population decline and exhibit a growing proportion of old age and retired people, whereas growing urban centers are still experiencing in-migration by younger age classes and thus an increase in fertility (Kabisch and Haase 2011). Shrinking cities report low fertility rates (predominantly lower than 1.5; Urban Audit), high old age-dependency ratios (ratio of the population aged 65 years or over to the population aged 20–64), a comparatively high share of very old persons (>80 years old), and a declining total annual population over several years or decades (Kabisch et al. 2012).

12.4.2 Land-Use and Infrastructure

The impacts of shrinkage on urban land-use are complex because shrinkage affects both urban fabric and open space in an uneven manner (Haase and Schwarz 2009, 2012). Cities in the U.S., for example, experience the so-called doughnut effect, which is created when city centers become hollows consisting of brownfields and unused plots, and the suburbs grow (Beauregard 2009). In Eastern Germany, urban shrinkage has led to what has been called perforation of the urban fabric wherein specific parts of the city face a more drastic demolition, the appearance of derelict land, and thus an alteration of the built space (Fig. 12.4) (Haase and Schwarz 2009, 2012; Haase et al. 2012; Schwarz et al. 2010). In Eastern Europe, despite the emergence of brownfields as a result of de-industrialization, such dramatic land-use changes still are not yet observed. The impacts of shrinkage on land-use can lag; for instance, it takes time until a vacant building is demolished or a new land-use is finally established. Often, various kinds of interim land-uses can be observed, which represent the subtler processes of land-use change (Lorance Rall and Haase 2011). However, urban shrinkage should not only be associated with losses and negative connotations; it also creates new spaces along with affordable land available for alternative land use options such as public or green spaces (Haase 2008).

As outlined above, population decline in shrinking cities leads to a decrease in residential density and to both oversupply and underuse of urban land, namely housing stock, infrastructure and services (Haase et al. 2007). This creates problems for both public and private suppliers with respect to the underuse of the building stock, primarily dense urban fabric, industrial buildings, and storage depots. Underuse, in turn, leads to housing and commercial vacancies and to a more rapid dilapidation of unused buildings (Bernt 2009). While in some places buildings are demolished to balance the housing or real estate market (Couch et al. 2005), in others they simply become unusable after a period of disuse. While a decreasing building stock density may lead to a relaxation in a densely built city, at a later stage such a decrease might, because of the vacant lots, lead to a perforation of the urban space in the form of a dissolution of the street or block structure (Haase et al. 2007). Moreover, ongoing construction activities in the peri-urban areas of shrinking cities reinforce this decline of the inner city (Nuissl and Rink 2005; Haase et al. 2012).

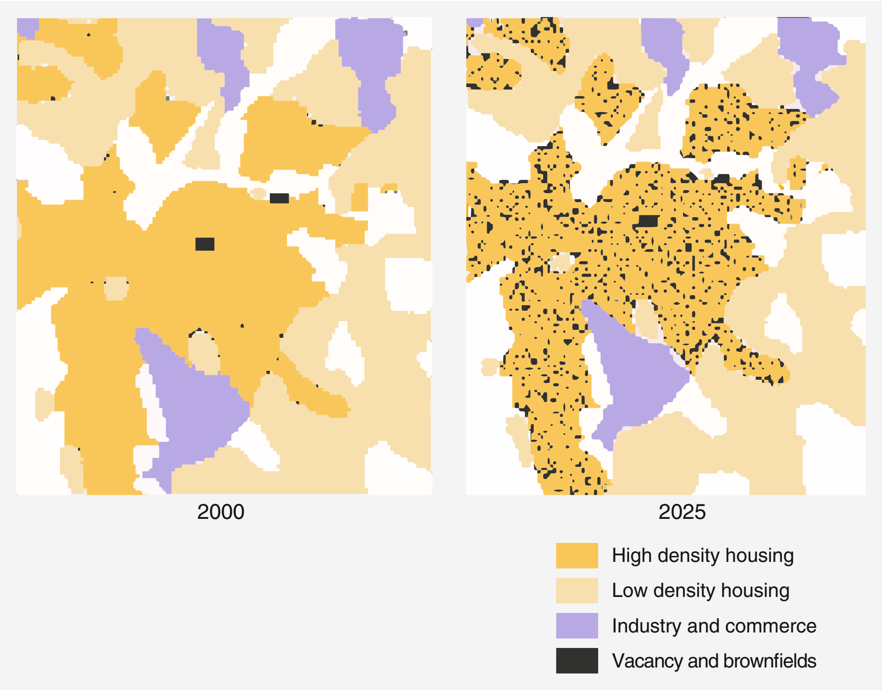

Fig. 12.4 Simulation of potential land-use perforation in the inner parts of the city of Leipzig 2000–2025 (Modified from Haase et al. 2012. Published with kind permission of © Environmental Modelling and Software 2012. All Rights Reserved)

This uneven land-use development of inner-city and peri-urban space can be found in almost all shrinking cities across the world.

Land-use perforation itself poses challenges for superficial and subterranean urban infrastructure provision. This is obvious for network-dependent infrastructure, like water, sewage or electricity: vacant houses and derelict land no longer need supply of water or electricity or a transport for waste water, so the pipes and cables leading to this house are no longer used. In an area with a larger proportion of vacancy and derelict land, under-utilization can pose severe problems for maintenance of the service for the whole area (Moss 2008). Social urban infrastructure like schools, daycare centers, roads and public transport are also influenced by vacancy. All of these infrastructures are optimized for a certain demand structure in an area, usually determined by population density and commercial or industrial activity. In the best-case scenario, efficiency decreases in areas with higher rates of vacancy (Blanco et al. 2009; Schiller and Siedentop 2005). In the worst-case scenario, an area might enter a vicious cycle of declining population, under-utilized and then dismantled infrastructure, so that the area becomes less attractive. Thus, even more residents relocate to another area in the city (Fig. 12.5) (Schwarz and Haase 2010).

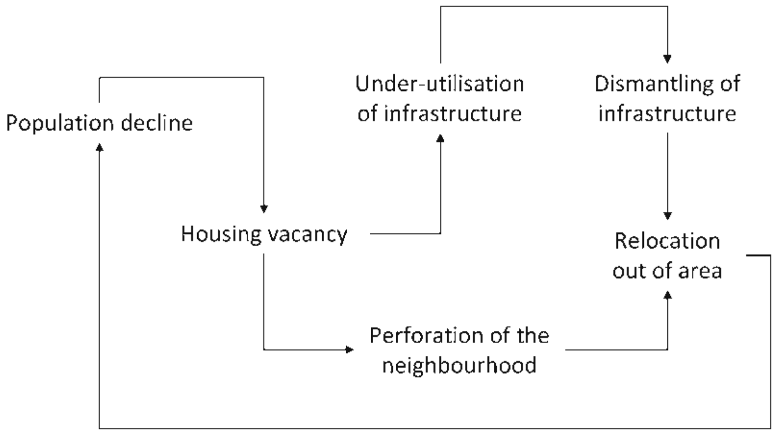

Fig. 12.5 Infrastructure-related problems for a shrinking city (Reproduced from Schwarz and Haase 2010. Published with kind permission of © International Environmental Modelling and Software Society International Congress 2010. All Rights Reserved)

12.4.3 Green Space, Habitats and Biodiversity

Despite or even because of their demographic decline, shrinking cities may provide a desired quality of life for all different age classes, including young, middle-age, and elderly dwellers (Haase 2008). Shrinkage may provide opportunity for the creation of new green surroundings and recreational facilities that play a key role in urban quality of life and influence the migration behavior/balance (Schetke et al. 2010). Access and functionality of open/green space is essential to healthy work-life-balance, as well to a healthy aging. The presence of open and green space (which emerges under the conditions that shrinking cities provide) is crucial to a positive evaluation of one's neighborhood (Sugiyama et al. 2009) and to the accomplishment of healthy activities, such as walking to nearby open spaces (Saelens et al. 2003). Furthermore, the amount of green space close to where people live has a significant impact on their perceived health (Schetke et al. 2012) and can restore ability to focus (Kaplan and Kaplan 1989). Nearby green spaces offer opportunities for private and municipal gardening, food production (Kremer and DeLiberty 2011) and leisure activities (Lorance Rall and Haase 2011; Stigsdotter and Grahn 2004) Additionally, a fair distribution of open and green spaces helps to redress social inequalities by providing different groups of people the opportunity to use and be exposed to these settings (Kuo et al. 1998).

In shrinking cities, a substantial number of vacant lots are found in housing estates, and commercial vacancies occur within inner-city shopping malls or in the form of large-scale brownfield land. Thus, shrinkage has consequences for building and population densities, number and form of green spaces as well as the percentage of impervious surface (Sander 2006; Schetke and Haase 2008).

In Leipzig, Mehnert et al. (2005) found a positive correlation between the total amount of urban green infrastructure (parks, allotments, cemeteries, forest, etc.) and the suitability of habitat for breeding birds (e.g., for the green woodpecker, *Picus viridus*). Strauss and Biedermann (2006) reported an increase in species richness in cities with an increase in inner-city grassy brownfields. Such open or wasteland patches are niches in which rare species thrive (Bolund and Hunhammar 1999; Shochart et al. 2006). For a more extensive discussion on patterns and trends of urban biodiversity, see Chap. 10. Concluding from these empirical studies, one can expect a positive impact of land-use perforation in shrinking cities on biodiversity in terms of how derelict and vacant land become both resource and habitat. Particularly in dense residential districts of a shrinking city, a comparatively high number of residents will benefit from an increasing biodiversity on vacant land.

In order to measure how land abandonment affects biodiversity, several recently demolished sites allocated in high-density, nineteenth-century and socialist-era prefabricated housing estates in Leipzig were analyzed in terms of their spatial shape, configuration and the resulting habitat quality for the Whitethroat (*Sylvia communis*), a bird indicator species of urban land quality. The pre- and post-demolition situations at 50 sites were compared using the following well-documented landscape metrics indices (Uuemaa et al. 2009; Walz 2011): Largest Patch Index (LPI) of open land uses, Edge Density (ED), Habitat Suitability Index (HSI) and Shannon Diversity Index (SHDI). The HSI was calculated using the approach of the ecological niche, which is formalized by the sum of cells with a certain probability of species presence (cf. Mehnert et al. 2005). For calculation and mapping purpose, the Biomapper software tool was used (cf. Hirzel et al. 2002). The results of the study are shown in Fig. 12.6. ED and patch size (LPI) are the variables that most benefit from selective block demolition compared to only slight changes in Shannon diversity due to the uniform grasslands that emerged after demolition at most of the vacant plots. For species such as the Whitethroat, demolition seems to offer an increase in its preferred open habitat conditions. At a superior spatial level, the perforated urban landscape of shrinking cities possesses a higher share of large green, brown, and derelict land uses than densely built-up cities and thus present opportunities for biodiversity enhancement through the deliberate management of vacant land.

From an ecologist's point of view, urban land-use perforation results in structural diversity of urban land-uses and an increase in the amount of edges. Concepts for the redesign of this urban land-use perforation have been discussed in a preliminary form, such as division of the remaining urban core into either equitable sub-centers or a polycentric structure with fewer dense or even empty patches. Lütke-Daldrup (2001) discusses the perforated built-up body as the most probable urban development pathway for shrinking cities of the West. From a broader perspective, however, effects of land-use perforation on urban ecosystems and biodiversity have not yet been statistically verified through many empirical studies (Haase and Schetke 2010).

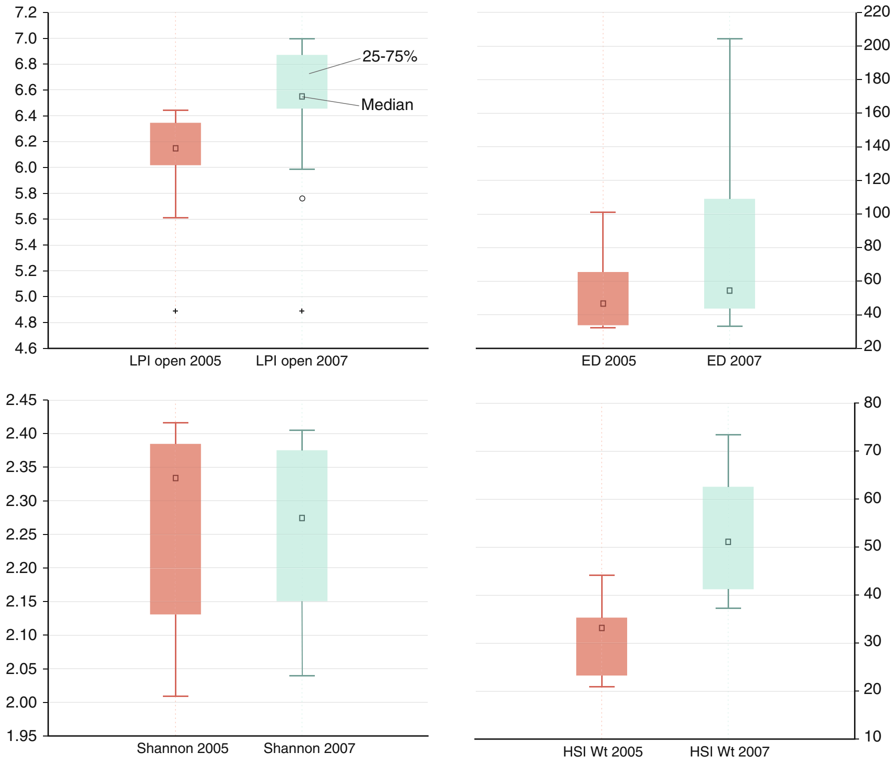

Fig. 12.6 Largest Patch Index (*LPI* open) of open land uses (such as park, allotment, courtyard, brownfield, waste land etc.), Edge Density (*ED*), Shannon Diversity, and Habitat Suitability Index (*HSI*) for a range of recently demolished sites in Leipzig – a comparison of the pre- and post-demolition status (2005 and 2007, respectively) (Modified from Haase and Schetke 2010. Published with kind permission of © Wiley-Blackwell 2010. All Rights Reserved)

12.4.4 Urban Ecosystem Services

There is a relationship between urban shrinkage, vacant land and urban ecosystem services which holds opportunities and new challenges for urban land-use development and related policy-making. Urban shrinkage leads – as reported – to changes in urban land-use and land cover and, consequently, in impervious cover (Haase et al. 2007). Quantity and quality of ecosystem services provisioning, green infrastructure and biodiversity in cities depend on the same variable, the impervious cover (Haase and Nuissl 2010). Over the past years, new empirical knowledge, including remotely sensed and field data about urban ecosystems in shrinking cities, was collected by teams in Eastern Germany (Breuste et al. 2013; Endlicher et al. 2011; Langner and Endlicher 2007) and in the Rust Belt region of the U.S. (Burkholder 2012). These studies have helped to provide a better picture of connections between shrinkage and ecosystem services in cities.

Given the vast amounts of vacant land, what is the role of vacant land in the formation of ecosystem functionality in cities? Significant opportunity lies in the establishment and provision of ecosystem services through strategic design and management of large vacant sites in order to create a web of sustainable land-uses (Burkholder 2012). Vacant built land (with houses), vacant sealed land (without houses), and unused or reused open lands have the potential to enlarge the space/area of ecosystem services provisioning and thus can contribute positively to urban residents' quality of life. Urban ecosystems provide fresh air, air temperature cooling, and stormwater regulation. The relationship shown in the diagram in Fig. 12.7 implies that it is important to know how much of shrinkage-related new land use is available, where it is available, and how urban ecosystem services can be enhanced there. Both field research and modeling are needed to address these issues. For an extended discussion of urban ecosystem services, see Chap. 11.

There are some empirical studies that suggest that land-use patterns of shrinking cities offer opportunities for a re-development of urban nature. European shrinking cities provide great potential for ecosystem services provisioning; for example, modeling results shown in Fig. 12.7 for the city of Leipzig report that there are synergies to be found between the services of recreational usability and new habitat qualities on one hand, and the conditions of perforated land patterns and new edges in the urban space on the other. Moreover, afforestation areas in former built-up areas possess considerable potential for carbon storage (Strohbach and Haase 2012; Strohbach et al. 2011, 2012) and, perhaps even more importantly, for air cooling by tree shade. Leipzig is a highly representative site for a shrinking city in Europe and thus the results obtained here are meaningful also for other European cities. However, it is important to note that urban land-uses providing ecosystem services are predominantly cultivated and managed sites, meaning that urban ecosystem services provisioning in general differs from that offered by natural ecosystems, as there is often a need for additional input by people (for example, energy and maintenance) in urban ecosystems.

12.4.5 Resource and Land Consumption

Resource consumption is a particular concern in cities due to the high concentration of population consuming environmental goods and ecosystem services. The consumption-emission-balance between the city and the rural surroundings is—in general and regardless whether or not a city shrinks—not easy to establish. To give one example: Whereas direct carbon emissions of rural inhabitants are often higher than those of urban ones; the indirect carbon emissions of urban inhabitants are higher compared to rural inhabitants. Some of the factors influencing this phenomenon are differences in diet structure and composition (Kennedy et al. 2009). Comparing urban and a rural areas, urban areas exhibit lower per capita carbon emissions because of the higher population density associated with cities. Nonetheless, in cities, the total carbon emissions increase with income and a more complex life style so that

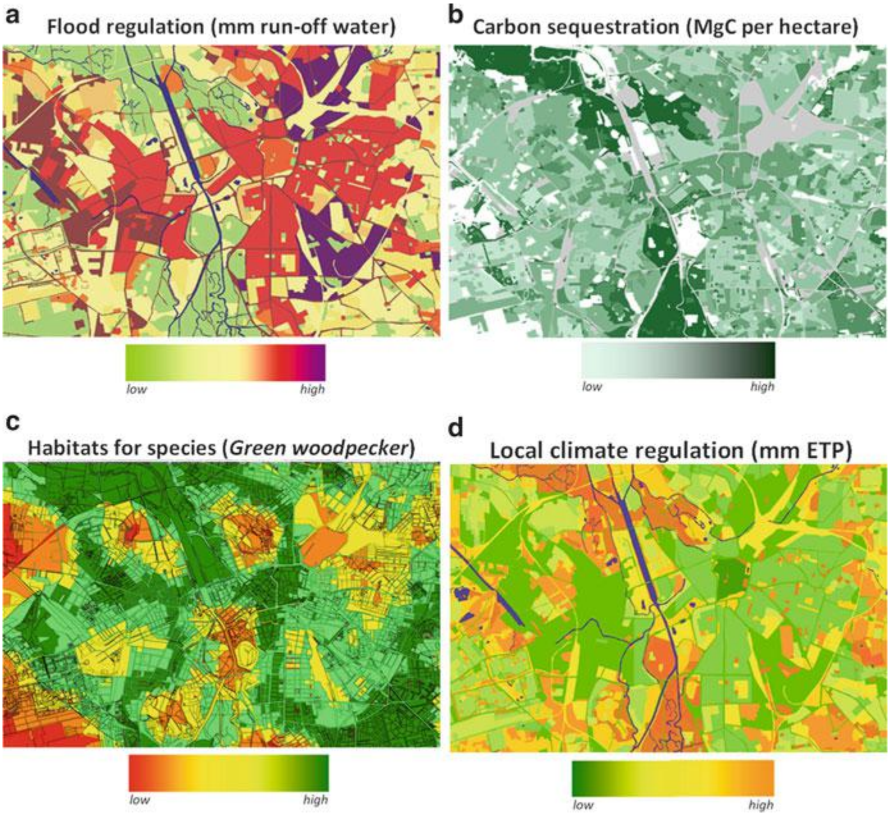

Fig. 12.7 Modeling and mapping urban ecosystem services using four core services in the shrinking city of Leipzig, Germany. (**a**) Impervious (*red color*) land cover hinders rainfall infiltration and flood regulation which is provided by open or vegetated soils (*green*). Open spaces in the built space of the inner city show potential to regulate heavy rainfall and, simultaneously, provide lots of local climate regulation (Fig. 12.7d; Haase 2009). (**b**) Urban forests and trees are able to sequester and to store CO_2 in form of C in their roots and thus help to remove it from the atmosphere and to lower the carbon footprint of a city. Land-use perforation in any built-up structure which is followed by tree growth helps to improve carbon sequestration (Strohbach and Haase 2012). (**c**) Urban structure type with heterogeneous built patterns, nineteenth century-type buildings and bordering to floodplain forests (*green*) provide optimal habitats for urban species such as the green woodpecker (Mehnert et al. 2005). (**d**) Urban green spaces such as forests or parks provide local climate regulation and cooling by tree shading and evapotranspiration flows (*orange*). Impervious surface (*green*), contrariwise, leads to surface runoff and impedes climate regulation (Haase 2009) (Reproduced from Haase 2012. Published with kind permission of © UGEC Viewpoints 2012. All Rights Reserved)

urbanization brings about an increase in carbon release compared to the savings in per capita values (Kennedy et al. 2009). However, shrinkage and declining population numbers do not result in an automatic decrease of natural resource consumption such as land, energy, or water, because the per capita requirements on environmental resources, ecosystem services and housing space are increasing overall (Haase 2012).

In both growing and shrinking cities, the number of households increases due to the demographic transition towards smaller one- and two-person households (Liu et al. 2003). Consequently, the demographic transition towards smaller households in shrinking cities, coupled with rising per capita housing space and resource use, might lead to further land consumption, enlargement of transport infrastructure, and ecosystem decline.

In a recent study on population and land use development in Germany, Kroll and Haase (2010) found that neither a decreasing nor an aging population imply reduced land consumption for housing and transportation. The per capita living space increased by a factor of 1.5 from 1990 to 2006, and one/two-person households' per capita living space increased faster compared to four-person households by a factor of 2.5 in 20 years (Gans and Schmitz-Veltin 2006). Thus, the trend towards smaller household sizes acts as an "invisible" driving force behind ongoing land consumption in shrinking cities. Moreover, in some shrinking cities, sprawling settlement development still continues because of specific housing preferences, such as single-family houses and spacious housing with backdoor gardens. Thus, shrinking cities are in many cases characterised by a vacant or emptying core and a more prosperous fringe (Haase and Nuissl 2010).

12.4.6 Urban Footprint, Sustainability and Resilience

How can we evaluate the impact of shrinkage on urban sustainability? This section suggests using the following two measures and indicators: urban footprint, and the integration of different dimensions of sustainability and resilience. In order to estimate the urban footprint of urban shrinkage, carbon emission and storage at a response unit of 1 ha – a typical size of an area that undergoes change in shrinking cities – is compared for different land covers, among them different land-use types which are typical for shrinking cities as explained in Sect. 12.4.2 (Table 12.1): Although it becomes clear that afforestation at demolished housing or commercial sites cannot compete with carbon storage values of a tropical rainforest, Table 12.1 shows that urban afforestation can help to sequester CO_2 from the atmosphere. It further shows that dense housing with backyards and trees can store a comparatively high amount of carbon. If such old, multi-story houses with backyards and trees are maintained and this land use enlarged, this could lower the per capita carbon footprint of a city. For example, each resident of Leipzig emits 1.8 tons of CO_2 per year (which equals 480 kg C) and thus 1 ha of afforested brownfield can balance the annual CO_2 emissions of about eight residents of the shrinking city of Leipzig. Since there are 7,000 ha of brownfields in Leipzig at the moment, afforestation could balance the annual emissions of 56,000 dwellers, which comprises more than 10 % of the total population.

In order to calculate trade-offs and synergies for sustainability as a consequence of land-use perforation caused by shrinkage, the indicators discussed above in Sect. 12.4 were integrated using the FLAG model computed by SAMISOFT

Table 12.1 Carbon storage in the shrinking city of Leipzig exemplified at typical urban structures/response units and compared to the same area of tropical rainforest

	Sequestration (Mg C ha^{-1})	References
Floodplain forest	98.31	Strohbach and Haase (2012)
High density housing with backyards and old tree vegetation	13.70	Strohbach and Haase (2012)
Low density single family homes with small lawn-type gardens	4.20	Strohbach and Haase (2012)
Afforested brownfield	4.02	Strohbach et al. (2012)
Lawn-covered brownfield	–	Strohbach et al. (2012)
Arable land	–	Strohbach and Haase (2012)
Tropical rainforest	303	Lü et al. (2010)

(Nijkamp and Ouwersloot 2003). FLAG evaluates different land-use states (e.g., the land-use state prior to and after demolition and land abandonment, in relation to predefined standards). It uses critical threshold values (CTVs) (Leeuwen et al. 2003) derived from scientific literature and/or individual urban development targets, such as environmental quality standards (Schetke and Haase 2008). Within the FLAG approach, calculated indicator values are set against the background of standard minimum values (CTV_{min}), target values (CTV) and maximum values (CTV_{max}). Besides the determination of threshold values for quantitative indicators, the integration of qualitative indicators is realized by indicating 0, 1, and 2 as upper and lower threshold values. Uncertainties due to the indicator estimation or calculation are acknowledged by the FLAG system since it defines a validity space and not a concrete value that has to be matched. Using an intensely shrinking and perforated area in the eastern part of the city of Leipzig, one can bring environmental and social components together. Figure 12.8 shows a differentiated picture of what results from demolition and site clearance: generally, an increase in the number of green bars of the ecological indicators (indicated as "biophysical") could be detected, which indicates the positive impacts of land-use perforation. The number of open (and temporary/interim) green spaces increases. In contrast, the black bars remain, which means that perforation does not influence those areas that represent the worst environmental situations (Schetke and Haase 2008; Haase and Schetke 2008).

Urban shrinkage allows us to contemplate a resurgence of nature into inner urban areas that are densely populated and have long been part of the built environment. In this vein, ideas regarding "urban wilderness" for recreational and educational purposes are of interest to planners and landscape architects who are faced with urban shrinkage or decline (Rink 2005). The shrinking city of Leipzig has made the novel suggestion of creating urban greenery in the form of (1) temporary gardens or interim use agreements (Lorance Rall and Haase 2011) at core city demolition sites (as a kind of planned alternative to sites that would otherwise remain vacant), and (2) spontaneous vegetation on former brownfields (as a kind of unplanned alternative). De Sousa (2003) perceives green sites developed from inner-urban brownfield sites as "flagships" or experimental fields that serve as models for the future provision of green space with the objectives of improving local biodiversity and human

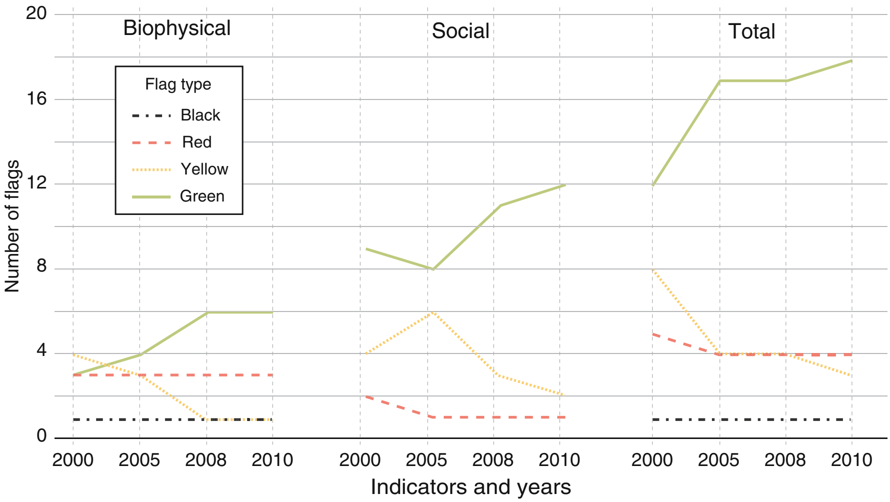

Fig. 12.8 FLAG model result for eastern Leipzig – a comparison of the land-use states of 2000, 2005, 2008, and 2010 under conditions of shrinkage (vacancy, demolition). There are three dimensions of sustainability shown: the ecological, the social and the socio-ecological (= total). The *colors* of the columns indicate: *black* = there is a need to stop the development immediately to avoid further damage; *red* = reverse development needed in terms of a turn into a more socially and environmentally sustainable development; *yellow* = target value in terms of a negative social and ecological development is reached but not exceeded; *green* = acceptable development and no change needed. Overall, the FLAG colors show the direction of development which should be followed, but do not indicate if an area will change or not (Modified from Schetke and Haase 2008. Published with kind permission of © Environmental Impact Assessment Review 2008. All Rights Reserved) (Color figure online)

lifestyles. Shrinkage also results from demolition of multi-story housing stock, which forms a transition towards more spacious housing and living conditions in densely urbanized environments. Larger apartments with integrated patios and terraces that contain vegetation, as well as higher shares of urban green within the neighborhood are emerging (Haase 2008). Of course, this does not mean that shrinking cities are more sustainable and resilient than growing ones, but that they bear great potential to develop into resilient cities by following paths of smart urban growth and/or decline.

12.5 The Policy Nexus: Re-thinking and Governing Shrinkage

How do cities respond to shrinkage in terms of governance, policies and institutions? Are there ways to “live with shrinkage” and options to return to a more stable development? Bernt et al. (2012) list a variety of responses to shrinkage. Overall, these types of governance responses vary along two spectra between (a) policy responses that range from passively acknowledging but neglecting shrinkage conditions in favor of growth-orientated strategies, to actively mediating and using the benefits of shrinkage – that is, more space and more green – for urban quality of life development (adaptation); (b) policy responses that range from focusing investment in areas of decline (typically inner urban areas and peripheral social housing estates) versus in areas with the best growth potential (typically suburban and urban fringe areas) (Verwest 2011). In terms of regional specifics, Bernt et al. (2012) distinguish between (1) 'western' holistic explicit growth or stabilization strategies dealing implicitly with consequences of shrinkage, and (2) 'post-socialist' pro-growth strategies emphasizing job-creation based on attraction of inward investment and European funding, rather than considering causes and consequences of shrinkage. Whereas (1) tries to get at the root and the reasons causing shrinkage and creatively address its effects, (2) attempts to combat shrinkage without making an effort to understand the reasons behind the shrinkage or using its potential.

Both strategies are exogenous – based mainly on external resources. Of course, these external resources are combined with local knowledge. Generally, the development strategies of shrinking city regions in western democratic countries with market economies are a mix between growth strategies supporting economic development – especially in the field of service economy – combined with strategies dealing with the social and physical consequences of shrinkage. These strategies are integrative and holistic because they are not limited only to the support of economic and business development, but also deal with urban regeneration, re-usage of brownfields, investment in the living environment with respect to urban nature and the promotion of social cohesion, and generally appear to be closely aligned with notions of sustainable development. There are, of course, several synergies and trade-offs between land-use policies and governance in shrinking cities and the provisioning of urban ecosystem services and biodiversity. Consider the example of brownfield reuse. One possible strategy, a rebuilding of a brownfield with dense housing, will interfere with services such as climate regulation or the moderation of hazardous rainfall events through absorption of stormwater. A second strategy, terraced houses or urban villas, would automatically give more space to surrounding nature and backyards. A third strategy, an afforested or park-like reused brownfield, might provide access to green space for even more people. Afforestation or trees in a new park would offer additional potential of carbon sequestration. In addition to these described examples, there are many more strategies for possible brownfield reuse. Lorance Rall and Haase (2011) describe one German strategy of governing interim use sites: this program allows individuals or companies to take over the development of private brownfields and waives property taxes in return for a promise of regular maintenance. Thus, the cities can vastly increase public green space in these neighborhoods by following a participatory approach. Finally, yet another approach, urban "guerrilla gardening," the illegal adoption and maintenance of unused areas by residents for a short time, represents an interim form of brownfield reuse in shrinking cities. For an additional discussion on the implications of urban governance of biodiversity and ecosystem services, refer to Chap. 27.

12.6 Conclusions

There are a growing number of shrinking cities around the globe. Due to demographic change, more cities will experience population decline in the near future; this includes countries experiencing rapid overall population growth, such as China and India. Shrinkage is even evident in Africa, which is otherwise thought of as a continent of rapid urban growth. As the urban population increasingly ages, eventually megacities will also face urban shrinkage. Shrinkage can be identified by a number of typical land-use patterns and features such as residential and commercial vacancies, under-use of infrastructure, demolition and brownfields. On the one hand, these effects of shrinkage are problematic, as they might negatively affect social cohesion and attractiveness of living in various neighborhoods in cities. On the other – with particular focus on urban ecosystems and biodiversity – urban shrinkage offers great potential for nature conservation and green space development, and thus also might contribute to urban quality of life. Brownfields hold space for biological succession, habitat development, and cultivation of trees and plants. There are many different ways that cities across the world can cope with shrinkage. Cities in Europe, for example, follow different strategies to respond to population decline and under-use of space. Ultimately, urban shrinkage does not contradict the idea of the compact city; it simply requires new and alternative ways to define and to shape prevailing urban structures and spaces to keep them alive.

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Chapter 13
Regional Assessment of Europe

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In many of the areas presently occupied by European cities, settlements were formed already in Neolithic times, when the continent was colonized by agriculturalists (9500 B.C. onwards). The re-colonization of European plants and animals after the last Ice Age, which covered large areas of Europe, was not completed before human influence began to cause local disturbances, meaning that the native biodiversity has evolved under human influence. The long history of urban development in Europe, and the location of cities in fertile river valleys, are at least two reasons of why many European cities are often characterized by higher species richness of plants and animals than some of the surrounding rural areas. The long history of co-evolution may be a particular factor explaining why European plants and animals worldwide tend to successfully establish in areas with dense human population.

Europe is today one of the world's most urbanised regions, with approximately 75 % of the population living in urban areas; a figure that is expected to increase to 90 % in 2100. Over the past 50 years urban sprawl has accompanied the growth of urban areas across Europe and during 1990–2000, urban areas increased 5.4 % (or more than 80,00 km^2). This rapid growth mostly occurred in countries and regions with high population density and economic activity (UN-Habitat 2010). Although urbanization in Europe in recent decades has been mostly in the form of spatial expansion rather than population growth, there are also prominent examples of cities that grew very significantly in terms of the number of inhabitants, such as Istanbul with 600 % growth in population and 700 % in the built-up area expansion in the border of İstanbul Metropolitan Municipality between the mid-1950s and the beginning of twenty-first century (EEA 2006; Tezer 2005). At the same time, in some areas of the early industrialized Europe, such as in the Rhine–Ruhr area in Germany, North-West England, Silesia in Poland, the Czech Republic, or Alsace in France, have a range of larger cities that are shrinking in population. This creates new opportunities for innovative use of former residential and industrial areas which have become brownfields (Haase 2013).

The growth of urban areas contributes to an increasing pressure on biodiversity, most importantly by land cover changes, socio-cultural factors, economic development, environmental factors, and administrative failures (EEA 2003). These translate into ecological problems such as habitat fragmentation, degradation and destruction, over-exploitation of natural resources, the spread of alien species, climate change, pollution and waste production.

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The long-lasting urban expansion combined with an alteration of the natural environment, such as soil sealing and land consumption, may also explain why it was first recognized in Europe that nature could adapt to urban areas, and that new niches for species could be provided. The roots of urban ecology, environmental protection, and sustainable urban development can also be found in this continent. For example, researchers in Berlin started already in the 1970s to extensively investigate the city's biodiversity, including plants, animals, and habitats. The data was used in the urban planning of Berlin, and was the first example in the world of systematically incorporating biodiversity data in urban planning. This example of "biotope mapping" was soon followed by other European cities, and today many large cities have long-term monitoring data on vascular plants, different animal groups such as birds, and habitats that are used for city planning and nature conservation. There are also long-standing traditions of designating areas for nature conservation within their borders, for example the National Urban Park in Stockholm (Barthel et al. 2005).

The awareness of goods and services provided by abiotic and biotic urban natures to city inhabitants, and the knowledge about urban ecosystem services, are beginning to find their way into urban planning and land management, especially in Western Europe (Colding 2010; Bendt et al. 2013). The urban space itself needs increasingly be designed to better reflect environmental values and to counteract 'environmental generational amnesia' among urban populations. An interesting form of institutional arrangements for civic management of ecosystem services is *urban green commons*. They include green spaces of diverse land ownership in cities that depend on collective organization and management and that allow residents and citizens to actively work with urban nature in ways that support ecological processes and that promote environmental learning in cities, while allowing for a collective caring of different pieces of land (Colding and Barthel 2013). In the city of Berlin, for example, a fiscal crisis in the early 2000s led to cuts in the funding for public green spaces, which in turn has led to an increase of civic engagement in the management of the urban greens, and an increase in urban gardening (Rosol 2010). However, significant barriers to effective adaptation of ecosystem-based approaches into policy-making, planning and management remain. For example, a recent study in Poland, which represents an example of the countries in Central and Eastern Europe, indicates that these barriers include insufficient funds, lack of local spatial management plans, regulations that downplay the significance of urban greenery, and the fact that the society perceives other issues as more pressing (Kronenberg 2012).

Despite the challenges, there is a growing recognition across the region that to support a sustainable urban development and counteract the current negative changes to the ecosystems connected to urbanization, there is a need to reform institutions and governance mechanisms. In this context, it is essential to counteract the dominant and on-going privatization trend of public land in cities (Lee and Webster 2006), and to safeguard a diversity of property-rights regimes to land in cities (Colding and Barthel 2013). Discussions on future climate changes faced by European cities, further increases the realization that cities need to be progressively adaptable to changes, such as less predictable rainfall and temperature regimes.

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Chapter 14
Regional Assessment of North America: Urbanization Trends, Biodiversity Patterns, and Ecosystem Services

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North America contains some of the most urbanized landscapes in the world. In the United States (U.S.) and Canada, approximately 80 % of the population is urban, with Mexico slightly less (Kaiser Family Foundation 2013). Population growth combined with economic growth has fueled recent urban land expansion in North America. Between 1970 and 2000, urban land area expanded at a rate of 3.31 % (Seto et al. 2011) creating unique challenges for conserving biodiversity and maintaining regional and local ecosystem services.

At the continental scale, North America has only a small amount of its surface in developed land cover (Latifovic et al. 2012). As of 2005, approximately 0.9 % of the continent was classified as developed land (Fig. 14.1) (Commission for Environmental Cooperation 2013). Although not directly comparable along the three countries, Canadian mapping efforts reported approximately 0.2 % (or approximately 199,700 km^2) of the country classified as "settlements" (Statistics Canada 2012) and in Mexico, the governmental mapping agency reported about 0.6 % (or about 118,400 km^2) of the country as "human settlements" for the 2002–2005 time period (Jimenez Nava 2008). However, this relatively small urban and built up land area has had intense impacts on non-urban landscapes (Grimm et al. 2000).

In the U.S., urbanization continues to drive conversion of a variety of land covers and uses to urban development with approximately 4–6 % (an approximate range of 323,200–484,800 km^2) of land classified as developed in the conterminous U.S. (U.S. Department of Agriculture 2009; Multi-Resolution Land Characteristics Consortium 2012). Between 1973 and 2000, new developed (urban and built-up) land cover in the conterminous United States came primarily from conversion of agriculture, forest, grassland/shrublands, and wetlands (Auch et al. 2012). Agricultural land cover supplied the most new developed land during this time period (an estimated 34,142 km^2) and wetlands the least (an estimated 2,792 km^2). Conversion of agriculture to developed land was a consistent pattern across the country whereas conversion of forest to developed land was more concentrated in the eastern half of the U.S., as well as the Pacific Northwest, and grassland/shrubland conversion occurred mostly in the western half of the U.S. Wetland to developed land cover conversion was primarily concentrated in the Southeast (Sleeter et al. 2013). Overall, the U.S. Geological Survey's Land Cover Trends project estimated a 77,529 km^2 increase in developed land in the conterminous U.S. between 1973 and 2000, a 33 % change from 1973 (Sleeter et al. 2013).

Developed land use conversions directly impact other land covers and the ecosystem services they provide (see Chap. 10). Even though estimates of developed land in North America as a whole are small compared to the continent's total land extent, geographic scale is critical. The land change intensity at the local or even regional levels can be much more important than urbanization at the national or

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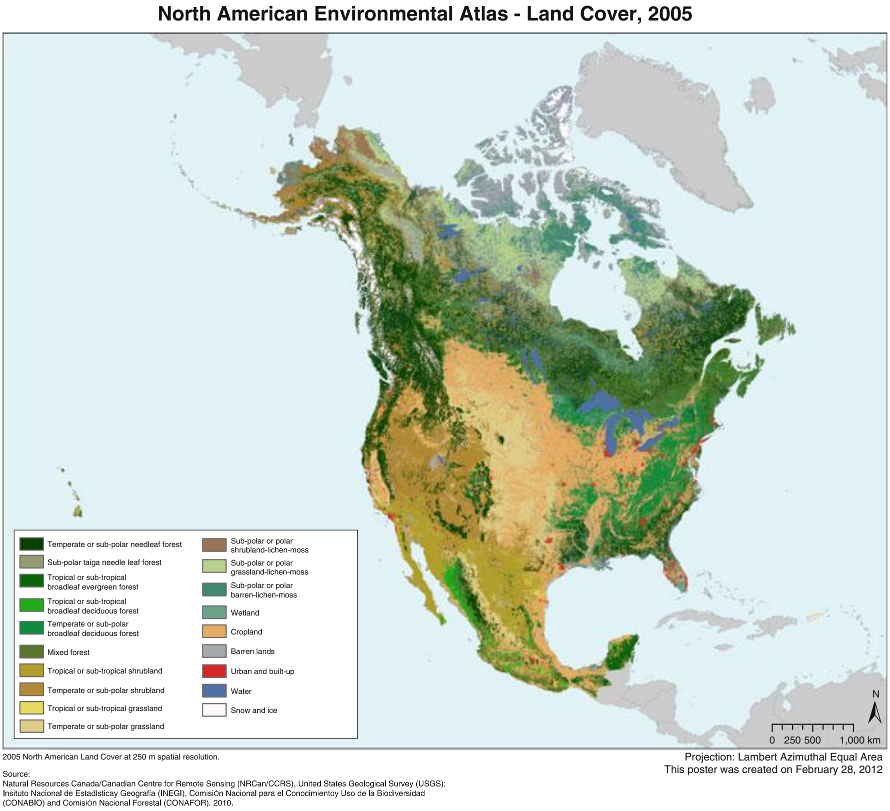

Fig. 14.1 Land cover in North America, 2005 (Published with kind permission of ©Commission for Environmental Cooperation 2005, part of The North American Land Change Monitoring System (NALCMS), a joint project between the United States, Canada, and Mexico. All Rights Reserved)

continental scales. Urbanization also reaches far beyond the local or regional hinterlands relying on additional land uses including agriculture, forestry, and mining to supply urban populations with ecosystem-derived goods and services. The indirect impacts of urbanization by land uses ancillary to supporting metropolitan regions can also affect non-urban ecosystem services and bring land change to remote rural areas of the continent.

Given diverse histories, cultures, and social-ecological traditions in North American cities, dynamics of urbanization vary widely across the continent. Cities in the U.S. and Canada share a complex pattern of shrinking and/or shifting patterns of population in central parts of the cities coupled with sprawling development in outer suburbs and exurban areas. Predictions for future urbanization patterns range from additional shrinkage in cities with decaying urban cores to rapid expansion in urban regions where new economic centers have been developed, with continued rapid growth in megacities such as New York City (see Chap. 19) and Mexico City. However, despite decades of theoretical and methodological improvements, land change models are still poor in predicting future growth patterns (Pontius et al. 2008).

Metropolitan areas often include substantial amounts of natural and semi-natural remnant habitats that are under threat of development or impaired by habitat changes tied to changing land management practices. For example, vacant land is an underutilized yet persistent part of the urban fabric in inner cities and older suburbs (Burkholder 2012). In the U.S., vacant land in cities of more than 100,000 people has historically varied between 19 and 25 % of total land area, while for cities with populations greater than 250,000, vacant land regularly comprises between 12.5 and 15 % of total land area (Kremer et al. 2013). Research on urban vacant land is growing, but has yet to reveal the value of this significant proportion of urban land area for biodiversity and ecosystem services (but see McPhearson et al. 2013).

Urban areas contain a diverse range of habitats created and managed by homeowners, property managers, and local governments. Biodiversity conservation programs in North American cities are enhanced by a long tradition of urban wildlife and urban forestry programs run by state/provincial and local governments. These programs have resulted in habitat conservation and restoration projects, tree planting and urban greening efforts (McPhearson et al. 2010), and efforts to involve local residents in conservation projects near where they live. For example, the MillionTreesNYC program in New York City, a public-private partnership between the city's Department of Parks & Recreation and the non-profit New York Restoration Project, will plant one million new trees in the city to expand canopy cover and increase the delivery of related ecosystem services (McPhearson 2011). To date over 650,000 trees have been planted since the program began in 2007 (see Chap. 19).

Non-governmental organizations have also been involved in biodiversity conservation programs in North American cities. Their efforts include volunteer-led monitoring and restoration projects, programs promoting conservation practices in yards and gardens, and education and advocacy programs (Connolly et al. 2013). Indeed, urban ecosystems represent unique opportunities to expand urban environmental education (Tidball and Krasny 2010; McPhearson and Tidball in press). In the U.S., extension programs run by state universities provide information on conservation practices to urban residents and to local governments.

Rapid growth, land use, tourism and development, and regional and global demand for natural resources have been altering the land and seascapes of North America, which, combined with Central America, is home to four Biodiversity Hotspots (Myers et al. 2000) and the most biologically important desert wilderness areas on Earth. Stretching south from California, U.S. and its unique chaparral and redwood forests toward Panama through woodlands, deserts, and rain forests, North and Central America is rich in unique and threatened wildlife, including black howler monkeys, yellow-headed parrots, California condors, and rodents found nowhere else on Earth.

Urban areas increasingly expand into wild lands (Pickett et al. 2011) affecting the biodiversity in these habitats, which often include endemic species and habitats critical for the provisioning of urban ecosystem services. Cities are no longer compact, but rather sprawl in fractal configurations (Batty 2008). Indeed, even for many rapidly growing metropolitan areas, suburban zones are growing much faster than other zones (Katz and Bradley 1999). These new forms of urban development including exurbs, edge cities, and housing interspersed in forest, shrubland and desert, bring people possessing urban financial equity, habits, and expectations into daily contact with habitats formerly controlled by agriculturalists, foresters and conservationists (Pickett et al. 2011).

Cities often harbor rich biodiversity, and this is true of North American cities. In New York City for example, 85 % of the diversity of flora in New York State exists within the city's municipal boundaries (see Chap. 19). However, the composition of urban and suburban ecosystems differ from wild and rural ecosystems. Species richness has increased in urban forests of the U.S. as a whole, but this is largely due to the presence of exotic species (Zipperer et al. 1997). Exotic species often have a large presence in urban vegetation. In the U.S. urban flora in general, the proportion of exotics has steadily increased over time (McKinney 2002). Rapoport (1993) found the number of non-cultivated species decreased from fringe toward urban centers in several Latin American cities. For example, in Mexico City, there was a linear decrease in the number of species per hectare from 30–80 encountered in suburbs to 3–10 encountered in the city center. Paths in rural recreation areas (Rapoport 1993) and in urban parks (Drayton and Primack 1996) enhance the presence of exotics (Pickett et al. 2011). In an urban park in Boston, of the plant species present in 1894, 155 were absent by 1993, amounting to a decrease from 84 to 74 % native flora. Sixty-four species were new. Similar patterns were found in New York City. In a review of historical records of urban flora in NYC, as of 2000, 42.6 % of the native plant species have been extirpated (DeCandido et al. 2004).

Urbanization affects biodiversity through direct and indirect changes in biotic interactions and trophic dynamics that affect the viability and distribution of species (Marzluff 2001; Hansen et al. 2005). Land cover change and human activities introduce novel disturbances, chronic stresses, unnatural shapes, and/or new degrees of connectedness (Urban et al. 1987). Ecological studies are showing complex relationships between settlement patterns and selective phenotype trait diversity (Faeth et al. 2005). Such complexity is particularly evident when examining the relationship between urban development and biodiversity across a gradient of urbanization. Marzluff (2005) showed that bird diversity in the Central Puget Sound region (U.S.) peaks at intermediate levels of human settlement primarily because of the colonization of intermediately disturbed forests by early successional native species, despite the extinction of native forest birds which increases linearly with loss of forest. Intermediate disturbance due to increased landscape heterogeneity appears to drive diversity (Marzluff 2005).

The vast majority of all urban ecological studies so far have been conducted in cities in Europe or North America (Chap. 27), yet there is still a lack of experimental approaches, and most urban biodiversity studies have focused on either birds or plants. There is also a lack of long-term data, but two Long-Term Ecological Research (LTER) sites in North America, Baltimore and Phoenix, are generating valuable information on the dynamics of the urban landscape from an ecological and biodiversity perspective (Grimm et al. 2000; Pickett et al. 2011). Recent findings from observations in urban ecosystems are showing that new environmental gradients and novel ecosystem functions emerge from complex human-natural interactions, indicating the need to revisit traditional concepts and methods for studying biodiversity and ecosystem function in urbanizing regions (Alberti 2010).

Long-term study of urban systems can serve as model systems for examining the interaction of social and biophysical patterns and processes more broadly (Collins et al. 2001; Redman et al. 2004). In addition, many of the changes in urban areas anticipate the otherwise unprecedented alterations that will follow global environmental change in other ecosystems (Grimm et al. 2008). As urbanization continues to expand, city planners and policymakers need to consider how ecological resources can be strategically developed and managed sustainably to meet the needs of urban populations (McPhearson et al. 2013). Developing a blueprint for mapping and modeling biodiversity and ecosystem services (Crossman et al. 2013) in urban regions will be important for cross-city comparisons and more nuanced understanding of the contribution of urban ecosystems to human livelihoods in cities and urbanized regions (see Chap. 10).

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Chapter 15
Regional Assessment of Oceania

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Oceania is defined by the United Nations as the islands within Polynesia, Micronesia and Melanesia, Australia and New Zealand. The islands in the Pacific Ocean were urbanized relatively recently (typically following independence in the latter half of the 1900s,) but has increased rapidly since the 1970s due to both high population growth rates and inward migration to the amenities of urban centers. In addition, changing economic realities associated with agriculture such as fewer rural jobs due to larger, more productive farms, makes it difficult for people to make a living in the rural areas. At the same time the greater provision of services in urban areas help attracting people to the cities.

Excluding the population of Papua New Guinea, more than half of all Pacific Islanders now live in urban areas. In some atoll states, urban growth has produced very high population densities, comparable to those in densely populated Asian cities. Many of these urban communities continue to lead subsistence lifestyles. This makes them particularly susceptible to ecological degradation resulting from catchment deforestation, pollution of shallow groundwater, and disposal of wastes on near-shore marine ecosystems.

Both Australia and New Zealand are highly urbanized, with over 85 % of their populations living in urban areas (World Factbook 2010a, b). However, the densities of their cities are relatively low by global comparisons. Further, Australia's large land area and relatively small population size (22 million in 2012 (Index Mundi 2013)) makes it one of the world's most sparsely populated countries, with fewer than three people per km^2.

Presently urban areas in Oceania occupy 10,450 km^2. This area is projected to double by 2030, with the majority of growth to be concentrating around existing urban centers. With many urban areas in New Zealand, Micronesia, Melanesia, and Polynesia positioned within biodiversity hotspots, this future urban growth is likely to have significant deleterious affects on biodiversity.

It is increasingly recognized that the impacts of urban growth and associated agricultural expansion on biodiversity need to be mitigated if ecosystem services generated by this biodiversity are to be maintained in the future (see Chap. 22). While the majority of ecosystem services research in Australia and New Zealand has focused on agroecosystems and farming practices, there is an increasing effort to understand the nature and role of ecosystem services in cities. In Pacific Island countries the major priority remains economic and social development. Nevertheless, there is growing regional recognition that maintaining the ecological support systems of vulnerable islands is essential to their sustainability.

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Chapter 16
Local Assessment of İstanbul: Biodiversity and Ecosystem Services

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Abstract İstanbul, with a population of over 13 million, is Turkey's most populated metropolitan area and the economic powerhouse of the country. The city is located in a region that has rich biodiversity due partly to its unique location at the crossroads of two continental landmasses and two large water bodies. The geographical characteristics and the city's topography allow for diverse micro-climatic zones to exist in a relatively small area, 5,461 km^2. Moreover, due to millennia of human settlement including sixteen centuries as an imperial capital, many exotic species have found their way to the region. This assessment provides an overview of the main challenges and opportunities that İstanbul is faced with in regards to biodiversity conservation and support for the ecosystem services upon which the city depends, while simultaneously managing population growth and economic development. The assessment will also highlight the Ömerli Watershed, which supplies most of the city's drinking water. The watershed's freshwater provisioning capacity has been degrading due to urbanization in its catchment area, while the demand for water in the city overall has been increasing. An ecological-asset evaluation of the watershed has been carried out to develop an ecosystem services-based spatial decision-making framework. The evaluation is part of the urban biosphere reserve initiative that may be a solution to prevent further decrease of the watershed's biodiversity and degradation of its ecosystem services.

Keywords Illegal settlements (gecekondus) • Urban watersheds • Ecosystem services • Freshwater • İstanbul

Key Findings

- Although at a slower pace than in the past decades, the population of İstanbul will continue to grow due to immigration, and the provisioning of freshwater will continue to be of critical concern for the foreseeable future.
- Most of the biodiversity-rich areas are lacking formal protected status.
- While the metropolitan government is cognizant of environmental problems of the city, biodiversity and ecosystem services are not integrated into its spatial and strategic plans.
- Ömerli Watershed, providing most of the drinking water of İstanbul, has the highest ratio of illegal urban development among all the other basins in the city's boundaries that provide freshwater.
- The Urban Biosphere Initiative may provide a rational approach for the integrated use and protection of ecological assets in the Ömerli Watershed.

16.1 Geography and Historical Background

İstanbul, a world heritage site straddling two continents at a strategic location, has been the capital of four empires uninterrupted for almost 1,600 years from AD 330 to AD 1923 (Necipoğlu 2010). The city also lies astride on the seaway between the Black Sea to the north and the Marmara Sea to the south. Both European and Asian sides of the city have hilly topography with the highest point being Aydos Hill on the Asian side (537 m). İstanbul's climate is broadly characterized as warm-summer Mediterranean but includes many microclimatic variations. The city's location, several climatic zones, diverse geo-spatial characteristics, and long history of human settlement, have all contributed to the area's rich biological diversity. The location and climate make the region a crucial crossroads of migration routes, supporting seasonal movement of many species (Yaltırık et al. 1997).

The earliest human settlements have been dated to 8,000 years ago, which suggests that this area (Thrace and Anatolia Peninsulas of İstanbul Province) was one of the major migration routes of humans in Paleolithic periods (Özdoğan 2010). The city came to be known as Constantinople ("the city of Constantine") after the Roman Emperor Constantine who, in AD 330, proclaimed the city the sole capital of the whole Roman Empire. After AD 395, it remained as the capital of the Eastern Roman (eventually known as Byzantine) Empire. For the better part of the Middle Ages, Constantinople was the largest and wealthiest city on the European continent, and at times even the largest in the world (Chandler 1984; Modelski 2003). In 1453 the city became the capital of the Ottoman Empire and, already by the end of the fifteenth century, its population reached two hundred thousand, making it the second largest city in Europe. Together with the weakening of the Ottoman Empire, however, the city gradually lost its importance in a process that proceeded well into the twentieth century.

16.2 İstanbul's Transformation from the Mid-1900s

Starting in the late 1940s and early 1950s, İstanbul has undergone great changes. From 1955 to present, İstanbul's population and built-up area have grown rapidly. Throughout the city, new public squares, boulevards, and avenues were constructed or existing ones revamped (Gül 2009). Also, as a response to social, cultural, and economic changes across the country, migrants mostly from rural Anatolia started flowing to the city. Especially after the 1970s, the population of the city rapidly increased following the prospect of finding jobs in the booming metropolis and the promise of a better life than what the rural livelihoods had to offer (Tümertekin 2007) (Fig. 16.1).

In spite of the major demographic shifts in the late 1950s and 1960s, far too few investments in planned mass housing projects were made to meet the demand. The early squatter areas (*gecekondu* in Turkish, meaning 'built overnight') emerged

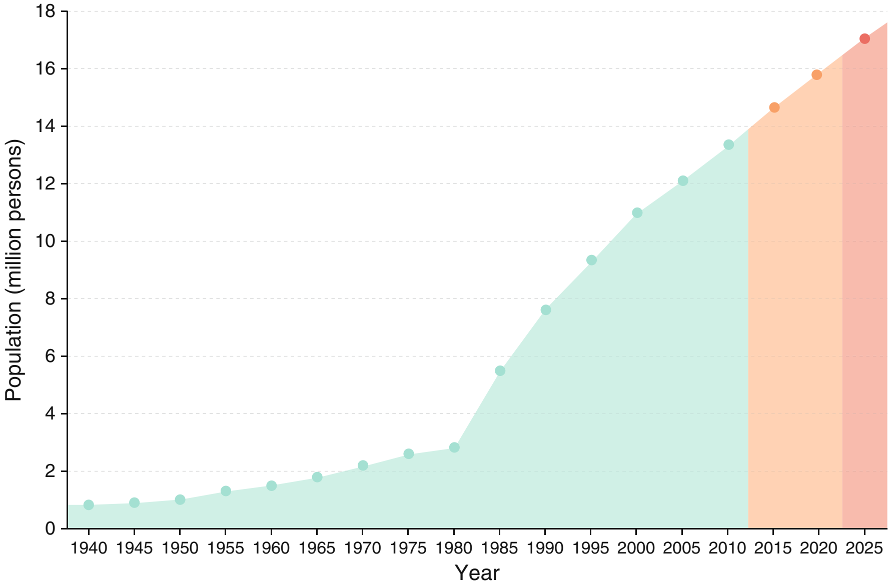

Fig. 16.1 Population of İstanbul from 1940 to 2025. The projections for 2015 and 2020 are from TurkStat (2012). The projection for 2025 is the authors' calculation based on TurkStat (2012) projections

as a consequence of urgent housing needs and the lack of the ability of the government to supply social housing for low and middle-income households. Since then, this uncontrolled rapid development has been creating a heterogeneous and scattered urban fabric. The resulting environmental and socio-economic problems have led to several administrative challenges (Tezer 2004; Erkök 2009).

While the laws and regulations are in place to protect the forests and basins, the enforcement is often lacking. More disconcerting is that the illegal residential settlements in these ecologically sensitive areas are tolerated and legalized with political motives (Bekiroğlu and Eker 2011). Several amnesty laws were issued in the past to legalize illegal settlements (Uzun et al. 2010). Rather than preventing the construction of new illegal settlements, the expectation of upcoming amnesty laws further encouraged illegal and unplanned developments on the outskirts of the city throughout the 1980s. Furthermore, the illegal or unplanned developments in the recent decades are no longer driven by a shortage of housing and they are now more speculative in nature. More importantly, these developments are increasingly within the watersheds that are critical for the water provisioning to the city (Tezer et al. 2011). In a parallel process, gated communities sprung up across the forested hills of the scenic Bosphorus as well as in outlying areas around the city while high-rises (both commercial and residential) mushroomed in more central locations. The population increase, the densification of the city centre and the strengthened commercial profile of the city paved the way for it to eventually aspire to be a "Global City" of the twenty-first century (Keyder 1999).

The costs of the development are many, including erosion of the traditional İstanbul culture, with its vernacular architecture, and the loss or dwindling of biodiversity and ecosystem services in and around the urban areas (Keyder 1999; Tezer 2005; Tezer et al. 2008). The rapid expansion since the 1980s of new settlements into the forests and water basins north of the city threatens one of the most critical ecosystem services that the city depends upon: the provision of freshwater. Today, İstanbul is home to more than 13 million people (65 % on the European side; 35 % on the Asian side), about 18 % of the national population, and contributes more than one fifth of Turkey's GDP, that is approximately US$150 billion (TurkStat 2012). While the last two population censuses indicate that the rate of population increase has been slowing down to 3 % annually, the absolute population growth is still high enough to continue to affect changes in urban structure and place significant pressure on natural resources.

16.3 Governance and Institutions

The İstanbul Province has a governor (*vali*) that is appointed by the central government in the capital Ankara. This provincial government used to be the main urban administration body with numerous district municipalities having limited responsibilities within their own jurisdictions. In the early 1980s, when metropolitan municipalities were established as the country's largest urban settlements, most of the responsibilities of the provincial governments were transferred to these new local authorities. Since 1984, İstanbul has a metropolitan municipality whose mayor is elected by the citizens of İstanbul for five-year terms. However, the delegation of power from Ankara to local governments did not result in a true civil engagement in urban governance in İstanbul. An opaque management structure is still prevalent in local municipalities, which permits frequent misuse of political power (Tekeli 2009).

The management of ecosystem services has been plagued with poor coordination among the multiple responsible authorities. The fragmented governance structure and the complicated legal system are major problems particularly in the management of water resources for İstanbul that experiences chronic water shortages. For example, even though drinking water and sewage services are responsibilities of the İstanbul Water and Sewage Works (İstanbul Su ve Kanalizasyon İşleri, İSKİ) of the İstanbul Metropolitan Municipality, the management of forests within the administrative boundaries of İstanbul falls under the responsibility of the İstanbul Forest District Directorate. The Directorate is ultimately tied to the Ministry of Forestry and Water Works and thus to the central government. For the forested areas around the city, in general, the primary objective remains to be timber production. Water provision is a lesser objective along with recreation and wildlife protection (Bekiroğlu and Eker 2011). Moreover, the city is increasingly relying on water sources that are located further from the city itself, and are therefore under the auspices of other governance bodies such as State Hydraulic Works (Devlet Su İşleri, DSİ) or local municipalities.

To alleviate the operational and legal difficulties of bringing water from sources beyond the city's boundaries, the authorization of İSKİ has recently been extended to management of the lakes, dams, and other water infrastructure beyond the city's administrative boundaries. On the other hand, in Turkey, the General Directorate of Forestry sets aside those areas that are critical for clean water provision as protected lands, which are under the purview of the General Directorate of Nature Conservation and National Parks. While by law, these organizations should cooperate in their operations, there is little coordination among them (Bekiroğlu and Eker 2011).

With the rising popularity of concepts such as "ecosystem services" and "natural capital", the forests of İstanbul have taken on a renewed meaning in the eyes of the planners, city officials, and concerned citizens as tangible and intangible assets of the city. In this vein, a promising initiative is the Urban Biosphere Reserve (UBR) approach (Tezer 2005). The initiative envisions an integrative policy instrument targeting the sustainable management of urban aquatic habitat within the Ömerli Watershed, a critical source in helping meet the drinking water demand of the city (see Sect. 16.5). Such novel governance approaches in urban administration can be critical in ensuring preservation of ecosystem services and conservation of biodiversity in the face of relentless urban development. This in turn would help ensure that rapidly growing urban areas would not choke themselves off by cutting off their life-support systems.

There is an active civil society in İstanbul on matters relating to the conservation of biodiversity and ecosystem services. It would, however, be hard to claim that a majority or even a substantial portion of the city's inhabitants are genuinely concerned about such issues related to the long-term wellbeing of the city. Based on surveys commissioned by the Urban Age Programme, a network of researchers from various research institutions around the world, while İstanbul residents seem to be concerned about environmental problems more than those in, for example, London or São Paulo, climate-related ones such as water shortages and heat waves rank the highest (Page et al. 2010). Impacts on biodiversity ranks a distant 11th in a list of 18 environmental concerns directed at survey participants from İstanbul. Only 33 % of participants identified this as a primary environmental concern. This is perhaps not unexpected for a city that has always faced water shortages in its history. Obviously, the impacts of water shortages and heat waves are much more visible to İstanbulites compared to the contribution of biodiversity to their well-being.

Among the civil society organizations in Turkey that place ecosystem services and biodiversity to the top of their agenda is the Turkish Society for the Protection of Nature (Doğal Hayatı Koruma Derneği, DHKD). The DHKD has in the past conducted research on sensitive areas of İstanbul that are important for conservation of biodiversity (DHKD 1999; Byfield et al. 2010). It also used to be actively involved in initiating or furthering the protected statuses of these areas. Likewise, Doğa Derneği (Nature Society), founded in 2006, pursues much the same goals as the DHKD.

16.4 Current State of Biodiversity and Ecosystem Services in İstanbul

İstanbul, while famous with its cultural heritage, is not as well known for its natural heritage and the richness of its biodiversity. The unique geographic location and diversity of natural characteristics can be classified into five different natural habitats (Table 16.1, DHKD 1999). In İstanbul, there are almost 2,500 native-vascular floristic and fern species (Byfield et al. 2010). There are seven Important Plant Areas (IPAs) and four Important Bird Areas (IBAs) (Byfield et al. 2010; Magnin and Yarar 1997). These areas are also collectively labeled as Key Biodiversity Areas, KBAs (Eken et al. 2006). Large portions of these IBAs and IPAs are unprotected and under intense pressure from urban expansion (Byfield et al. 2010). Only limited protection is afforded to those sections that have "natural site" designations such as those located within the Bosphorus Forefront Area, watershed protection zones, and nature parks.

İstanbul, located on one of the major bird migration routes, is home to four IBAs. However, two of these IBAs, the Büyükçekmece and Küçükçekmece Lakes, once popular hunting spots, have already been extensively urbanized (Magnin and Yarar 1997). The region is also home to flora that is threatened with either local or global extinction. Two hundred and seventy of these plants are in the national list of threatened rare and endemic plants (Avcı 2008). One of these IPAs is the Ömerli Watershed, which does not only harbor many endemic or endangered plant species but also provide a vital ecosystem function as a freshwater resource (Tezer 2005). Although the Ömerli Watershed is not originally categorized among the IBAs of İstanbul, its location is nevertheless very significant for bird abundance: after the construction of the reservoir it has gradually become home for more than 100 bird species (Tezer et al. 2011).

Open-pit coal mining and quarries along the Black Sea shores of the city supplied, until recently, part of the city's demand for fossil fuel and construction material. These mining operations, long practiced in the region, especially along the Kilyos-Terkos coastal strip destroyed coastal dune habitats that are critical for many endemic species (Byfield and Özhatay 1995). In part due to coal mining, and in part due to the expansion of residential areas, the coastal habitat decreased from about 450 ha in early 1980s to about 155 ha in early 2000s along the Kilyos-Terkos coastal strip (Doğru et al. 2006).

With its centuries-long history as an imperial center, the city houses many exotic species brought from various places around the world. Some of these such as magnolia (*Magnolia grandiflora*) native to the southeastern U.S. and horsechestnut (*Aesculus hippocastanum*), native to the northern Greece have long been familiar elements of the İstanbul cityscape as well as its cultural fabric (Yaltırık et al. 1997; Lack 2000). On the other hand, one of the most well-established invasive species is the tree of heaven (*Ailanthus altissima*) from China that grow in derelict areas around the city, near highways and railroad tracks as well as parks (Avcı 2008).

Table 16.1 Nationally and globally important habitats in İstanbul

Habitats	Existing threats and damage
Grasslands	
Once, southern parts of the European side of İstanbul were covered completely with rich floristic species of limestone grasslands. However, most were either lost to rapid urbanization or converted to cropland (wheat and sunflower). There are still some remnant grasslands which accommodate rich endemic species of flora and fauna.	It is the most degraded habitat in the province. It is estimated that less than 10 % of the initial grasslands preserve their natural characteristics at present.
Forests	
They represent the largest natural habitats. İstanbul's forests are very rich with floristic species and they form the western terminus temperate rainforests along the southern coast of the Black Sea. The majority of İstanbul's forests are used for firewood and fire-coal extraction according to the strict regulations of the Ministry of Forestry. Forests still constitute the largest land cover in the province (almost half of the total area).	Vulnerable to wildfires. Moreover, illegal and unplanned development and agricultural land expansion are two other major threats. The planned third bridge crossing on the Bosphorus and its connecting highways on both sides of İstanbul may also cause serious degradation.
Heathlands	
Once, the southeast part of the Asian side was covered with large heathlands (*Ericaceae*) (estimated 95,000 ha). Although they are extensively damaged, they still accommodate the most diverse rare and endangered species. İstanbul's heathlands represent the largest and the most intact remnant habitat of this kind in the eastern European and Mediterranean regions.	The majority of the habitat in urban areas is degraded today. The major threats to these habitats are the pressures originated by urban development, agricultural expansion, and poorly devised afforestation efforts carried out without proper ecological evaluations.
Coastal Dunes and Habitats	
The coastal dunes of İstanbul come second after the heathlands in terms of having the most diverse rare and endangered species. The most important are located in 15 different locations along the Black Sea coast of İstanbul.	In the past, the total area of coastal dunes along the Black Sea coast of İstanbul was more than 5,600 ha. However, more than half of these habitats have been destroyed since the 1960s. Urban development and construction of highways caused serious degradation and loss of the coastal dunes along the Marmara Sea coast. Mining and residential development remain to be the major threats for this habitat.
Wetlands	
Büyükçekmece, Küçükçekmece, and Terkos lakes on the European side; Riva and Ağva streams on the Asian side are the important wetlands of İstanbul. They all have rich aquatic habitats. Terkos wetlands in particular have the most diverse aquatic habitat in Turkey.	Terkos, Büyükçekmece, and Riva are used to supply drinking water. Hence, they are better protected under the regulations of the İSKİ. However, Küçükçekmece's wetlands are under the threat of agricultural and industrial expansion as well as residential development. Moreover, the third International Airport may have serious impact on the water quality, should it be built within the Terkos Watershed as planned.

Source data from DHKD (1999). Prepared by and published with kind permission of

Table 16.2 Drinking water supply–demand and deficit of İstanbul

Year	Population	Demand (10^6 m^3/year)	Supply (10^6 m^3/year)	Deficit (10^6 m^3/year)
1995	8,417,000	771.0	451.0	320.0
2000	10,019,000	939.0	757.0	182.0
2005	11,332,000	1,298.0	762.0	536.0
2010	12,915,000	1,635.5	952.5	682.0

Source data from http://www.iski.gov.tr

Parks and gardens scattered around İstanbul are mostly remnants of imperial woods and gardens. These gardens and parks harbor an impressive biodiversity in İstanbul populated by both native and exotic species collected over millennia. They also perform an important function as green spaces for which İstanbul, once famous for its gardens, is sadly lacking (Kara and Demirci 2010). Cemeteries, historically an important part of the urban fabric of the city, changed little as the city transformed and expanded around them; these cemeteries provide refuge to native terrestrial gastropods (Örstan and Kösemen 2009) and potentially to many floristic species as well. Prince Islands, within the administrative boundaries of İstanbul, have so many exotic tree species that they are said to have become arboretums in their own right (Yaltırık et al. 1993).

The Marmara Sea, the Black Sea, the Bosphorus strait, and the inlet Golden Horn as well as the nearby freshwater bodies all used to harbor rich aquatic biodiversity. Especially during the twentieth century, increased urbanization brought with it an increase in maritime traffic, the discharge of untreated effluents (i.e., sewage from the residential areas and wastewater discharge from the industrial facilities), and overfishing. All these factors played their role in decimating the once abundant aquatic life in these water bodies (Avcı 2008). Thanks to the significant improvements in treating the effluents and a massive rehabilitation effort, the Golden Horn regained some of its former aquatic biodiversity (Yüksek et al. 2006). However, its recovery will probably remain incomplete because the water quality in the estuary is influenced by the Black Sea and the Marmara Sea whose aquatic biodiversity has been severely degraded and remain so due to effluents from the urban areas (Uysal et al. 2002; Albayrak et al. 2006). Importantly, the provision of seafood, a critical ecosystem service these water bodies had been providing to İstanbulites for ages, is now severely degraded (Turkish Ministry of Environment 2002).

In addition to the vulnerability of the city to destructive earthquakes, the other major environmental concern in İstanbul has historically been the persistent challenge of securing water needs of its inhabitants. The region lacks large freshwater sources and provisioning of sufficient water to the city has been a persistent problem throughout the ages (Crow 2012; Çeçen 1992). The forests north of the city have been crucial since the Roman times in provisioning of the drinking water to the inhabitants of the city. This historical challenge continues today as the city's drinking water deficit has kept on increasing even though the supply has more than doubled over a 15-year period (Table 16.2). At the same time, especially since 1980s, the forests and water basins north of the city have been experiencing considerable degradation due to urban expansion. This expansion not only further reduces what little habitat had been left to support native biodiversity but also degrades the water-provisioning capacity of those areas.

Of the seven major watersheds that have historically supplied more than three quarters of the city's drinking water use, Küçükçekmece Lake has been seriously contaminated as a result of intense urbanization in its basin; thus, it is no longer a source of drinking water (Kucukmehmetoglu and Geymen 2008). The other two freshwater basins, Alibeyköy and Elmalı have also lost much of their capacity due to similar concerns with contamination from urban effluents (Tezer 2005). Even the least degraded Darlık basin suffers from illegal constructions that now occupy about 10 % of the basin (Bekiroğlu and Eker 2011). Lately, the plans for the third bridge and the connecting highways (Northern Marmara Highway) to cross through forests and watersheds are causing much controversy. Another controversial plan is the construction of the third International Airport within the Terkos Watershed. In a prime example of top-down decision-making, these two significant development projects are planned under the authority of the Central Government in Ankara.

16.5 Case Study: The Ömerli Watershed[1]

The Ömerli Watershed (ÖW) covers an area of 621 km^2 and spreads across two provinces, İstanbul and Kocaeli. Almost 71 % of its land area is in İstanbul (Fig. 16.2). It is the most important of the seven watersheds that provide drinking water to İstanbul and it has exceptional biodiversity (Albayrak 2012). As the second largest drinking water source for İstanbul, it supplies about half of the drinking water demand of the city (Albay and Akçaalan 2003). However, among all watersheds that provide drinking water to the city, the ÖW also faces the most acute pressures of unplanned urbanization (Tezer 2005).

The ÖW had a rural character until the 1970s; especially during the 1980s, it experienced rapid population increase due to immigration. The watershed's population was 24,000 in 1980; it increased by 540 % to 154,000 by 1990, then by 140 % from 1990 to 2000 to 371,000 (Baykal et al. 2003). According to the İSKİ, the population in 2005 was 394,208. Notwithstanding these estimates, it is difficult to determine the exact population of the watershed because of the incompatibility of administrative and watershed boundaries.

The ÖW has been classified as one of the "122 Important Plant Areas (IPA) of Turkey" in a study of the DHKD, as it contains at least 37 rare plants and extraordinary biodiversity (Byfield et al. 2010). However, despite the IPA designation, the area is not formally protected. The only tool to control urbanization is the watershed

[1] This section is largely excerpted from Tezer, A., Ulugtekin, N., Goksel, C., Ertekin, O., Terzi, F. 2011, *Ömerli Watershed: Ecological Assets and Bird Atlas*, Cenkler Matbaası, İstanbul. This book is produced under the TUBITAK Project No.108K615 "Integrating Ecosystem Services into Spatial Planning".

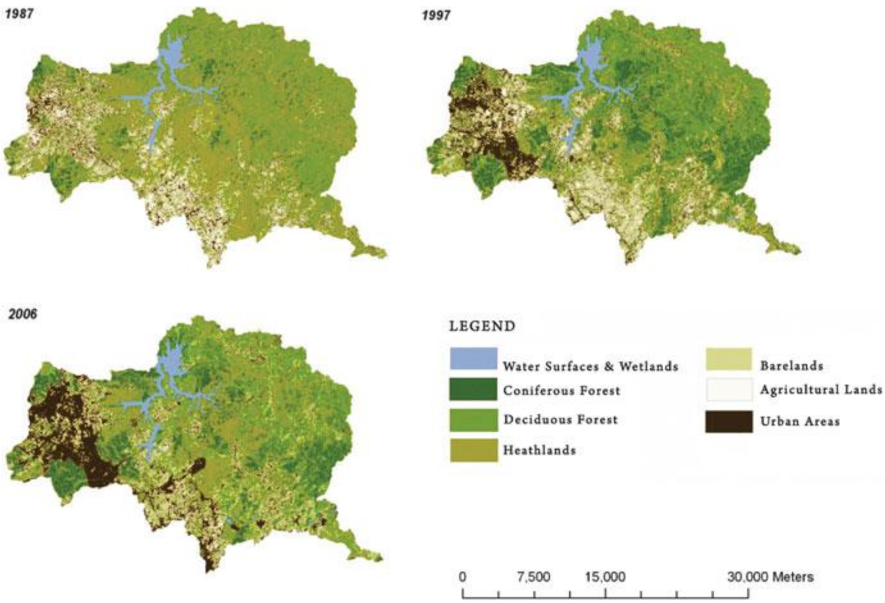

Fig. 16.2 Land use and land cover changes in the Ömerli Watershed, 1987–2006 (Reproduced from Tezer et al. 2011, p. 14 and published with kind permission of ©Azime Tezer, Necla Uluğtekin, Çigdem Göksel, Özhan Ertekin and Fatih Terzi 2011. All Rights Reserved)

buffer zones that were established by the İSKİ Regulation, with the aim to keep development away from the dam reservoir to protect the water quality. The sole criterion for their establishment is the distance from the reservoir, without regards to the ecological characteristics of the watershed. There are four such zones from nearest to farthest from the reservoir: absolute, short-distance, intermediate-distance, and long-distance (Tezer 2005).

16.5.1 Land Use and Ecology

The ÖW has a very rich habitat mosaic comprised of wetlands, heathlands, natural and planted coniferous forests, deciduous forests, meadows and peatlands, and a dam reservoir which was constructed between 1968 and 1972. Before the construction of the Ömerli Dam, the southern areas of the watershed were mainly agricultural lands and the northern parts were generally oak-coppice forests and heathlands (Suher 1963).

The watershed's heathlands are part of the extensive heathlands located on the Kocaeli Peninsula. These heathlands are the largest remnants of their kind across southeastern Europe and the eastern Mediterranean region. Heathlands are rare habitats that exist in humid and temperate regions with acidic soils. They provide a valuable biological diversity of rare birds and plant species, insects (butterflies, oxybelus, coleoptera, etc.), reptiles, and amphibians (Byfield et al. 2010). The forests represent the largest type of land cover in the watershed (63 % of the basin in 2006). Forest areas have been identified as sensitive areas, and consist of oak-coppice forests of Thrace region and black-pine forests. They also contain many rare species such as *Cirsium polycephalum*, *Lathyrus undulatus*, *Cyclamen coum var.coum*, *Galanthus plicatus subsp. byzantinus*, *Lilium martagon* and *Osmunda regalis* (Özhatay and Keskin 2007; Tezer et al. 2008). In the category of water resources and wetlands, the watershed consists of the dam reservoir, the streams nourishing the dam reservoir and wetlands, seasonal ponds and peatlands (Özhatay and Keskin 2007). Agricultural areas expanding on the southeastern part of the watershed are grouped as irrigated, not-irrigated, greenhouse, and other agricultural areas.

In 2000, the watershed area consisted of 51 % forest, 35 % agricultural land and meadows, 10 % settlements and industrial uses, and 4 % water surfaces (Baykal et al. 2003). Another survey found that in that same year, the watershed contained 3,082 ha of residential land, 177 ha of commercial land and 352 ha of industrial land (İlze and Kurt 2003). The industrial areas were located in all four protection zones of the watershed, with strong negative impact on its biological diversity, soil, water and hydrogeological quality (İlze and Kurt 2003; Hürfikir 1994).

There has been significant changes in land use and land cover in the watershed between 1987 and 2006 (Fig. 16.2). Natural areas covered primarily by heathlands and woodlands decreased by 5,000 ha between 1987 and 2006, from 46,000 ha to about 41,000 ha. At the same time, agricultural areas declined by 82 % while built-up areas increased by 169 %. For example, the urban land in the Sultanbeyli District rapidly expanded within the long-distance protection zone around the reservoir and today extends over 3,000 ha with a population of over 282,000 people. The district is unique in İstanbul due to its predominantly illegal urbanization within its boundaries, which has been on-going for decades and directly causes severe degradation of the area's ecology. One notable example was in 1987, when 1,350 ha of state-owned forested area had its status as "forest land" removed by the Directorate of İstanbul Environment and Forestry Department, due to the degradation caused by illegal urban expansion. Although the urban expansion in the district has slowed down in recent years, such actions still encourage further degradation of the forested areas (Özyetgin-Altun 2011).

Land use changes including unplanned residential development, road construction, and construction of the Formula 1 racetrack contributed to ecosystem changes in the basin. Pollution from residential, industrial, and agricultural areas is also an important factor. The changes typically result in fragmentation and degradation of habitats that accommodate rich biodiversity as well as degradation of water and soil quality. The reservoir in particular has been polluted by sewage, industrial wastewater, and soil run-off. The increase in pollution in the reservoir has been shown to lead to frequent toxic blue-green algae blooms from late summer to mid autumn (Albay and Akçaalan 2003).

The first Environmental Master Plan for the ÖW was prepared in 1984 to control the impact of rapid illegal urbanization. The plan identified watershed protection zones around the reservoir and defined spatial development conditions accordingly. This protection zone approach in the watershed was sustained in the 1995 Metropolitan Master Plan; especially any new construction was banned within the absolute protection zone around the Ömerli reservoir and its connecting streams.

16.5.2 Recommendations

An ecological-asset evaluation of the ÖW was carried out with the aim to develop an ecosystem services-based spatial decision making framework. This research was informed by the input from the relevant stakeholders and an ecosystem services-based spatial zoning has been developed to guide the watershed management (Albayrak 2012). The zoning approach is in line with UNESCO's biosphere reserve program to control carrying capacity of the ecosystems and sustain ecosystem services. Thus, in the determination of the spatial zoning for the ÖW, both its biodiversity and socio-economic characteristics in the watershed were taken into consideration.

The ÖW already has protection zones put in place according to the regulations of the İSKİ, for which the distance from the reservoir is the sole criterion. However, distance-based watershed management regulation is no longer seen as adequate to maintain the integrity of the ecosystems and thus preserve ecosystem services (MEA 2005). Identification of ecologically sensitive areas in the case of the ÖW can be the basis for development of a watershed management model that reflects the local ecosystem characteristics (Fig. 16.3). Ecologically Sensitive Areas (ESA) as identified in the evaluation are: Water surfaces and wetlands (ESA-1), areas of rare and endemic plants (ESA-2), heathlands (ESA-3), sensitive forest areas (ESA-4), ground water reserves (ESA-5), and grasslands (ESA-6). The water quality, which is directly related to the quality of the natural environment, can only be maintained by ensuring the well-being of these ESAs. The existing protection zones must be modified in accordance to these ESAs.

16.6 Concluding Remarks

İstanbul's metropolitan area population is projected to be 14.6 million by 2017 and 16.6 million by 2023 (TurkStat 2012), which is slightly above the 16 million cap placed on the city's population for 2023 in the İstanbul Environmental Master Plan on sustainability grounds (İMP 2009). To accommodate the increase in

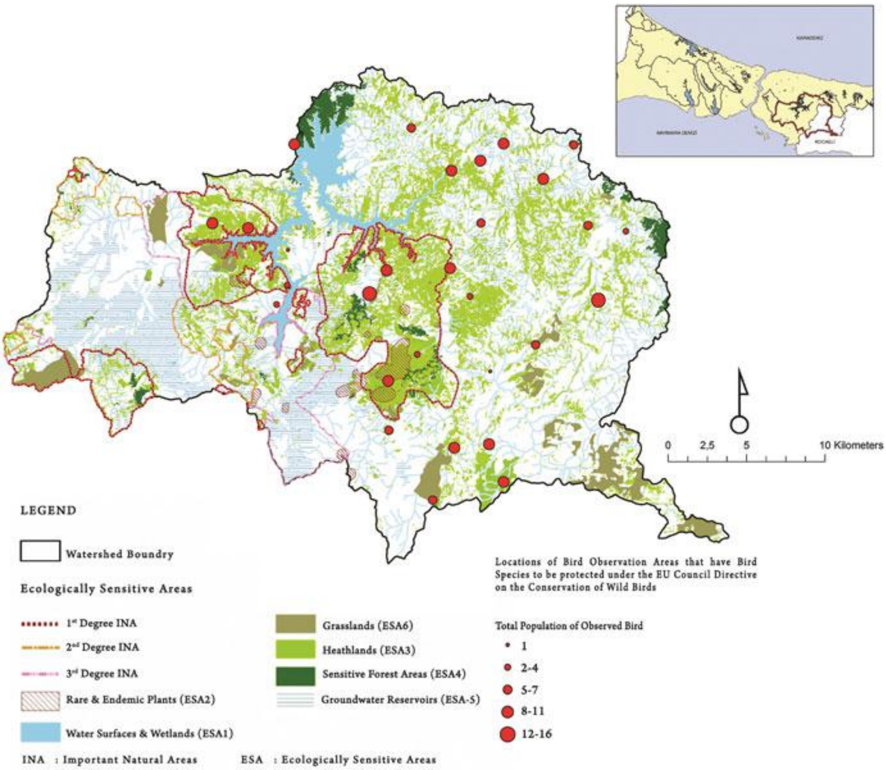

Fig. 16.3 Ecologically Sensitive Areas (ESA) in the Ömerli Watershed (Reproduced from Tezer et al. 2011, p. 26 and published with kind permission of ©Azime Tezer, Necla Ulugtekin, Cigdem Goksel, Ozhan Ertekin and Fatih Terzi 2011. All Rights Reserved)

population, the city will mainly either expand or grow denser, although the population density of the city is already one of the highest in all Europe (Urban Age 2009). Furthermore, the city will continue its economic boom for the foreseeable future led by the growth in its commercial and service sectors. To the extent the growth of the city is accommodated through expansion, more of the natural areas and critical watersheds will come under pressure of urban expansion (Terzi and Bölen 2012).

The 2006 Environmental Master Plan of İstanbul and its revision in 2009 failed to identify effective solutions in regard to illegal urbanization, degraded forests, biodiversity conservation, and ecosystem services and the sustainable use of natural resources. Although, for the first time in the master planning tradition of İstanbul, “significant biodiversity areas” were specifically identified at least in the plan, the identified areas represented only a portion of the actual extents of the biodiversity hotspots of İstanbul (Byfield et al. 2010; Özhatay and Keskin 2007; Tezer et al. 2008).

İstanbul Metropolitan Municipality, cognizant of the implications, has prepared two successive strategic plans for the city, the latest for 2010–2014. While the strategic plan acknowledges the importance of environmental sustainability together with social and economic sustainability, the emphasis regarding the environmental sustainability seems to be almost exclusively on the "fight against global warming" and "adaptation to climate change" (İMM 2010, p. 10). Under the "Environmental Services Management" section, improving environmental protection practices, extending green zones, and developing practices for prevention of marine pollution are listed. There is reference to the natural heritage of the city, in vague terms, and several areas are –though irrespective of the IPA and IBA designations and limited in extent– indicated as biodiversity hotspots in the plan. However, it is notable that neither biodiversity nor ecosystem services are mentioned in the plan report. In general, while the report acknowledges environmental problems and needed initiatives, it is lacking a functional understanding of the importance of biodiversity and ecosystem services for the well-being of the city's inhabitants.

The prospects of biodiversity and ecosystem services in and near İstanbul's urban areas do not look promising in the face of projected changes in the demographic and economic structure of the city. For example, the proposed third bridge over the Bosphorus is a point of contention. The proposal is questioned on its soundness from urban and transportation planning perspectives (Geymen 2013; CUP 2010; Kubat et al. 2007; Tezer 2004). In particular, the induced urbanization around the new transportation infrastructure accompanying the bridge would increase the urbanization pressure on the northern forests and watersheds. Another example that puts the future environmental sustainability of the city in question is the Canal İstanbul project. The ambitious project is part of a grandiose vision of İstanbul and targeted to be completed in 2023. It aims to divert the maritime traffic that now crowd the Bosphorus by building a canal on the European side about 45 km from the Bosphorus as an alternative sea route between Black Sea and the Marmara Sea (Fig. 16.4b). However, aside from its expensive price tag, the various implications of the project on ecosystems and biodiversity are far from certain (Kundak and Baypinar 2011). There is also a rough blueprint for two new cities along the Black Sea coast that are supposed to relieve the population pressure away from the central metropolitan areas of İstanbul. The problem with such a strategy is that it would simply extent the metropolitan area well along the Black sea coast, decimating in the process the coastal and forest ecosystems, some of which are important conservation sites and important freshwater sources for the city (Şekercioğlu et al. 2011). All these developments will most likely speed up the degradation, fragmentation, and loss of the forests and the other key habitats in İstanbul (Figs. 16.4b and 16.5; Table 16.3; Tezer et al. 2012).

The master plans for the İstanbul Province have a history of protecting the forests and watershed areas to the north of the city from development. In particular, the plan prepared in the late 1950s proposed urban development in a linear form along the

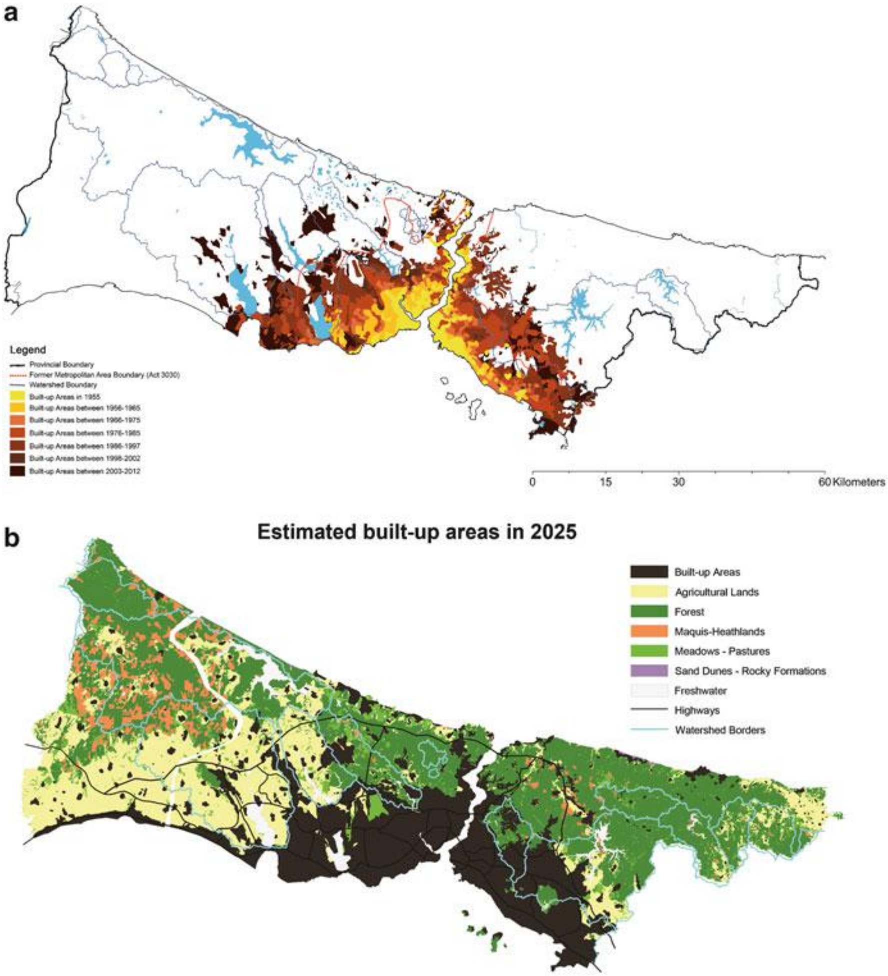

Fig. 16.4 (**a**) Urban expansion in the watersheds of İstanbul Province 1955–2012 (Source data from Tezer Kemer 2005, prepared by and published with kind permission of ©Azime Tezer 2013. All Rights Reserved). (**b**) The projected changes in land cover by 2025 assuming all planned development projects are realized (Modified from Tezer et al. 2012, and published with kind permission of ©Azime Tezer 2013. All Rights Reserved)

east–west axis not to encourage the development to expand through the sensitive northern habitats and natural resource areas. Though imperfectly implemented, this policy was continued in the later plans until the most recent Environmental Master Plan. The planned developments if realized would mean a definite move away from the basic urban development policy of İstanbul that always safeguarded the areas that have been critical for the provisioning of water but also have increasingly been recognized for their value for conservation of biodiversity.

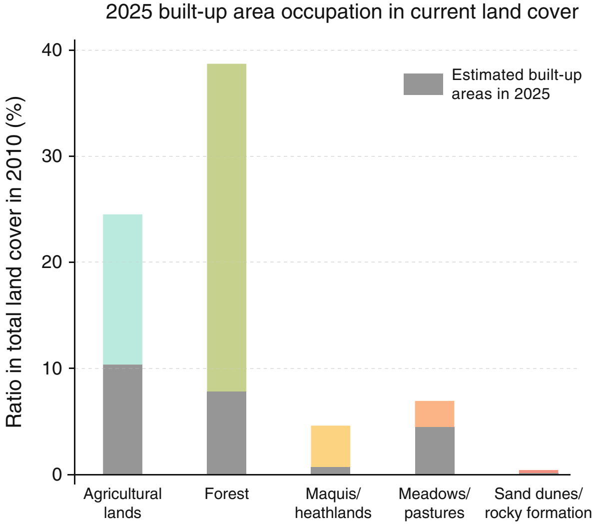

Fig. 16.5 The projected expansion of built-up areas by land cover category in 2025 assuming all speculated development projects are realized (see also Fig. 16.4b) (Prepared by and published with kind permission of ©Azime Tezer 2013. All Rights Reserved)

Table 16.3 Percentage of land cover of key habitats and built-up areas in İstanbul in 2010 and in 2025

Land use/cover category	Land use/cover ratio (%) 2010	2025	% of remaining key habitats in 2025
Built-up	23	46	–
Agriculture	24	14	58
Forest	39	31	80
Maquis-heathlands	5	4	86
Pastures-meadows	7	3	37
Sand dunes-rocky formations	0.28	0.22	78
Freshwater	2	2	100

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Postscript

Late May and early June of 2013 witnessed wide-spread public protests in İstanbul's Taksim Square area and Gezi Park. The protests were triggered by the heavy-handed response of the Central Government to objections by locals to the planned "development" of the park near Taksim Square.

The park is located at the heart of the most vibrant neighborhood of the city, has a diverse and rich cultural fabric, and is frequented by citizens and tourists from all walks of life. It is also one of the few green spaces in İstanbul's Central Business District (CBD). In yet another example of top-down decision-making (see Sect. 16.4), the Central Government had decided to virtually eradicate the park and instead turn the area into a built-up environment with a project that would include constructing a replica of a military barracks that was demolished some 70 years ago (Fuhrmann 2013; Occupytaksim 2013). The intention was supposedly to use the building mainly as a shopping mall and a residence-hotel-museum complex.

The city has already only 1 m^2 of green space per person within its central built-up area (Urban Age 2009). Therefore, there is no justification to replace the park with a replica of a building that has no particular historical or architectural importance, especially in the absence of sufficient supporting documentation of the original barracks to guide the reconstruction process. However, in spite of persistent objections from various stakeholders and planning professionals, the Central Government insisted to go ahead with the project (Yıldırım 2012; Docomomo Turkey 2013; ICOMOS 2013). There is widespread consensus among the public that this insistence reflects a number of disconcerting factors, one of the most important being the absence of proper public deliberation on planning decisions (Occupytaksim 2013).

On July 3, 2013, it was revealed that a court had actually cancelled the project back in June 6. The declaration of this ruling was apparently delayed for procedural reasons. The ruling, in principal, precludes the Central Government's earlier decision of holding a plebiscite on whether to go ahead with the planned project, which is highly contested. Still, what the future holds for Gezi Park remains to be seen. From a broader perspective, the massive protests in İstanbul may be a step to force the Metropolitan Municipality to adopt a governance style that is more transparent and more participatory. #direngezi

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Chapter 17
Local Assessment of Stockholm: Revisiting the Stockholm Urban Assessment

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Abstract In the year 2003, the Stockholm Urban Assessment (SUA) was selected as a sub-global assessment within the global Millennium Ecosystem Assessment (MA, Ecosystems and human well-being: synthesis. Island Press, Washington, DC, 2005). This chapter revisits SUA and fills in important knowledge gaps in the assessment as well as provides insights on urban resilience building. The chapter applies a critical perspective on the present urban development trajectory of the Stockholm metropolitan area. It emphasizes the need to understand ways in which informally managed green spaces contribute to ecological functions in urban settings. The chapter provides a background of the Stockholm region and the current challenges it faces, followed by a synthesis of the major insights conveyed in SUA related to informal ecosystem management. The chapter concludes by proposing policy recommendations of general implications for urban resilience building.

Keywords Sub-global assessment • Informal management • Urban growth • Keystone species • Stockholm

Key Findings

- Despite the political ambition to preserve the green structure in the Stockholm region, it is increasingly becoming fragmented by urban expansion; with some 50 % having disappeared from the most centrally located green areas since the mid 1970s.
- Wetland habitats have greatly declined in the area due to habitat loss and land-use change with loss of biotopes for amphibians, wetland birds, and insects.
- Hardwood deciduous forest, especially old oak forests (*Quercus robur* and *Quercus petrea*) has played a central role in the historical development of the cultural landscapes of the region for considerable time, representing important keystones for maintaining biodiversity.
- Local management practices, informal institutions and local ecological knowledge play a key role for sustaining habitats for wetland dependent organism groups, declining pollinator populations and insect-controlling birds. Informally managed land makes up a considerable part of the green structure in the Stockholm region.
- It is predicted that the average increase in temperature for Stockholm will be 4–6 °C by the year 2100. This will result in a longer plant season, with an increase of biomass. While oak woodlands and their associated flora and fauna is predicted to be promoted by temperature increase, research in the MA Stockholm Urban Assessment suggests that such prediction is highly uncertain.

17.1 Introduction

In the year 2003, Stockholm was selected as one of the sub-global *urban* assessments within the Millennium Ecosystem Assessment (MA 2005). This assessment was the Stockholm Urban Assessment (SUA) (Colding et al. 2003). When it ended in 2005, the research merely had begun and with results and insights being far from synthesized. Hence, this chapter revisits SUA and fills in important knowledge gaps in the assessment.

The SUA analytic framework has shaped the research in what has been referred to as the *Stockholm school of urban ecological research* in which knowledge generation of informally managed urban ecosystems is a key characteristic (Fig. 17.1). Informal management draws on local institutions (i.e., rules and norms) that tend to be created, communicated, and enforced outside of official government sanctioned channels (North 1990; Colding and Folke 2001).

The overall objectives of SUA in 2003 were to: (1) expand knowledge from the structure to the function of natural systems in greater metropolitan Stockholm; (2) to understand how knowledge of ecological processes and dynamics are incorporated into institutions; and (3) to assess the potential for learning and combining and making use of different types of knowledge systems.

To understand the ecological dynamics of different types of land use and what role informal institutions, local ecological knowledge, management practices and social networks play in the resilience building in Stockholm, the SUA-researchers initiated social-ecological inventories of a selective set of urban land use, including allotment gardens, residential gardens and cemeteries, golf courses, the social-ecological history of the National Urban Park as well as knowledge on protected areas (Fig. 17.2).

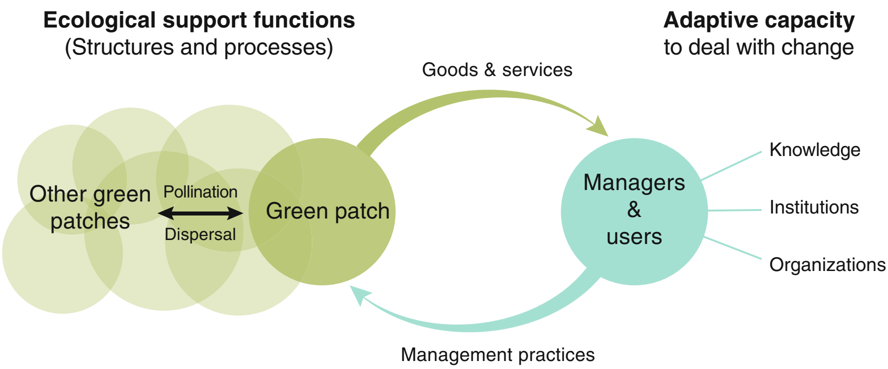

Fig. 17.1 The SUA analytical framework. The *left-hand* side depicts the ecological inventories made in SUA, involving studies of the ecological linkages of local green area patches. The *right-hand* side depicts relationships studied in the social-ecological inventories, including studies of informal and formal institutions (Modified from Colding et al. 2003, p. 8 and published with kind permission of ©Johan Colding 2003. All Rights Reserved)

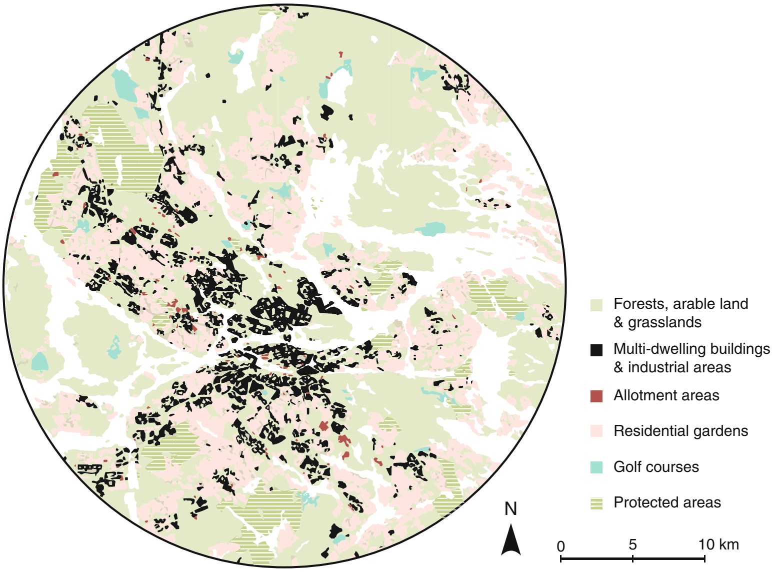

Fig. 17.2 The SUA study area and the spatial distribution of assessed land uses. The area has a radius of 20 km, representing 15 % of the total land area of Stockholm County (Modified from Colding et al. 2006, p. 239 and published with kind permission of ©The Royal Swedish Academy of Sciences/Elsevier 2006. All Rights Reserved)

The origins of the analytical approach used in SUA derive from institutional analyses of long-term resource management in small-scale, local communities (Berkes and Folke 1998; Berkes et al. 2003; Colding and Folke 2001). A similar analytic framework as used in SUA (Fig. 17.1) was developed in a project, entitled *Linking Social and Ecological Systems for Resilience and Sustainability,* where knowledge was developed from local cases that showed historic and successful adaptation to ecosystem resilience and which also unraveled management practices and social mechanisms with a capacity to cope with resource and ecosystem change (i.e., Berkes and Folke 1998; Berkes et al. 2003). To transfer the analytic framework to urban systems has been the most distinct trademark of SUA, with new insights generated on urban social-ecological systems and their dynamics. This chapter summarizes the key insights of SUA on informal urban ecosystem management.

17.2 The Stockholm Metropolitan Area

The Stockholm County represents 1.5 % of Sweden's land surface, constituting the most populated region with some 2,050,000 inhabitants or 21.5 % of the total population of Sweden (Statistics Sweden 2010). The area is the most rapidly growing in Sweden with urban densification (compaction) being identified as the most desirable urban development trajectory (RUFS 2010; Stadsbyggnadskontoret 2010). The area consists of a total land and water area of 6,785 km^2, extending some 180 km from north to south. Forty-six percent of the land area constitutes forests, 18 % agricultural lands, 14 % settled areas, and 22 % other land uses (Statistics Sweden 1998). Out of the 2,920 km^2 of forests, some 4,5 % is formally protected (Östlund and Lagerblad 2011).

The SUA study area makes up about 1,010 km^2 of the central parts of Stockholm County, referred to as the Stockholm metropolitan area (Fig. 17.2). The outer fringe area consists of a suburban-rural landscape that includes edge cities interspersed among agricultural lands and managed forests. The central part includes, among others, the National Urban Park, a 27 km^2 area, protected as a natural interest in law and representing a key study site in SUA in which several historical and social-ecological research assessments were conducted.

17.3 Key Characteristics and Challenges in Stockholm

17.3.1 Ecological Determinants and Their Changes

In a European perspective, the Stockholm region holds a considerable area with green structure. The Stockholm metropolitan area is situated in a fissured-valley landscape, with sediment-filled valleys, formerly agricultural fields and some wetlands, now harboring most of the routes for transportation and settlement. Between the valleys rises morain or bedrock heights, with the main of the green structure being mostly forests, but also former pastures. The green wedges constitute the nucleus of the green structure and together with large areas for recreation in the region's outskirts play an important role in the generation of ecosystem services. For example, about 40 % of the CO_2 generated by traffic and about 17 % of total anthropogenic CO_2 can potentially be accumulated by the green structure of Stockholm County (Jansson and Nohrstedt 2001). Despite the political ambition to preserve green wedges in regional planning, the wedges are becoming more and more fragmented by urban expansion (Fig. 17.3).

While it has not been possible to acquire figures regarding the loss of green structure in the most recent decades, urban growth resulted in 8 and 7 % of green structure loss in the 1970s and 1980s respectively. Red-listed species have declined since the mid 1970s with some 50 % (i.e., 223 red-listed species) having disappeared from the most centrally located green areas (Gothnier et al. 1999). Some red-listed species in this area constitute relic populations of the warmer Bronze Age period (Ekelund 2007). Today, some 1,080 species are classified as red-listed (Artdatabanken 2010), but also common species groups show a sharp drop in abundance, e.g., amphibians, reptiles and some bird species.

Wetlands, especially open fens and wet alder forests, have greatly declined in the area due to habitat loss and land-use change with loss of biotopes for amphibians, wetland birds, and insects.

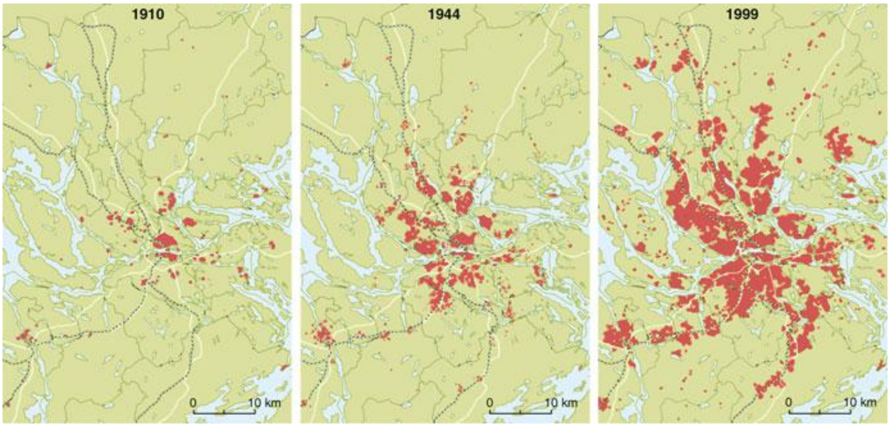

Fig. 17.3 Urban development pattern in Stockholm County. The figure displays (from *left* to *right*) urban growth in the Stockholm County from the years 1910, 1944, and 1999 respectively (Prepared by and published with kind permission of ©Jerker Lokrantz/Azote 2013. All Rights Reserved)

Hardwood deciduous forest and trees, especially old oak forests (*Quercus robur* and *Quercus petrea*) are considered to be a most valuable biotope for biodiversity. Oaks played a central role in the historical development of the cultural landscapes for a considerable time, especially in wooded pastures. This has resulted in one of the largest oak woodland areas around Lake Mälaren. Oaks constitute 18 % of all trees found in the National Urban Park, some of which are at least 500 years old (Hougner et al. 2006). Because the National Urban Park holds one of Europe's largest populations of giant oaks the park plays a critical role in the resilience building of oak forests from an international perspective, considering that the epidemic oak disease has led to a decline of oak forests over wide ranges in Europe (Führer 1998; Barklund 2002).

As a keystone species, oaks produce a unique set of niches for flora and fauna that depend on old hollow trees, hosting up to 1,500 species of insects, mosses, fungi and lichens and providing nesting and feeding sites for many birds and bats (Hougner et al. 2006). Of all red-listed insects in NUP, 80 % are linked to old growth oak trees and lime trees (Gothnier et al. 1999). Studies in SUA (Lundberg et al. 2008) revealed that natural regeneration of oaks depends on an intimate chain of ecological relationships with the Eurasian jay (*G. glandarius*) representing the key link in this chain (Fig. 17.4).

17.3.2 Effects from Climate Change on Biodiversity, Ecosystem Services, and Resilience

As the global average temperature rises due to climate change, it is likely that the climate zones will be relocated northwards, meaning that Stockholm could end up in a climate zone similar to the one of Berlin at an increase by 5 °C (Ekelund 2007).

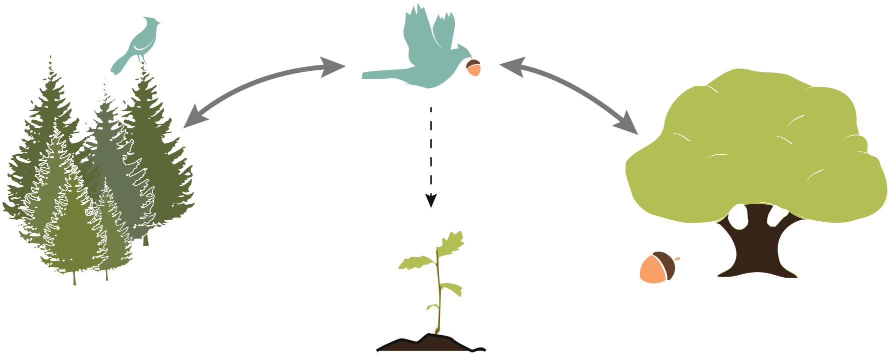

Fig. 17.4 The natural oak forest regeneration complex. This complex is necessary to consider in natural regeneration of the oak- dominated landscape in the National Urban Park. Oak forest regeneration depends on Siberian jays for dispersal and planting of acorns. In turn jays depend on the presence of dense coniferous forests for egg laying and for hiding offspring from predators. Hence, in order for successful natural oak forest regeneration in NUP, it is not only critical to preserve jay populations but also coniferous forest stands within or in close proximity to NUP (Modified from Hougner et al. 2006, p. 368 and published with kind permission of ©Ecological Economics 2006. All Rights Reserved)

It is predicted that the average increase in temperature for Stockholm County will be 4–6 °C by the year 2100 (Länsstyrelsen i Stockholms län 2011). This will result in a longer plant season, with an increase of biomass.

It is estimated that a longer plant season could have positive implications for agriculture and forestry since larger profits of the harvests are to be expected, unless plants and trees are stressed by an increased amount of diseases due to increased humidity and heat or drought during hotter summers (Ekelund 2007). Table 17.1 summarizes potential effects due to climate change until year 2100 as estimated in a recent survey by the Stockholm County Administrative Board (Östlund and Lagerblad 2011). While it is highly uncertain what the effects would be, certain species are likely to be favored with a change of climate while others will be disadvantaged. For example, fish species that depend on cold water in lakes and in the Baltic Sea will be disadvantaged (Ekelund 2007). Warmer summers also favor algae blooms in the Baltic Sea, Lake Mälaren and other lakes, with higher water temperature increasing the risk of growth of poisonous algae that could have a damaging influence on marine animals and plants.

A temperature increase is already taking place in the region with some species having changed their behavior. Warmer and earlier spring means that some migratory birds arrive earlier to the area than previously (Ekelund 2007). If the tree line is offset northwards the beech can grow in this area on a more permanent basis while spruce trees will be disadvantaged. The spruce will be exposed to much greater competition from deciduous trees than is the case today due to deciduous trees being favored by heat and that they can handle winter storms better. Looking ahead, there is a risk that the entire spruce ecosystem may disappear from Stockholm (Ekelund 2007).

Table 17.1 Analysis of predicted climate change in Stockholm County by year 2100 (based on Östlundoch and Lagerblad 2011)

- Average temperature increase of 4–6 °C
- Average precipitation will increase by 10–30 % (more during winter)
- Prolonged plant–growing season of 100–140 days
- Extreme precipitations more common
- Number of snow days is reduced by 65–100 days
- Flows in water bodies will increase greatly during winter, but be reduced during summer
- An increase of up to 1 meter of the Ocean surface until year 2100

Resulting consequences:

- Increased heat waves
- More favorable climatic conditions for mosquitos, tics, bacteria and mold
- Increased local problems of flooding related to precipitation and extreme rainfall
- Flooding from increased ocean surface level
- Increased risk for land slides and erosion, affecting built–up areas and infrastructure
- Increased risk for a decline of water quality
- Prolonged pollen season

Oak woodlands and their associated flora and fauna are, on the other hand, predicted to be promoted in a warmer climate, encouraging both growth rates and the dispersal of oaks (Ekelund 2007). Several of the red-listed insects and other invertebrates that depend on oak currently live at the edge of their northward climate zone. In recent assessments there is a clear increase of several threatened species due to a hotter summer climate (ibid). Hence, current populations of red-listed species associated with oaks are considered to function as source populations and could plausibly contribute to an increase of biodiversity in this area with warmer climate. However, the situation of potential species increase may in fact only be temporary, suggesting that a critical threshold for ecosystem compositional change has not yet been reached in the Stockholm area. As studies within SUA indicate, natural regeneration of oaks is carried out by the seed dispersal service performed by the Eurasian jay (Hougner et al. 2006; Lundberg et al. 2008). The jay depends on dense spruce tree stands to build its well-hidden nest in order to avoid predators (Fig. 17.4). In a future of a warmer climate, which could eventually cause the entire spruce ecosystem to disappear from the Stockholm region, the jay will likely be disadvantaged. Thus, the red-listed species that today are associated with oak woodland may indeed be under threat in the future because conditions for jays and the associated natural regeneration of oaks deteriorate with a rise in average temperature (Colding et al. 2013a).

The example of the Eurasian jay highlights the importance of taking ecosystem services into account in assessments of future climate impacts on biodiversity, to view ecosystems as moving targets that change over time (Holling and Sanderson 1996), and to realize that resilience building is very much about disclosing the relationships that determine critical thresholds in ecosystems.[1]

[1] As SUA indicated, studies of urban ecosystem services may also be fruitful for disclosing relevant and appropriate scales for management intervention (Janssson and Colding 2007), economic valuation (Jansson and Polasky 2010), and for detangling the intimate chain of ecological relationships that make up different functions in ecosystems (Hougner et al. 2006).

17.3.3 Population Increase

During the 1990s, there was an annual population increase of about 18,000 persons in Stockholm County. When SUA started in 2003 it was estimated that about two million inhabitants would live in the county by the year 2010 (Colding et al. 2003).[2] This prediction turned out to be very accurate; as of year 2010 the total population had increased to 2,054,343 (Statistics Sweden 2010); i.e., 11 % population increase. According to recent statistics, it is estimated that this trend will continue, meaning that the county likely will hold some 2,400,000 people in year 2030.

Besides the loss of ecosystems mainly due to a decrease in the area of cultivated lands for building and infrastructure development, several environmental effects are associated with population growth in the region. These include acidification due to airborne pollution; increased nitrogen eutrophication in forest, lakes and watercourses; eutrophication from phosphorus and nitrogen in the Stockholm archipelago; the drainage of open cultivated lands dominated by covered arable lands; and a decrease of wetland areas due to cultivation and settlements (Colding et al. 2003).

17.3.4 The Lack of Regional Planning of the Green Structure

A main goal of decision makers in the Stockholm region has been to make the region one of the world's leading development areas, and to promote international competitiveness, high and equal living conditions, and a long-term sustainable environment (RUFS 2010). This should be reached based on the regional strategies adopted by local and regional policy makers and planning authorities. These strategies include business development, education and research, housing and infrastructure development, and climate and energy adaptation (Kämpe 2011).

Physical regional planning was in the later part of the 1990s mainly geared at maintaining the capacity for economic growth in the region (Colding et al. 2003). While policy makers nowadays recognize the importance of addressing climate change, strategies to address this issue draw primarily from more advanced technological solutions, such as more efficient energy systems to reduce CO_2 emissions (Colding et al. 2013a). It is, however, increasingly clear that social and ecological systems truly are interconnected across spatial and temporal scales and therefore the physical urban environment needs to capture such integration in considerably new ways at the local levels of urban design and form (Barthel et al. 2010a).

Currently, it is extremely difficult to reach certain regional planning goals, such as to protect the green wedge system and to integrate environmental issues in the physical planning. One main reason is the system of self-governing local municipalities (RUFS 2010). Actions taken by one municipality affect adjacent municipalities' use of the green structure. Exploitation pressure of one municipality may sometimes be

[2] In year 2002, the population in the area was 1.849.200 (www.ab.lst.se).

so high that well-considered decisions of planning cannot be taken by one municipality alone (Colding et al. 2003). Thus, there is an expressed need for inter-municipal coordination to reach the goals of sustainable development for the region.

17.3.5 Formal Institutions and Biodiversity Management in Stockholm

Biodiversity management in Stockholm County holds a long tradition of being *formal* in character, setting aside valuable areas for nature conservation by the state and local municipalities. The proclamation of nature reserves and other protected areas has been the cornerstone in the preservation of species and ecosystems in Sweden. A number of formal institutions determine how green areas are used, managed and maintained that influence biodiversity. The Environmental Code and the Planning and Building Act – represent the two most important legal measures regulating biodiversity governance. The Environmental Code contains overall regulations with regard to how public interests are taken into account when government authorities and municipalities deal with cases of conflicting interests concerning the use of natural resources (Svensk Författningssamling 1998). The Planning and Building Act governs spatial planning and states that each municipality shall draw up an up-to-date Municipal Comprehensive Plan. The plan indicates where urban development is suitable. Such plans reflect future trends of land use in the study area and constitute an important tool in the analysis of trends and conditions. Furthermore, several international conventions influence biodiversity management, such as the Convention on Biological Diversity (CBD) and the European Union network, Natura 2000.

17.4 The Stockholm School Approach

17.4.1 Reconsideration of Formal Management

The SUA research highlighted that the present, formal governance system of biodiversity is fraught with several shortcomings. For one, formal measures, such as setting aside legally protected areas and other legislative measures for biodiversity conservation do not automatically lead to effective conservation (Colding and Folke 2001; Colding et al. 2003). For example, the protected areas in Stockholm constitute a patchwork quilt of ecosystems that do not match critical ecosystem interactions and dynamics, missing the important aspects of landscape connectivity (Borgström 2003; Colding et al. 2006; Ernstson et al. 2010, Löfvenhaft et al. 2002). One clear example of this was found in a nature reserve in southern Stockholm where the terrestrial and freshwater environments were managed separately with very limited communication across the areas of jurisdiction (Borgström 2009). This division of management effectively cuts the watershed into pieces and the scales of the areas hydrology are disregarded. It was also found that many formal institutions lack the flexibility to adapt to an ecosystem approach. This is reflected in that management of green areas is rigid and that there were more contacts between managers handling the same kind of area (e.g., cemeteries) than between neighbouring green space managers, implying a neglect of plausible spatial ecological connections such as species migration routes (Borgström 2003).

One of the shortcomings of protected areas is that it is often financially costly to manage such areas (Berkes 1996; Horowitz 1998; Colding et al. 2006). For example, in the London region, parts of the protected green belt have become severely degraded due to lack of money partitioned for management (Greater London Authority 2001). A resulting consequence of lack in management funding is a 'separation of attributes' of green areas like public parks. This entails that the rights to green space habitats often become separated due to congestion and lack of management (Lee and Webster 2006). In Stockholm city, for example, there are several instances of public parks having become degraded due to underfunding. In conjunction with restoration of these parks, local government agencies often open up for several types of private establishments, such as cafés, amusement areas, etc.; hence, parts of these parks become privately enclosed, often resulting in green-area loss (Colding 2011).

17.4.2 Informally Managed Ecosystems in Stockholm

One important concluding insight of SUA is that the present formal biodiversity governance system in combination with high pressure for available urban land run the risk of overlooking the ecological functions and the social potential that local stewardship groups play in the management of urban green space. Lessons from the work of local communities show that local management practices, informal institutions and local ecological knowledge play an important role for sustaining local ecosystems and natural resources (Ostrom 1990; Berkes and Folke 1998; Berkes et al. 2003). Informal institutions involve rules and norms that tend to be created, communicated, and enforced outside of officially government sanctioned channels (North 1990). Such locally developed institutions represent an example of local self-organization around ecosystem management and have the potential to reduce transaction costs related to management of ecosystems since they draw on the commitment and self-interest of the local stakeholders involved in such management (Colding and Folke 2001).

By adopting informal institutions in ecosystem management, a greater number of people and/or organizations also become stewards of land. Colding et al. (2006) refer to such stewardship groups as *green-area user groups*, denoting users and landholders that manage land individually or in cooperative form, for example in associations, clubs or similar organizational units. Hence, informal institutions that work for conservation may potentially provide benefits that can be capitalized on in urban ecosystem management designs, especially in cases where financial constraints make biodiversity conservation ineffective. In contrast to formal governance, green-area user groups rely on a wide array of informal institutions and draw on local ecological knowledge in management activities. When land managed by such groups are taken into account in urban landscapes, the actual coverage of green space is often considerably greater than what is normally presented in official land estimates. Colding et al. (2006) found based on the calculation of garden area of two real-estate classes registered as 'low-building' and 'part-time' (summer houses), that they on average consisted of 83.6 % garden habitats (i.e. all natural areas minus buildings and impermeable surfaces of a real estate). Allotment gardens, residential gardens and golf courses represent informally managed land in the Stockholm metropolitan area. These lands were found to cover as much as 18 % of the total land area which represents well over twice the area covered by protected areas and over half of the urban land demarcated as green wedges (Colding et al. 2006). However, in Stockholm there has been a lack of official recognition of what value these land uses hold for the generation and maintenance of biodiversity and associated ecosystem services. In the following sections, these functions are more specifically elaborated upon.

17.4.3 Urban Garden Habitats and Ecosystem Services

As of year 2006, there were 128 allotment gardens in the Stockholm metropolitan area, covering 0.3 % of the total land area, and ranging in size between 3,450 and 70,000 m^2 (Colding et al. 2006; Andersson et al. 2007). Assessments in SUA showed that such informally managed lands promote the generation of critical ecosystem services, hence, work in complementary ways with protected areas and other natural land to support biodiversity. For example, both allotment and domestic gardens were found to be valuable habitats for native pollinators, representing an important functional group for sustaining flora and food production in this area (Colding et al. 2006; Andersson et al. 2007). While allotment areas only covered a tiny portion of the land area, their role in providing high-quality habitat for the inner-city urban core was estimated to be significant due to their rich abundance of flowering plants and due to a prolonged season for nectar supply facilitated by active gardening. Investigated bumblebee diversity ranged from 5 to 11 species, with 8 species being the average number found in a typical allotment area in Stockholm (Colding et al. 2006; Ahrné et al. 2009). Moreover, physically isolated allotment areas in the highly developed urban matrix were found to be functionally connected by native bees (Fig. 17.5). Networks of small habitats have been found to sustain considerable pollinator diversity (Cane 2001).

Moreover, Andersson et al. (2007) found that allotment gardens had higher pollinator abundance than formally managed public parks and cemeteries, and held a different community structure of seed dispersing and insectivorous birds. Many gardeners cultivate some flowers with the only intention being to feed pollinators and many

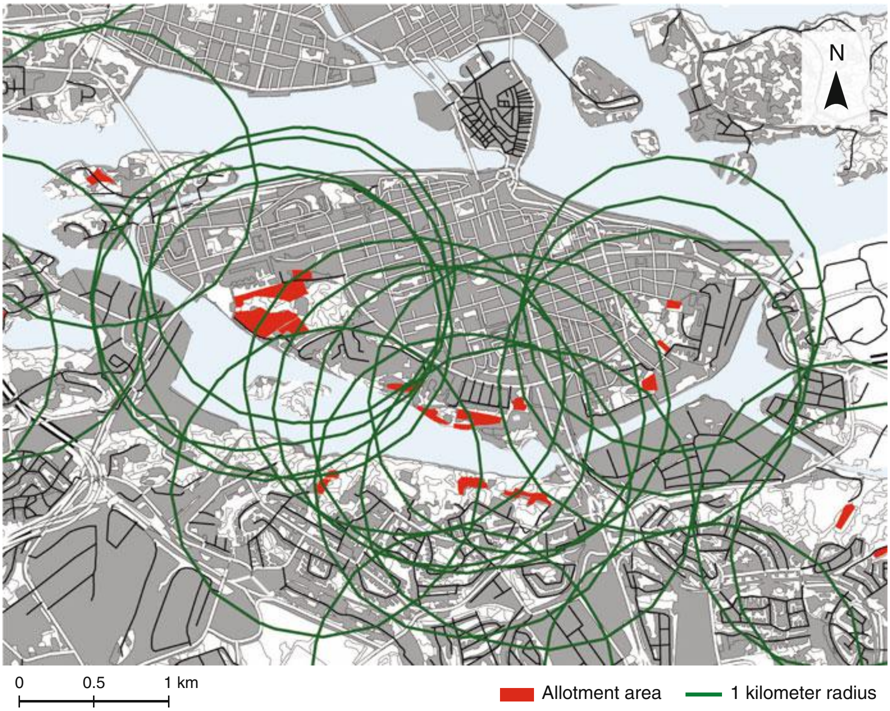

Fig. 17.5 A network of allotment areas in Stockholm city. The allotments displayed in the figure can be considered functionally connected by invertebrate metapopulations. City parks, cemeteries, and other inner city green areas may benefit from pollination by the insects found in allotment areas, especially bees, since foraging distance between allotments and other areas is small. The *circles* around allotment areas have a 1.0-km radius, i.e. within the foraging range of most bumblebees (Modified from Colding et al. 2006, p. 241 and published with kind permission of ©The Royal Swedish Academy of Sciences 2006. All Rights Reserved)

improve nesting opportunities for wild bees. These informal institutions support the abundance of wild bees and thus the ecosystem service of pollination (Andersson et al. 2007), not only within allotment gardens, but also over large areas of the urban landscape (Osborne et al. 1999; Greenleaf et al. 2007). The enhanced pollination service feeds back to the gardeners, since pollination underlies the generative capacity of flowers, fruits and many vegetables, which are the prime concern for gardeners.

Allotment gardeners were also found to hold considerably more knowledge about ecological dynamics than staff responsible for maintenance and management of public parks and cemeteries (Andersson et al. 2007). This includes knowledge about interactions between organisms, the interplay between organisms and site-specific abiotic conditions, as well as about ecological processes. Barthel et al. (2010b) refer to allotment gardens as communities-of-practice for environmental learning among participants that involve acquisition, transmission, and modification of ecological practices and local ecological knowledge (Wenger 1998). In the study by Barthel et al. (2010b), oral communication was found to be the most important means of transmission of local ecological knowledge and practices in allotment gardening with 57 % of the respondents reported to learn about garden management through daily talks with other gardeners. In comparison, 18 % learn about gardening by talking with external experts. Newcomers tap into the community-of-practice primarily through conversations with experienced neighbors, and through teaching by appointed mentors.

Domestic gardens often cover substantial tracts of land in city-regions (Jeffcote 1993; Gaston et al. 2005). In the Stockholm metropolitan area, domestic gardens are mainly found at some 5–7 km from the city center, and can be characterized as suburban green patches, covering quite extensive and cohesive chunks of green space, sometimes located in direct adjacency to nature reserves (Colding et al. 2006). In many parts of Stockholm, domestic garden habitats provide cohesive green belt structures, promoting the dispersal of organisms between ecosystems in the urban matrix. In the Stockholm metropolitan area, 16 % of the land area is managed as private, domestic gardens (Colding et al. 2006). In a recent study of breeding bird diversity in three types of residential housing developments in Stockholm, a total of 36 bird species, representing 14 families, were found. The bird communities were dominated by a number of species generally found in all housing types at relatively high abundances, and included both neo-tropical insectivores and an overall high diversity of insectivores (Andersson and Colding in review).

17.4.4 Golf Courses and Wetland Species

The Stockholm metropolitan area holds 24 golf courses, comprising 1.4 % of the total land area and 2.1 % of the green wedge structure. One fifth of these are located partially within or adjacent to nature reserves. These golf courses are constructed on former arable land and heavily regulated in environmental legislation regarding chemical inputs (Colding et al. 2009). In terms of area, a median-sized golf course of 57 ha is comprised of quite large natural areas, considering that roughly 70 % of a golf course represents non-playable areas with semi-natural vegetation of trees and grasslands. Hence, some 40 ha of a typical golf course in the Stockholm area consist of varying “natural” habitats. In comparison, a median-sized nature reserve in this area is 77 ha (Colding et al. 2006).

In an ecological inventory of the wetland fauna in ponds of golf courses, a total of 71 macroinvertebrate species were found (Colding et al. 2009). There were no significant difference between golf course ponds and off-course ponds (outside the golf course) either at the species, genus or family levels. Golf course ponds held a more homogenous species composition than ponds in nature-protected areas and ponds in residential parkland, according to a within-group similarities test. A total of 11 species of odonates (i.e., dragonflies and damselflies) were identified, including the red-listed large white-faced darter dragonfly (*Leucorrhinia pectoralis*). Although anuran occurrence did not differ between golf course ponds and off-course ponds, the great crested newt (*Triturus cristatus*) was significantly associated with golf course ponds. Among the taxa of conservation concern found on golf courses, the four amphibian species are nationally protected in Sweden. Golf courses provided over a quarter of all available permanent, freshwater ponds in the Stockholm metropolitan area, which the GIS results revealed (Colding et al. 2009).

17.4.5 The Role of Social Networks in Informal Urban Ecosystem Management

In a historical and social inventory of the National Urban Park a total of 69 interest groups were identified as being involved in the use of the park (Barthel et al. 2005). Of these, 25 represented green-area user groups that had a direct role in managing habitats within the park, contributing to sustaining ecosystem services such as seed dispersal and pollination. Results of this study suggest that incentives should be created to widen the current biodiversity management paradigm in the Stockholm metropolitan area, and actively engage local stewardship associations in adaptive co-management processes of the park and surrounding green spaces.

Ernstson et al. (2008) found that social networks also play a role in the protection of urban ecosystems by constructing "protective stories" that have political influence and facilitate collective action, as well as transfer and sustain knowledge related to the politics of space. A network among 62 civil society organizations created a core-periphery structure between conservation groups in the core, and user groups in the periphery to protect the National Urban Park. This made it possible to stop larger-scale exploitation plans in the park, and an almost day-to-day monitoring by user groups active in the park landscapes to hinder smaller-scale exploitations. Furthermore, through making use of scientific reports on habitats and dispersal patterns, and reports of cultural values, organizations could articulate in popular discourse how the park sustained holistic values, creating a "protective story" for the National Urban Park.

17.5 Lessons for Urban Resilience Building

17.5.1 Informal Institutions and Management

As shown by the studies in SUA, informal ecosystem management in Stockholm contributes to sustaining habitats for wetland dependent organism groups, declining pollinator populations, insect-controlling birds as well as making up a considerable part of the urban green structure in greater metropolitan Stockholm. International studies indicate that urban garden habitats hold a rich flora of plants, including rare and threatened species (Maurer et al. 2000), as well as high numbers of invertebrates regardless if garden plants are native or alien (Colding et al. 2006). Some birds constitute effective pest-regulators on agricultural cultivars (Colding 2007). Garden habitats that attract insect-controlling birds could contribute in the resilience building of cities by buffering undesirable effects from climate change as a rise in temperature is expected to provide a more favorable climate for mosquitoes, tics, and noxious insects.

Garden habitats should to a greater extent be considered in the resilience building of declining pollinator populations. For example, the designation of residential areas with gardens could ideally be located within a close range from crop fields in order to promote pollination of cultivars, as areas with semi-natural vegetation of herbs and grasses are decreasing in the arable landscape (Colding 2007).

As SUA indicated, informally managed habitats on golf courses contribute in supporting biodiversity in the Stockholm metropolitan area. This result is supported by other international studies. Colding and Folke (2009) determined in a synthesis study that the ecological value of golf courses at an international level was significantly higher in comparison to other types of green-area land use, holding higher ecological value in 64 % of comparative cases. They found that the ecological value significantly increases with land that has high levels of anthropogenic impact, e.g., agricultural and urban lands. Results from the study revealed that golf courses represent a promising measure for restoring and enhancing biodiversity in ecologically simplified landscapes. They hold a real potential to be designed and managed to promote critical ecosystem services, like pollination and natural pest control, providing an opportunity for joint collaboration among conservation, restoration and recreational interests. A promising multifunctional planning tool with potential to integrate golf courses and other types of urban land use for promoting ecosystem services is *ecological land-use complementation* (Fig. 17.6).

Harnessing the diverse social-ecological knowledge complexes of green-area user groups could contribute to the resilience building of cities by helping people survive in times of economic crises (Colding and Barthel 2013). For example, allotment gardening has been shown to play a critical role in retaining and transmitting collective memories of how to grow food in urban settings and how to manage regulatory and supporting ecosystem services (Barthel et al. 2010b; Colding and Barthel 2013). The combined means by which knowledge, experience and practice of informal ecosystem management are captured, stored, revived and transmitted over time among allotment gardeners have been referred to as *social-ecological memory* by Barthel et al. (2010b).

17.5.2 Cognitive Resilience Building

While legal frameworks and formal managers are important in urban ecosystem management, it is important to recognize what role green-area user groups hold in generating ecosystem services. A social network perspective (Ernstson et al. 2008) could be helpful in identifying and mapping such groups as well as finding ideas of how to organize larger-scale urban ecosystem governance.

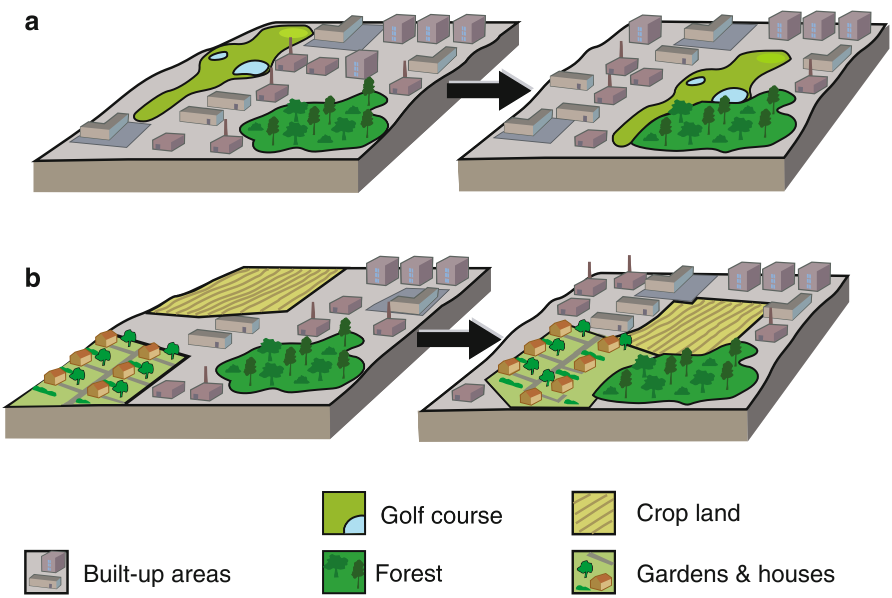

Fig. 17.6 **ELC** draws on the notion that proximity among habitats is important in order to promote ecological dynamics in urban settings, and that species in a patch are likely more affected by the qualitative characteristics of adjacent patches than by those of more distant parts in the urban landscape (Colding 2007). Hence, ELC represents a multifunctional land-use planning framework that describes how different types of land use could synergistically interact to support ecosystem services and the promotion of 'response diversity' in urban settings. Response diversity means that different organisms within a functional group respond differently to diverse types and frequencies of disturbance (Colding et al. 2003). In (**a**) a golf course with ponds with no forest patches could serve as suitable breeding-habitats for amphibians when located adjacent to a forest habitat due to landscape complementation. Similarly, in (**b**) when urban gardens are clustered adjacent to forest patches and crop fields, pollinators may be promoted. Different pollinators may 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. In this case, such a configuration could promote 'response diversity' to environmental stresses among pollinators. Accordingly, ELC- structures may promote ecosystem processes that are not provided for when land use is located as a single, isolated unit (Modified from Colding 2007, p. 50 and published with kind permission of ©Johan Colding/Landscape and Urban Planning 2007. All Rights Reserved)

A take-home lesson from SUA is that the nurturing of resilience in urban landscapes depends, among others, on crafting spatial planning processes that better can integrate local stakeholders, their practices and informal institutions at different scales in the urban landscape. This insight parallels the notion of *collaborative* planning (Healey 2007) that has demonstrated that increased participation with civil society has the potential to mobilize collective action to achieve strategic urban governance targets (Neuvel and van der Knaap 2010; Tezer 2008; Ernstson et al. 2008; Colding and Barthel 2013; Colding et al. 2013b).

While adaptive co-management (Folke et al. 2003) was suggested as a viable approach at the start of SUA in 2003 as a complement to formal ecosystem management, such designs have in practice been hard to develop in the Stockholm region. Instead, ecosystem services should become more intimately built into urban form and design in order for city inhabitants to play a more active role in ecosystem management (Barthel et al. 2010a; Marcus and Colding 2011; Colding et al. 2013a). Integrating ecosystem services in urban form and design is important not only in order to make cities more resilient to climate change, it is equally critical to make the links between natural and human systems more visible in urban settings in recognition of that the man-nature dichotomy has lead to an increasing 'environmental generational amnesia' among city dwellers (Miller 2005), and resulted in 'extinction-of-experience' of nature in many cities (Pyle 1978).

As suggested by the SUA-research, the incorporation of a broader set of civic society in ecosystem management may provide several advantages, including economic benefits (Colding et al. 2006) and wider ecological learning among urban citizens (Barthel et al. 2010b). Moreover, it holds potential to promote *cognitive resilience building* in cities, referring to "the mental processes of human perception, memory and reasoning that people acquire from interacting frequently with local ecosystems, shaping peoples' experiences, world views, and values towards local ecosystems and ultimately towards the biosphere" (Colding and Barthel 2013).

17.5.3 Property-Rights Arrangements

It is important to recognize that designs for management of ecosystem services depend on a diversity of property-rights systems. Institutional research show that no single type of property rights regimes (i.e., state, private, and common property rights systems) can be prescribed as a remedy for resource overuse or environmental degradation; rather, policy should focus on establishing a multitude of property rights regimes that are designed to fit the cultural, economic, and geographic context in which they are to function (Hanna 1998). As recent SUA studies indicate, diversity of property rights regimes in cities promote diversity in management of urban land as well as access to land in cities (e.g., Colding 2011; Colding and Barthel 2013). Considering that today's institutions poorly match current changes in ecosystems (MA 2005; Folke et al. 2007), property rights arrangements hold potential to play a much greater part in the resilience building of urban landscapes than has hitherto been the case. The current global shift to private property rights in contemporary cities is therefore a worrying sign (Webster 2003; Lee and Webster 2006). As many cities lack the financial means to adequately withhold and manage public lands, privatization becomes a viable option. Common property managed urban ecosystems may in the near future offer an alternative to the alienation of public lands in cities, with potential to promote cognitive resilience building in cities and to integrate a greater set of local stakeholders in ecosystem management (Colding and Barthel 2013).

17.6 Conclusions and Policy Recommendations

The Stockholm region holds quite a rich amount of green structure. However, the region is facing a steady fast population increase. As the green structure of the region is successively decreasing, the resilience of the region may likewise decrease. While a critical threshold for ecosystem compositional change has not yet been reached, it may be achieved in the near future due to a rise in temperature. As indicated in this chapter, the formal governance system of biodiversity is fraught with several limitations, with the system of self-governing local municipalities making it awkward to reach the goals of regional sustainable development. Currently, more advanced technological solutions are promoted as a policy tool to reach climate regulation targets. A paramount objective of current SUA research is to raise awareness among planners and policy makers about what role the Stockholm green structure plays in terms of generating ecosystem services that in turn play a key role for climate adaptation (Colding et al. 2013a). This scientific mission is an important resilience building strategy in itself and an example of how science may influence policy developments in positive directions.

As this chapter also has highlighted, ecosystem management in the Stockholm region has a long tradition of being *formal* in character, conducted along municipal governance lines and being backed up by legal institutional frameworks. However, and as revealed here, informal management of ecosystems in this area is quite substantial and contributes to the ecological values that the region currently possesses. Nevertheless, informal management is seldom translated into informal *governance* in urban settings. One reason may be that informally managed ecosystems like allotment gardens, golf courses, and domestic gardens, lie outside the immediate control and management requirements of local government and administrative authorities (Gaston and Thompson 2002). This situation may arguably lead to local self-organization around ecosystem management being hampered. A prime example of the opposite can be found in the city of Berlin, where city planners and local decision makers have a long tradition of integrating different types of green-area user groups in co-governance of urban space (Bendt et al. 2013). Creating and nurturing a diversity of property-rights arrangements in cities represents a challenge with considerable potential for promoting self-organization around informal ecosystem management. Researchers, planners and local policy makers should more fully explore and harness this potential to broaden urban ecosystem governance in cities. As this chapter has shown, formal governance of biodiversity are not enough to safeguard biodiversity. In cities, biodiversity – the basis for *ecosystem services* – is also actively nurtured by way of informal institutions and management practices of diverse green-area user groups and social networks.

Based on the lessons conveyed in SUA, this chapter concludes by postulating the following general insights for promoting resilience in urban settings:

- The capacity of urban ecosystems to produce ecosystem services depends on a mix of formal and informal institutions

- Informal management of urban green space may increasingly become instructive in the challenge to mitigate loss of ecosystem services and for building resilience against undesirable effects from climate change
- Strive towards a shift away from a focus on biodiversity conservation to one integrating *ecosystem services* in urban form and design
- Create multifunctional urban social-ecological systems by way of ecological land-use complementation.
- Plan for support and integration of social networks and diverse interest groups in urban ecosystem management
- Strive to increase institutional diversity (e.g., property rights) in the governance of urban ecosystem services as a way to promote self-organizing informal ecosystem management
- Strive for collaborative urban planning and design that can integrate knowledgeable stakeholders at various urban scales to support continuous learning, management and stewardship of ecosystems and their services
- Devote more research to untangle the intricate chains of the generation of urban ecosystem services

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Chapter 18
Local Assessment of Chicago: From Wild Chicago to Chicago Wilderness – Chicago's Ecological Setting and Recent Efforts to Protect and Restore Nature in the Region

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Abstract From the time of its charter in 1832 the population of the City of Chicago grew explosively and the landscapes of the region were largely transformed both by the expanding physical footprint of the city and by the extensive development of agriculture in the hinterlands. This transformation was at the expense of highly biodiverse ecosystems that had been inhabited by populations of indigenous peoples who had themselves been agents in the historical development of the region's biota. As a consequence of both public and private community planning early in the history of the city, the region retained substantial open space in the city itself and its hinterlands. In this chapter we describe the factors that determined the structure of the biota of Chicago and review recent large-scale attempts to manage the biodiversity of the region. We discuss recent biodiversity conservation strategies mainly through the lens of Chicago Wilderness, a regional biodiversity conservation alliance that emerged over a decade ago and that now has more than 260 institutional members. These members represent federal, state, and local agencies, public land-management agencies, conservation organizations, and scientific and cultural institutions. Despite the progress we show that the footprint of the city continues to grow and that there is significant work to be done even on questions of the basic natural history of the Chicago area.

Keywords Urbanization • Biodiversity • Long-term transformation • Conservation initiatives

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Key Findings

- Chicago is the third largest city in the United States with a population of 2.7 million people
- The landscape in and around the city was influenced by indigenous population for centuries but has been radically transformed over the past century and a half by settler populations
- Enlightened planning at the turn of the twentieth century has resulted in the setting aside of considerable amounts of open space
- Chicago Wilderness is a regional biodiversity conservation alliance with over 260 institutional members representing federal, state, and local agencies, public land-management agencies, conservation organization, and scientific and cultural institutions
- The continued expansion of medium and high density housing in the Chicago area will intensify the need for effective biodiversity conservation.

18.1 Introduction

With a population of 2.7 million, Chicago is the largest city in the US Midwest and the third largest in the United States. The greater Metropolitan Statistical Area (MSA) to which Chicago belongs has a population of almost 9.5 million. The radical and rapid transformation of the landscape that has occurred over the past century and a half in order to accommodate a burgeoning population, might suggest that Chicago is not a promising place to undertake large-scale conservation efforts. However, the region supports conservation programs that have received local, national, and international recognition.

In this chapter we discuss the factors that shaped the biodiversity of the Chicago region and evaluate the conservation significance of these ecological systems in their current state. We start with a vignette that describes Midwestern landscapes in the years just before the emergence of Chicago, and inspect the way in which these systems were rapidly transformed from the middle nineteenth century to the present day. We then turn our focus to the governance and management of Chicago's ecosystems, with a particular focus on the work of Chicago Wilderness, a regional biodiversity conservation alliance that emerged over a decade ago and that now has more than 260 institutional members.

Even in the mid-nineteenth century, vast stretches of Midwestern natural landscape persisted in the Chicago region. It is against the record of the natural extravagance that predated the city that current efforts to preserve and restore are oftentimes assessed. Noting that these landscapes persisted up until only a century and a half ago is a reminder of how rapidly the landscape of this region was transformed to today's thriving city. We start the discussion by in the following sections examine those ecological factors that shaped the ecological systems of the Midwest before, and then discuss the anthropologically introduced stresses that resulted in the rapid transformation of the region.

18.2 Ecological History of the Chicago Region

18.2.1 An Ecological Vignette of Early Chicago: Colonel Colbee Benton's Sleepless Night

On August 19th 1833 Colonel Colbee Chamberlain Benton (1805–1880) left Chicago with Louis Ouilmette, a young man of French and Potawatomi heritage, to inform local Indian tribes that their federal annuities would be paid in September of that year. Benton's trip, recorded in *A visitor to Chicago in the Indian Days: Journal of the Far-Off West*, was taken 1 year after the end of the Black Hawk war which ended most tribal resistance to white settlement of the Chicago area. That same year the Potawatomis, a tribe that dominated in the lands that became Chicago since the 1690s, relinquished their rights to their lands in Illinois. At that time the white settler population was little more than 150 people. A few years later in 1837 Chicago was chartered as a city.

That Benton's journey was undertaken at time of tension between the indigenous and settler population is reflected in his descriptions of their trip. On the night of August 24th the pair of travelers passed through some oak groves and arrived at a small stream in a little prairie in Southeast Wisconsin and they camped there for the night. As night fell they heard Indians around their camp. Benton hid beside a large tree and at Ouilmette's suggestion he removed his straw hat since it was "a good mark to shoot at." Assessing the danger they found themselves in, Louis remarked that "there were occasionally some of the Sauks and Fox Indians wandering about in [that] part of the country, and from them [they] could not expect much mercy."

Benton didn't sleep that night. However, even if they had been "in danger of suffering from the power of their tomahawk and scalping knives" it was not fear that kept him awake. He remarked, in fact, there was something about their circumstances "so novel and romantic about it that it dispelled every fear…" He was far from home, everything looked "wild and terrible", he was surrounded by "savages", and yet it all seemed "lovely and romantic and beautiful". He felt happy.

So what kept Benton from his sleep? It was the noise! Some of the noise certainly may have emanated from the Indians who "mocked almost every wild animal." But also there were unfamiliar birds calling, as well as foxes and raccoons. In the distance, wolves howled and the owls hooted in concert with the wolves. The mosquitoes added their part to "the music". A sleepless, noisy, vaguely threatening night, and yet Benton declared that never before had he "passed a night so interestingly and so pleasantly…"

***

So here was Chicago around the time of its charter and slightly afterwards: a settler population which numbered in the hundreds, surrounded by a loud chorusing of people and wildlife. Prairies that stretched for over a hundred miles, and wildlife including gray wolf, bison, black bear and perhaps up to ten other mammal species that would disappear by the early years of the twentieth century. Benton was just one of the many early writers who explicitly recorded the diversity of the vegetated landscape of northeast Illinois and southeast Wisconsin as they traveled through it. Near Round Lake (Lake County, Illinois) Benton noted that he and Ouilmette ventured through little oak openings then out onto the prairie, walked alongside little streams with "heavy timber", and, very muddily, crossed "tremendous marshes". The prairie grasses were, as they often are described in these early accounts, so tall and wet that passing through on horseback was like "wading through water." Although the prairie was often likened to an ocean, undulating and free, the dominant metric for its depth was a man on a horse. Benton and Ouilmette shot, usually unsuccessfully, at any birds they could see: wild geese, ducks, loons, pigeons, a sand crane (successfully bagged), and a prairie hen (killed and roasted for the dog). Streams were home to "some monstrous pickerel and other large fishes." Dotted infrequently through this wilderness were the cornfields of Indians. Thus it was a variegated landscape supporting a rich diversity of life, human and non-human. A gloriously loud landscape it was then, one interesting and uncanny enough to keep a man awake and happy.

***

We present the encounter between Benton and Ouilmette and the native peoples in the vignette above to illustrate a turning point in the history of natural ecosystems in the Chicago Wilderness region. The encounter also represents an encounter between two social systems and not merely the individuals representing them. This is theme we will discuss later in this overview, but here we simply note that the "settler" and "pre-settler" social systems differed profoundly on issues such as their conception of nature, land ownership, land management and so on. Both systems has implications for the biodiversity or the region, though the social system of the settlers and the density of individuals associated with it has had inarguably a more rapid and extensive impact on the biota of the Chicago Wilderness.

In what follows we briefly describe the ecological history of the Chicago region. We use this description as a background to our account below of efforts to sustain the area's distinctive biodiversity. More details on the ecological history of the region can be found in Heneghan et al. (2012).

18.2.2 Ecological Development of Chicago's Ecosystems

Lake Michigan and the other Great Lakes formed as a result of the Wisconsin glacial advancement and retreat 16,000 years ago. The advance and retreat of the ice deposited gravel, sand, silt, clay and rocky debris throughout the region. The composition of soils and their drainage, a result of glaciation, have significantly shaped the Chicago region's biodiversity.

Climatic shifts have also influenced the successional development of the region's biodiversity. The present climate of the region is continental, with winters characterized by periodic incursion of cold Arctic air and at least two or three major storm systems resulting in significant snow accumulation. Average temperatures in January are typically below 0 °C. Because of the relative flatness of the terrain, wind-chill effects can be significant. Summers are dominated by warm humid air originating from the Gulf of Mexico, with summer temperatures averaging above 27 °C. Temperatures in all seasons are also influenced by the proximity of Lake Michigan, second most voluminous of the Great Lakes, which produces a so-called lake effect, resulting in cooler temperature nearer the lake in summer and warmer breezes during the cold season (at least when the lake is not frozen). Precipitation totals 86 cm a year on average, most of it falling as rain in the summer months (Greenberg 2002).

Considerable attention has been paid to reconstructing the post-glacial history of Illinois (King 1981; Baker et al. 1989; Nelson et al. 2006). The initial tundra-like post-glacial vegetation was briefly replaced by spruce (*Picea*), which in turn was replaced by deciduous trees as temperatures increased. Temperatures and precipitation vacillated for several thousands of years, and vegetation responded with conifers and deciduous trees alternatively dominating. The landscape configuration familiar to contemporary observers, characterized by a patchwork of woodlands, prairie and wetlands, emerged about 8,500 BP. Although these patterns remained highly dynamic, xeric oak-hickory forest dominated in the immediate Chicago region (Northern Illinois). In the last several centuries the region has experienced cooling and xeric trends alternating with warming and more humid periods. In the years before the large-scale clearing of vegetation associated with the establishment and growth of Chicago, a warming trend increased the prevalence of deciduous vegetation.

The role of fire considered in the context of edaphic and climatic variability in configuring the landscape and maintaining disturbance-dependent habitats across northeast Illinois has been contested among academic ecologists over the course of the last 100 years. Even by the 1930s, when Edgar Nelson Transeau wrote about the factors influencing the origins, development and maintenance of the Midwestern prairies, he could outline several competing hypotheses already extensively debated in the literature (Transeau 1935); for instance, prairies as "scars" persisting after the ecological conditions producing them had terminated but maintained by human intervention; prairies as persisting because of unfavorable soil conditions ("immature soils"); prairies as the "pyrogenic victory of Indians and pre-Indians" who maintained the prairies as pasture and hunting ground. To this list one can add the role of large grazers, especially bison, in maintaining prairie vegetation (Anderson 2006). Contemporary opinion is that the mixture of prairie, savanna, and forest vegetation in the Chicago region, the "vegetation mosaic", is influenced by both climate and fire (Anderson 2006). Research on the use of fire as a means of maintaining this mosaic has been prevalent since the 1960s. Although the use of prescribed fire as a management tool is generally understood and accepted by the public in the region, nevertheless successful implementation requires negotiation with the local community.

18.3 Urbanization and the Current State of Regional Biodiversity

18.3.1 Chicago Emerges

The suitability of lands southwest of Lake Michigan for the growth of an urban center is attributable to many of the same factors that influence the region's ecological communities. The lakes and waterways provide an abundant supply of freshwater, the young post-glacial soils are fertile, and there is an abundant supply of accessible resources, including significant supplies of timber and mineral ores from Wisconsin and Michigan. The early colonization of the region by European settlers was influenced by the region's proximity to a continental divide that provided portage between the Great Lakes and the Mississippi River and put Chicago at an important crossroads. Furthermore, Chicago is roughly located midway between pole and equator (coordinates 41°52′55″N 87°37′40″W) and its continental climates ensure relatively long and productive growing seasons. Despite the many ecological benefits, historian William Cronon (1992) points out, that the precise location of the young city had numerous shortcomings primarily associated with the marshiness of ground close to the lake, which required the raising of the city in its early years to prevent streets from becoming water-logged due to frequent floods.

After its founding in 1832, Chicago's population growth was unprecedented. By 1890 it had become the third US city to have a population of 1,000,000 (Encyclopedia of Chicago 2004). In 1900 it was the second most-populous city in the US. After 1900 the growth slowed but by this time there had been a major transformation of the region's landscapes. The exceptional climatic and edaphic favorableness of the Midwest for agriculture, combined with explosive population growth, resulted in rapid transformation not only of lands proximate to the metropolitan areas, but of entire biomes adjacent to the city. Of the estimated 8.9 million hectares of prairie originally in Illinois, 930 ha remain – a decline of 99.9 % (Steinauer and Collins 1996). In less than a century most of the natural landscape had been ceded to domestic and industrial use in the city, and to agriculture use of the land in the hinterlands. Around the end of the nineteenth century there was growing recognition that some of the natural heritage of the region should be retained.

Public and private community planners in Chicago who were dedicated to making the city a "good" place to live developed programs to retain substantial open space in the young city and its hinterlands (Abbott 2004). The Plan of Chicago in 1909 (the so-called Burnham Plan) is the most widely known culmination of such early efforts to ensure "that the city may be made an efficient instrument for providing all its people with the best possible conditions of living" (from the Plan of Chicago quoted in Smith) (Smith 2006). A central proposal of the plan was the "improvement" of the lake front by the construction of a shoreline parkway and the creation from largely undeveloped lands of the 1.3 km-long Grant Park. The plan also envisioned an outer park system, and made provisions for a system of widened streets and avenues. The majority of the open space set aside by planning efforts, however, was maintained as parks, often with formal gardens rather than representative remnants or examples of pre-settlement habitat.

In contrast to the parks, and more consequential for the conservation of the pre-settlement landscape was the creation of a system of forest preserves and conservation districts in the early years of the twentieth century. There are now 62,240 ha of land in this system across Chicago and surrounding counties (Packard 2004). The purpose of this system, as proclaimed in the 1913 act that created them, has an explicit conservation focus – the land was to be acquired “for the purposes of protecting and preserving the flora, fauna and scenic beauties” and, furthermore, “to restore, restock, protect and preserve the natural forest and said lands together with their flora and fauna, as nearly as may be, in their natural state and condition, for the purposes of the education, pleasure, and recreation of the public”.[1] Although the various county forest preserves represent substantial tracts of land, and a few contain good examples of the original landscape, very little is regarded as “exceptional quality” habitat (Packard 2004). Grazing, timber removal, fire suppression and other influences have resulted in a rapid shift of these landscapes from the ecological state at the time they were placed under protection. Indeed, land that was acquired and set aside a century or more ago has only relatively recently been managed specifically to protect rare elements of the biotic communities, oftentimes with a view to restoring elements of the pre-settlement landscape. Although the composition and structure of biotic communities of the region have been, as we have seen, in dynamic flux since the end of glaciation, there has been very considerable and greatly accelerated change in recent decades with consequent losses of much of the flora and fauna the preserves were established to protect. Since contemporary conservationists and land managers regard most of the land as being highly degraded, managers have been attempting to restore some of these lands to re-establish vegetation characteristic of the landscape that the early European settlers encountered.

18.3.2 Ongoing Urbanization

In order to illustrate in a concrete manner recent changes in the landscapes of the Chicago region, and to speculate about projected changes in the short-term future we have analyzed historical and projected patterns of housing density. The physical footprint of domiciles can illustrate how landscapes are transformed. To depict the magnitude of change in housing density in the Chicago region, we have mapped the housing unit density in the Chicago and its hinterlands in 1950 and

[1] See more details at: http://fpdcc.com/about/history

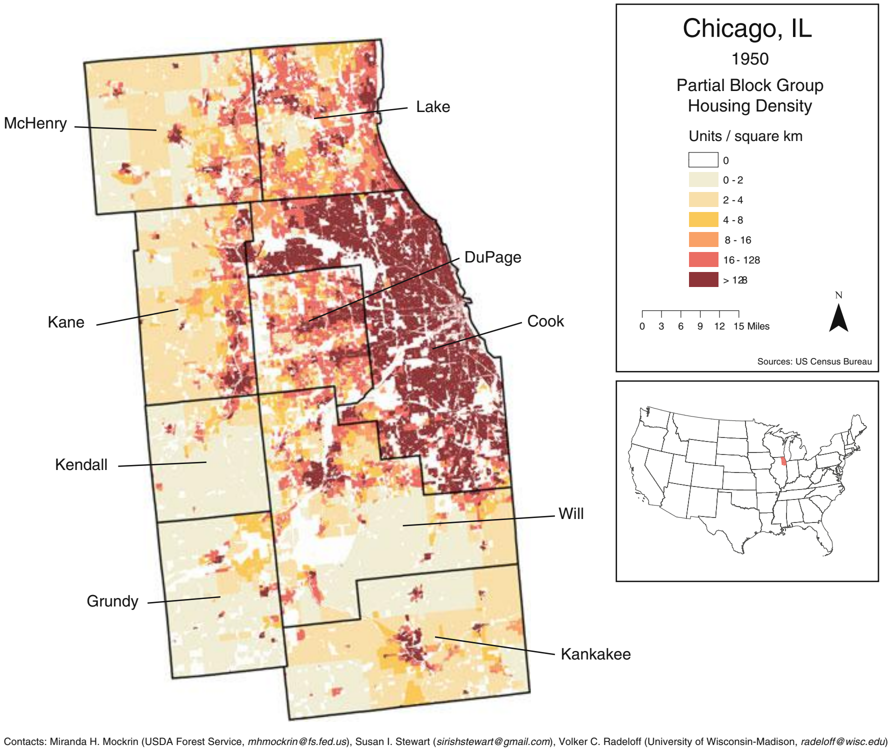

Fig. 18.1 Housing density in Chicago 1950. Housing backcasts and forecasts for the greater Chicago region were produced from 2000 Census housing and ancillary data at the partial block group scale using Bayesian simulation methods (Data source: US Consensus Bureau. Prepared by David Helmers and modified by Jerker Lokrantz/Azote. Published with kind permission of ©David Helmers 2013. All Rights Reserved)

2000, and have projected the expected housing density in 2050[2] (Figs. 18.1, 18.2 and 18.3). Housing backcasts and forecasts for the Chicago Wilderness region were produced from 2000 Census housing and ancillary data at the partial block group (PBG) scale, using Bayesian simulation methods. The use of ancillary data and Bayesian modeling is required because county level housing data, though readily available, lack the spatial detail required for understanding landscape-level social-ecological processes. Simulated future housing distributions employ Woods and Poole's econometric forecast for the US county population.[3] Housing forecasts combine the Woods and Poole population projections, current county-specific

[2] This important issue of the clash between social systems and its consequences for the lands in discussed in details by William Cronon (1983) Changes in the Land: Indians, Colonists, and the Ecology of New England. Hill and Wang, New York.

[3] http://www.woodsandpoole.com/

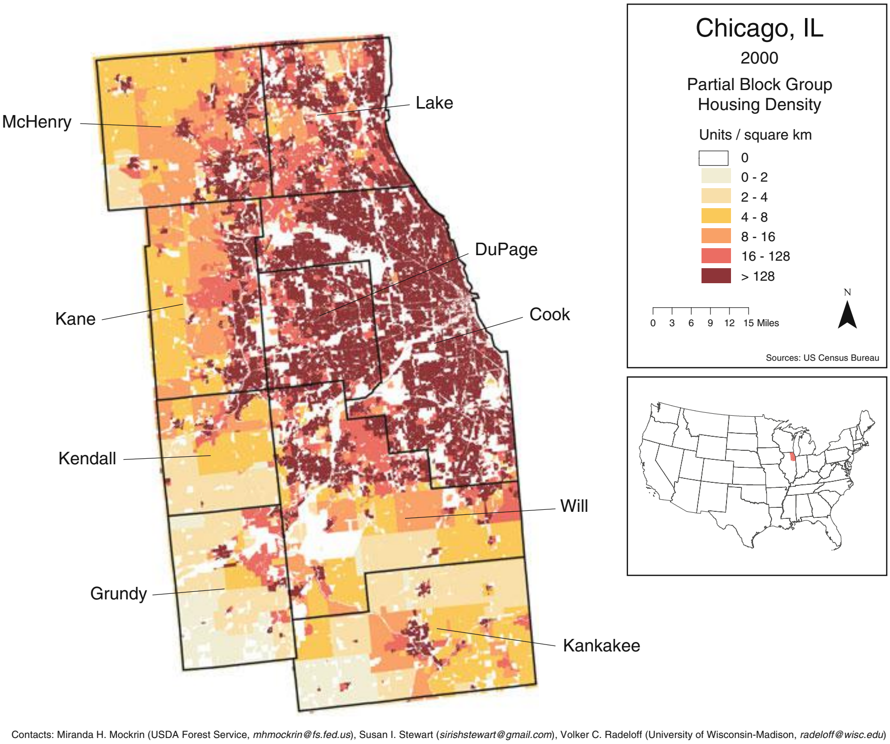

Fig. 18.2 Housing density in Chicago in 2000 (Data source: US Consensus Bureau. Prepared by David Helmers and modified by Jerker Lokrantz/Azote. Published with kind permission of ©David Helmers 2013. All Rights Reserved)

population-to-housing ratios, and the historical trends of housing growth or decline for each PBG in the county to estimate and distribute decadal changes. Our backcasts rely upon census data responses to the query: "in what year was this housing unit built?" Since responses do not account for those housing units that were destroyed or demolished, we therefore compare the sum of these PBG-level estimates to the county-level housing unit totals, then allocate the difference across the PGBs, proportional to the estimated count. Bayesian inferences are made iteratively to generate a range of estimates for both backcasts and forecasts. We have mapped the mean estimates.

The number of housing units almost doubled from 1950 to 2000 rising from 1.6 to 3.1 million. Between 2000 and 2050 the expectation is for nearly another 30 %.

Housing growth projections of the greater Chicago area show a steady expansion of medium and high density housing and loss of low density housing, predicting that housing density across the nine county region will reach 16 housing units per km^2 or higher, with exceptions limited to just southern Kankakee County, southwestern Grundy County, and northwestern McHenry County.

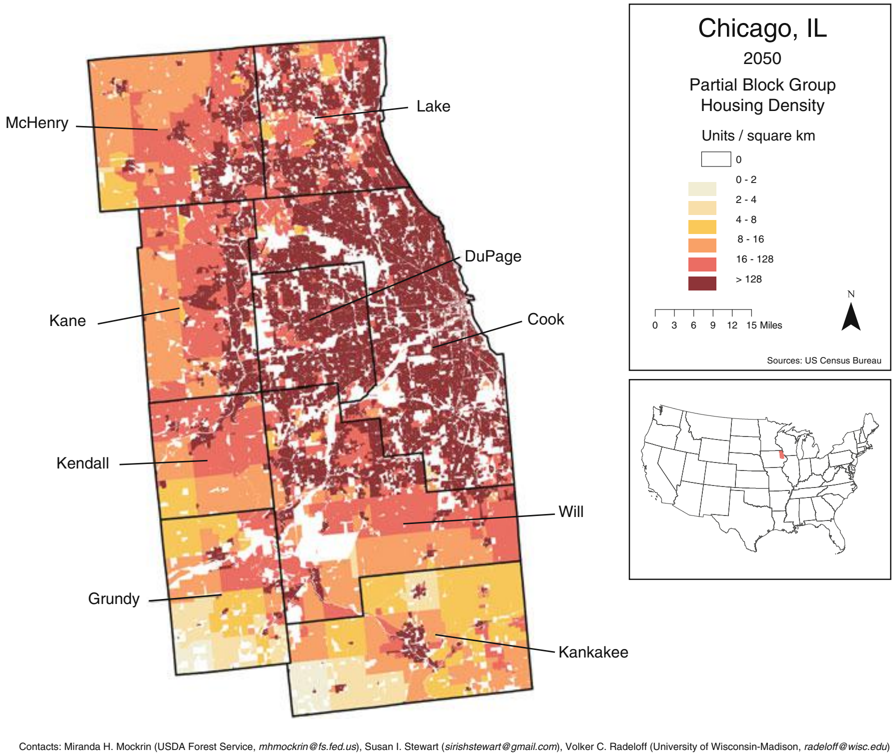

Fig. 18.3 Expected housing density in Chicago 2050 (Data source: US Consensus Bureau. Prepared by David Helmers and modified by Jerker Lokrantz/Azote. Published with kind permission of ©David Helmers 2013. All Rights Reserved)

18.4 Current Conservation Status of Ecological Communities of the Chicago Wilderness Region

The Chicago Wilderness classification scheme recognizes seven different terrestrial community types: forest, savanna, shrubland, prairie, wetland, cliff, and lakeshore communities (Chicago Region Biodiversity Council 1999). Each community type is finely subdivided; several sub-communities are recognized by the Nature Conservancy as critically imperiled globally. These include dry-mesic, mesic, and wet-mesic fine-textured soil savanna; dry-mesic fine-textured soil shrublands; wet-mesic woodlands; and wet-mesic sand shrublands. Many other sub-communities, including types of prairie, are classified in the Nature Conservancy's next most significant conservation category, imperiled globally. In addition to these endangered plant communities, the region also hosts animal assemblages of conservation significance – in fact, most rare plant communities have bird, reptile, amphibian and invertebrate assemblages of concern. Additionally, there are several rare mammal species targeted for conservation, including Franklin's ground squirrel, *Poliocitellus franklinii*.

Although there are extensive protected open lands throughout the Chicago Wilderness region (over 120,000 ha), the rarer community types are scarce. The Illinois Natural Areas Inventory identified only 4,200 ha of land with significant natural characteristics throughout the entire state (White 1978), which represents just seven-hundredths of one percent of the total land and water area of Illinois (The Chicago Wilderness Consortium 2006).

A recent report on the state of natural lands in the Chicago region concluded that the majority of the remaining natural areas surrounding Chicago are not healthy compared with the pre-settlement state of the region (The Chicago Wilderness Consortium 2006). Reasonably well-characterized stressors, such as fragmentation associated with urban development, invasion by non-native species, overabundant deer populations, modified hydrological conditions, and fire suppression, have contributed to the decline in the quality of the region's natural plant communities and animal assemblages – and continue to threaten them.

In the course of reviewing the current status of biodiversity in the Chicago Wilderness region, we noted that there had been very few attempts to estimate the number of extant species in each of the major taxa. Those estimates we found are compiled in Table 18.1. To get a more complete view we asked several regional experts on other taxa to provide additional information. These estimates and their sources are also included in Table 18.1.

18.5 Governing the Chicago Wilderness

We discuss the governance of those open lands set aside for biodiversity protection primarily through the lens of Chicago Wilderness. This is in recognition of the fact that this consortium has institutional members spanning federal, state, and local agencies, public land-management agencies, conservation organizations, and scientific and cultural institutions. Though there is no single governance structure for the 150,000 ha of open space considered to be Chicago Wilderness region and though partners do not relinquish autonomy, nonetheless institutional participants in Chicago Wilderness endorse a shared vision. The four priorities of the consortium are entitled greening infrastructure, leave no child inside, restoring nature and climate action.

18.5.1 Emergence of Chicago Wilderness as a Shared Governance Vision

Chicago Wilderness builds on the pioneering influences of architects, planners, and ecologists whose efforts eventually led to the establishment of the Forest Preserve District of Cook County in 1914. A number of additional factors contributed to the development of the alliance. Chicago gained some prominence, starting in the 1960s

Table 18.1 Number of species from regularly monitored taxa in the Chicago Wilderness region[a]

	Total	Native	Non-native
Plants	2,968	1,829	1,139
Macro-fungi	1,100		
Mammals	50	47	3
Birds	423	_	_
Fishes	164	146	18
Reptiles and amphibia	60	_	_
Butterflies	100		
Insects	20,000+	_	_
Molluscs	41	38	3
Earthworms	12	0	12

Plants: Swink, F and Wilhelm, G (1994) Plants of the Chicago Region. Indiana Academy of Science; Personal communication G Wilhelm (Conservation Design Forum) (2013). Fungi: Personal communication, Greg Mueller (Chicago Botanic Garden) (2013). Dr Mueller suggested that there are at least 20,000 fungus species in the region (in addition that is to the 1,100 macro-fungi above. Mammals: Greenberg, J A (2002) Natural History of the Chicago Region and http://www.mammalsociety.org/mammals-illinois. The three non-native mammals in the region are Norway rat (*Rattus norvegicus*), Black rat (*Rattus rattus*), and the house mouse (*Mus musculus*). Birds: Personal correspondence: Judy Pollock (Audubon Chicago Region) Geoffrey A. Williamson, Doug Stotz (The Field Museum) and Sheryl DeVore (2013). Fishes: Personal communication: Philip Willink (The Field Museum) (2013); Reptiles and Amphibia: Greenberg, J (2002) A Natural History of the Chicago Region. Karen Glennemeier (Audubon Chicago Region) counts 12 species of frogs and toads in this number. Butterflies: Personal Correspondence: Doug Taron (Peggy Notebaert Museum). Insects: The number of species here is an approximation made the Illinois Department of Natural Resources. Since Chicago has a high concentration of natural area remnants relative the state of Illinois, there is a probability that it will have many of the states species. However, the number is possibly a low approximation of that total species tally, as many of insect groups are poorly known. See: http://www.dnr.state.il.us/publications/pdf/00000679.pdf. Molluscs: Barghusen, L; Bland, J.; Klocek, R, (2010), A Field Guide to the Freshwater Mussels of the Chicago Wilderness, Field Museum of Natural History, http://fm2.fieldmuseum.org/plant-guides/guide_pdfs/386.pdf. Earthworms: Personal correspondence, Kristen Ross (University of Illinois, Chicago), Lauren Umek (Northwestern University), and Basil Iannone (University of Illinois, Chicago)

[a]In many cases the data is for Chicago area is defined as the 22 counties that surround Chicago, including 11 in Illinois, 7 in Indiana, 3 in Wisconsin, and 1 in Michigan

and 1970s, in the field of restoration ecology as some of the region's first prairie restorations were worked on at the Morton Arboretum in Lisle, Illinois, and on the grounds of the Fermi National Accelerator Laboratory in Batavia, Illinois. Also, at this time, a burgeoning movement of volunteer-led land stewardship was gaining momentum through the efforts of volunteer groups along the North Branch of the Chicago River (Stevens 1996). A widening segment of the general public also began to take note of local restoration efforts, and several conservation leaders saw the need to coordinate conservation and restoration activities on a regional scale.

In February 1993 representatives from 13 conservation agencies and non-profits gathered to explore a possible partnership to address biodiversity conservation needs across the Chicago metropolitan landscape (Ross 1997). This initial conversation included federal and state agencies, county forest preserve districts, and non-profit organizations that seemingly recognized that collaboration and synergy would improve the management of the land. The directors of these agencies and organizations crafted a Memorandum of Understanding and formed the alliance's four teams: Science, Land Management (now called Restoring Nature), Education, and Policy and Planning (now called Sustainability). Chicago Wilderness was publicly launched in April 1996 with an informal network of 34 founding organizations comprised of 8 federal agencies, 6 county forest preserve and conservation districts, 2 state agencies, 4 regional and local agencies, and 14 non-profit organizations. At the same time, the alliance announced the initiation of 28 regional biodiversity conservation projects due to a $700,000 grant from the US Forest Service (Ross 1997). Today the alliance is comprised of 262 organizations. The geography of Chicago Wilderness has expanded as well. Originally based on a much smaller region defined by nine counties (six in Illinois, two in Indiana, and one in Wisconsin), the current region is biogeographically based, spans parts of four states encompasses 34 counties, and includes more than 1,460 km^2 of protected open space. Currently the work of the alliance is organized around four core strategic initiatives.

18.5.1.1 Greening Infrastructure

Developed in 2004, the Green Infrastructure Vision (GIV) is a map-based representation of the goals of the Chicago Wilderness Biodiversity Recovery Plan. The GIV identifies over 1.8 million acres of Recommended Resource Protection Areas that surround, and/or connect the already protected core areas (1,460 km^2). The GIV serves as a macro-scale guide to focus land and water preservation and sustainable land-use practices. Implementing the GIV is a coordinated effort involving all alliance members in targeted community engagement. Since the first version of the GIV in 2004 it has been updated and refined. The updated vision (GIV 2.0) covers the seven-county northeastern Illinois metropolitan area.

18.5.1.2 Leave No Child Inside

The Chicago Wilderness Leave No Child Inside initiative seeks to reconnect the region's residents, in particular children and their caregivers, with the natural world. The initiative does this through public outreach and awareness efforts, and by working with Chicago Wilderness member organizations to increase nature-based programming and experiential opportunities.

18.5.1.3 Restoring Nature

Ecological restoration and management is a significant component of the work of many Chicago Wilderness members. Within this initiative, Chicago Wilderness is working to identify and advance regional goals and strategic actions related to the preservation, restoration, and/or management of natural plant and animal communities; establish opportunities to promote the exchange of information on best-management practices; facilitate the implementation of regional-scale restoration and management projects; and identify and secure restoration and management resources for the Chicago Wilderness region.

18.5.1.4 Climate Action

Recognizing the potential for climate change to jeopardize the conservation community's collective investments in the region, Chicago Wilderness developed its Climate Action Plan for Nature in 2010 to guide the alliance's work in preparing for and mitigating the impacts of climate change on regional biodiversity. The Climate Action Plan for Nature was the first plan in the Great Lakes region to specifically focus on climate impacts to biodiversity, and it identifies goals and broad strategies in the areas of adaptation, mitigation and education. A main goal of this plan is to update the Chicago Wilderness Biodiversity Recovery Plan from a climate change perspective. This effort was completed in 2012, and the Climate Change Update (climate.chicagowilderness.org) represents two and half years of collaborative work with over 100 regional practitioners, researchers and scientists to translate climate science into on-the-ground action that can be taken to help the region's natural areas be more resilient in the face of climate change. The Climate Change Update includes information on expected impacts to biodiversity as well as place-based adaptation strategies.

18.5.2 Biodiversity Recovery and Ecosystem Services

A foundational document for Chicago Wilderness is the Biodiversity Recovery Plan, an assessment developed in the years following the consortium's formation and which has guided the work in subsequent years. The goal of the Chicago Wilderness Recovery Plan "is to protect the natural communities of the Chicago region and to restore them to long-term viability, in order to enrich the quality of life of its citizens and to contribute to the preservation of global biodiversity" (Chicago Wilderness 1999). To emphasize: the purpose of protecting and restoring is both for the well-being of the region's human population, as well as being an effort on behalf of global conservation – for people and for the sake of the rest of nature. The Recovery Plan proceeds to present the case for the conservation and the proposed management of the region's biodiversity in both of these categories. The provisioning of ecosystem services is presented in the plan as a value derived from nature.

Though the discussion of the values of biodiversity conservation described in the Recovery Plan is generic, it does include some striking local examples of the types of ecosystem services derived from the protection of ecosystems. For example, it cites the cost of flooding on the Des Plaines River for local governments and property owners to be $20 million per annum, and associates this cost with the loss of wetlands, which would otherwise ameliorate some of this flooding. Similarly the loss of habitat due to urbanization of the region arguably necessitates the Metropolitan Water Reclamation District's multi-billion dollar construction of the Tunnel and Reservoir Plan (TARP), known as the Deep Tunnel, the proposed solution to flooding in the Chicago area. Although the Biodiversity Recovery Plan reiterates many of the well-known arguments for conserving biodiversity, there are, however, two key components worth stressing: (1) the Biodiversity Recovery Plan was a relatively early adopter of "ecosystem services" as a valuable framework in which to promote large-scale conservation efforts; and (2) the distinction between the different motivations promoting conservation has led recently to research attempting to evaluate the trade-offs and synergies in using ecosystem services or species protection as a guide for management planning. These diverse ecological, social and economic values, as articulated in the foundational documents of Chicago Wilderness, are central to the activities of the alliance; (3) The Biodiversity Recovery Plan is regional in scope; and finally (4) it emerged as a collaborative effort by local, state and federal agencies and by a range of non-governmental organizations, and research and educational institutions and universities.

18.6 Concluding Remarks

That significant biodiversity protection occurs in Chicago, a city of 2.7 million residents, is a consequence of the region's climate and its evolutionary and ecological history. It is also the result of decisions made by people both before and after the settlement of the region, by European and other non-indigenous populations. These decisions resulted in land protected from development and/or maintained to preserve the characteristic biodiversity of the area.

When the contemporary situation in Chicago is compared against the description of the region's natural heritage immediately prior to European settlement, the differences are stark and from a conservation perspective seem somewhat discouraging. One can barely walk for a mile (1.6 km) across tallgrass prairie in Illinois compared to the possibility of a 150 mile trek along the Grand Prairie back in the nineteenth century. That being said, the landscapes of both eras each represent social-ecological systems – in the pre-settlement case the human agents involved being primarily indigenous Native American populations, more recently highly populous and diverse urban populations dominate. Thus, both then and now human decision-making played a role in shaping natural components of the region.

Journalist Charles Mann in his assessment of the impact of Native American peoples on the America *1491: New Revelations of the Americas Before Columbus*, concluded: "Native Americans ran the continent as they saw fit. Modern nations must do the same." Now, we might quibble with the rather enormous license that this offers; nevertheless, the statement underscores the role of human agency in shaping ecological landscapes (second nature, (Cronon 1992)), both before and after the emergence of the great urban centers. The emergence of a conservation ethic, one that contrasts with the more cavalier attitude of early settler populations in the Chicago region, and one that informs the work of present day biodiversity conservationists and that inspires the work of Chicago Wilderness, should be seen as a remarkably positive development. We may not recover the losses of species, communities and ecological processes that have extirpated from the region; nonetheless it may be that we develop quite new social ecological systems. These new systems will undoubtedly be represented by highly cyborgian landscapes emerging from mixtures of technology and forces beyond the immediate ken of humans – systems that are hopeful, biodiverse, and resilient in the face of both ongoing anthropogenic disturbances and future human influences on the nature to which we undeniably belong and from which we futilely seek to escape.

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Chapter 19
Local Assessment of New York City: Biodiversity, Green Space, and Ecosystem Services

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Abstract New York City (NYC) is a complex megacity with a diverse human population situated in a constellation of estuarine and terrestrial ecosystems. With over 20 million people in the metropolitan region, NYC nevertheless has rich biodiversity that provides a broad suite of ecosystem services, with new assessments of biodiversity underway in city government, NGOs, and universities. NYC contains the most parkland of any U.S. city and currently about 21 % of the city is covered by tree canopy. PlaNYC, the environmental and economic sustainability plan for NYC, set a goal of planting one million new trees by 2017, with the hope that canopy cover will reach 30 % by 2030. However, significant challenges to local and regional biodiversity remain, including pollution, climate change, sea level rise, stormwater management, and human population growth. Nonetheless, NYC has made progress in improving the environmental quality of its urban ecosystems and in the provisioning of a broad range of urban ecosystem services. Three elements are key to this progress: (1) coherent governmental support in the form of an overarching long-term planning document, PlaNYC and the NYC Green Infrastructure Plan; (2) systematic investment in natural areas, green infrastructure and civic engagement by a rich variety of organizations; and (3) a commitment to the acquisition of data that facilitate informed decision-making.

Keywords Biodiversity • Ecosystem services • Green infrastructure • PlaNYC

Key Findings

- New York's population has grown 2.1 % between 2000 and 2010, with an expected increase to nine million residents by 2030
- Urbanization and development continue to put pressure on natural landscapes within the city
- PlaNYC, a long-term economic and environmental sustainability plan covering many aspects of natural resources, will guide urban planning and development for the near future

(continued)

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Key Findings (continued)

- Conservation and restoration initiatives include MillionTreesNYC, a plan to plant one million trees in the city, the Green Infrastructure Plan, which will invest over US$2 Billion in green infrastructure improvements, and disaster resilience planning
- In 2012 the Natural Areas Conservancy was established as a public-private partnership to contribute to comprehensive knowledge-based stewardship of New York's ecological areas.

19.1 Introduction

New York City: History

Founded in 1624 as "New Amsterdam"
Renamed New York in 1664
The population was 230,000 in 1790
The capital of the United States, 1795–1780
The world's first megacity, with a population of ten million in 1950

New York City (NYC) became the world's first global megacity in 1950 when its population reached ten million (Chandler 1987) and still ranks as one of the world's largest megacities with 22.2 million people living in the metropolitan region (U.S. Census Bureau 2010). NYC is both a complex social-ecological system and one of the world's great cultural and economic centers.

NYC's founding humans have extensively altered the landscape (Sanderson and Brown 2007). The watershed in which NYC exists was almost entirely forested in 1609 with small areas in agricultural cultivation by Native Americans. By 1880, approximately 68 % of the watershed had been converted to farmland, but as soil productivity declined and industry created new jobs, much of cleared land gradually reverted to secondary forest. The local rivers and streams were widely dammed for agriculture, milling, fishing, power, and drinking water.

Inspired by Burnham's plan for Chicago, the NYC Regional Plan Association created the world's second urban plan. NYC was an early leader in urban park development in the mid-1800s, including Central Park and Prospect Park, designed by the famous landscape architect, Frederick Law Olmstead. In the 1950s and 1960s, pioneering New Yorkers and others began outlining ways to encourage healthier, cleaner, and more sustainable modes of living. NYC owes its current sustainability vision to the foundations laid by William Whyte's *The Exploding Metropolis*, Jane Jacob's

The Death and Life of Great American Cities, and Ian McHarg's *Design with Nature*, all of whom used NYC as their laboratory for articulating the goals of livable and sustainable cities (McPhearson 2011).

19.2 The New York City Social-Ecological System

(Note that NYC will refer henceforth to the municipality)

New York City: Population

22.2 million residents in the metropolitan area.
8.3 million residents within municipal boundaries
10,630 residents/km^2
800 languages spoken
Regional GNP: US$1.4 trillion

Sources: Mackun and Wilson 2010; US Census Bureau 2011; Roberts 2010; Hoehn et al. 2009

NYC is the most populous and dense city of all U.S. municipalities (Mackun and Wilson 2010). The density of the city is matched by its diversity. 36 % of the city's population is foreign-born (Lobo and Salvo 2004) and NYC continues to be the leading gateway for immigrants to the U.S. (Monger and Yankay 2011). Over 800 languages are spoken in NYC, the most linguistically diverse city in the world (Roberts 2010). NYC's continued growth is supported by an energetic economy; the area's GDP in 2010 was approximately US$1.4 trillion (Greyhill Advisors n.d.), the largest regional economy in the U.S. and the second largest city economy in the world (Hoehn et al. 2009).

NYC lies at the confluence of several waterways that form one of the world's largest natural harbors, which is used extensively for import and export (Kurlansky 2006). NYC has a humid continental climate and summers are typically hot and humid with a July average of 24.7 °C.

NYC has the most urban parkland of any U.S. city (The Trust for Public Land 2011) (Fig. 19.1) and NYC's Central Park is the most visited city park in the U.S. (37–38 million visits per year) (Central Park Conservancy 2011), more than seven times as many visits as Grand Canyon National Park.

New York City: Area and Parkland

NYC comprises five boroughs: Manhattan, Brooklyn, Bronx, Queens, Staten Island
NYC's total area is 1,215 km^2

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35 % of this area is water
11,736 ha of municipal parkland
23 km of public beaches
Central Park in Manhattan covers 357 ha
Including federal land, NYC has the most urban parkland of any U.S. city

Sources: Roberts 2008; The Trust for Public Land 2011

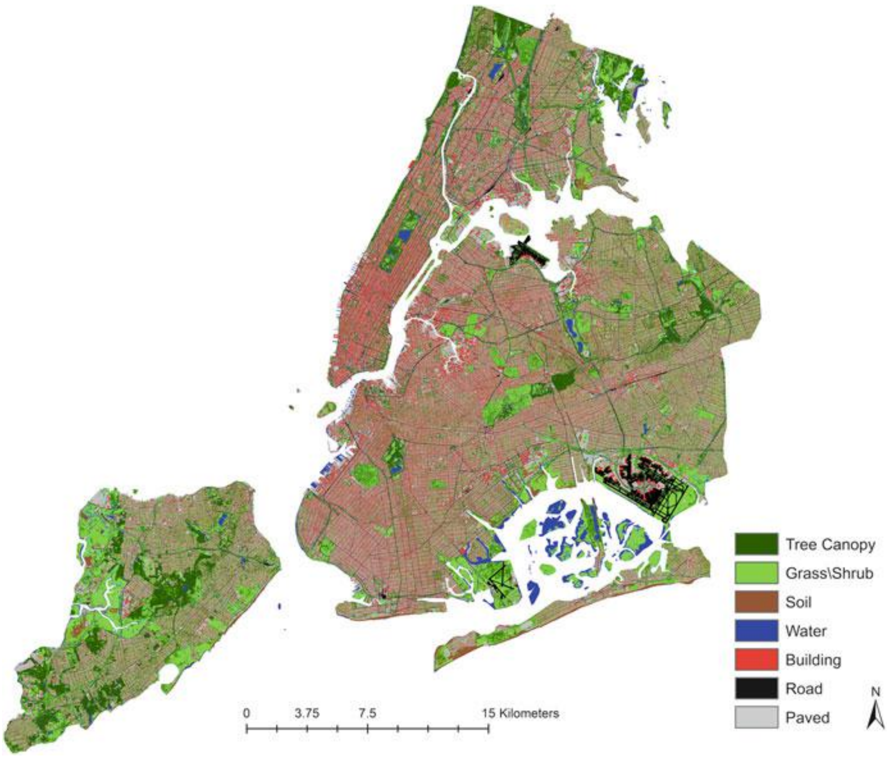

Fig. 19.1 Land use and land cover in the NYC municipal area showing dense urban development (*red*) and green space (*green*) (Data source: University of Vermont Spatial Analysis Laboratory and the New York City Urban Field Station, 2012, New York City Land Cover 2010. Prepared by and published with kind permission of ©Peleg Kremer 2013. All Rights Reserved)

NYC is also one of the greenest cities in the U.S. (Rogers 2011). The relatively small ecological footprint of New Yorkers compared to other American city residents is due primarily to high transit usage and multi-family housing, which is a consequence of NYC's population density (Owen 2004; Jervey 2006).

19.3 Challenges and Trends

19.3.1 Population Growth, Urbanization and Land Use

NYC's population grew 2.1 % from 2000 to 2010 (US Census Bureau 2011; Mackun and Wilson 2010). By 2030, the population is expected to increase by another 10 % to over nine million residents (City of New York 2006; PlaNYC 2007). Much of NYC's urban core is already dense, so intensified urbanization (i.e., land conversion) will present a challenge to maintaining regional biodiversity and ecosystem services (Pirani et al. 2012).

19.3.2 Urban Heat Island Effect, Climate Change and Sea Level Rise

The urban heat island effect (UHI) is a challenge in many cities (U.S. EPA Climate Protection Partnership Division 2008). UHI can be dramatic in NYC, with temperatures in the urban core and the surrounding suburban areas differing by up to 8 °C (Rosenzweig et al. 2009b). Potential future increases in overall regional temperatures combined with heat waves could significantly worsen UHI (Endlicher et al. 2008).

NYC: Predicted 2050 Climate Scenario

Temperature: +1.7 to 2.8 °C
Precipitation: +10 %
Average sea level: +17 to 30 cm

Source: Rosenzweig et al. 2009a

The NYC Panel on Climate Change (NYCPCC) reported that the city is vulnerable to rising sea levels, flooding from increased precipitation, and more extreme weather events. By 2050 it is predicted to be hotter, wetter, have higher average sea level, and experience more frequent and intense coastal flooding, and more frequent and intense heat waves (Figs. 19.2, 19.3, and 19.4) (Rosenzweig et al. 2009a).

Over 100,000 residents of New York live in the 100-year flood zone as currently mapped by the Federal Emergency Management Agency. As the seas rise and storm surges become more common, beaches and bluffs will suffer increased erosion, severe flooding and storm will disrupt public transit, and increased threat of saltwater infiltration into surface waters and aquifers will affect biodiversity (Frumhoff et al. 2007).

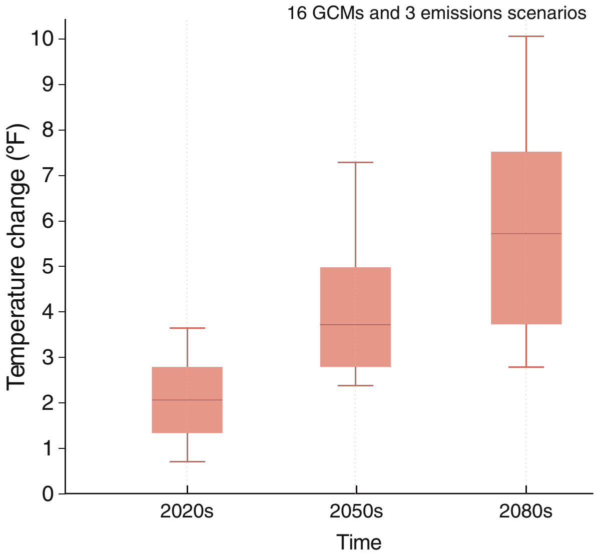

Fig. 19.2 Projected temperature changes by 30-year timeslice. The maximum and minimum values across the 16 GCMs and 3 emissions scenarios are shown as *black horizontal lines*; the central 67 % of values are shown in the *shaded areas*; the median is the *red line* (Modified from Rosenzweig et al. 2009a, p. 42, and published with kind permission of ©Rosenzweig 2009a. All Rights Reserved)

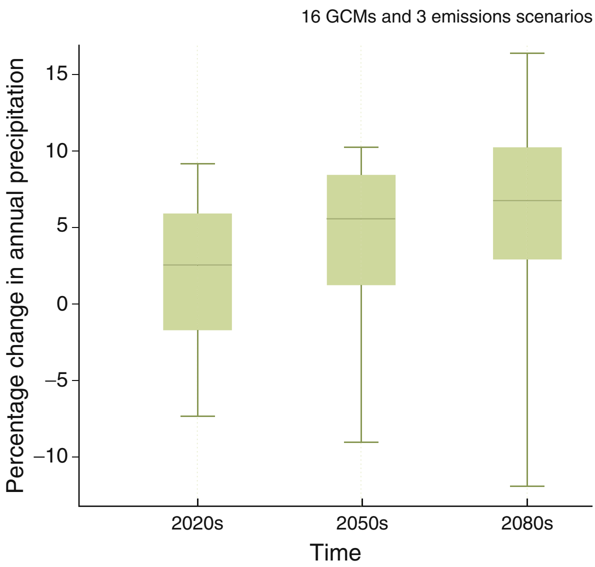

Fig. 19.3 Projected precipitation changes by 30-year timeslice. The maximum and minimum values across the 16 GCMs and 3 emissions scenarios are shown as *black horizontal lines*; the central 67 % of values are shown in the *shaded areas*; the median is the *dark grey line* (Modified from Rosenzweig et al. 2009a, p. 44, and published with kind permission of ©Rosenzweig 2009a. All Rights Reserved)

19.3.3 Stormwater

Stormwater runoff is a major contributor to water quality degradation in urban areas (NYS Department of Environmental Conservation n.d.). Seventy two percent of NYC's land area is covered by impervious surfaces (Alamarie et al. 2011). Like many mature cities, much of NYC has a "combined" wastewater treatment system.

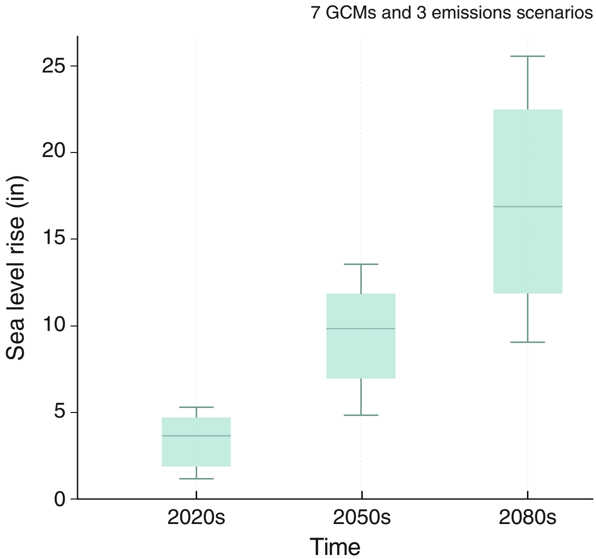

Fig. 19.4 Projected sea level rise changes by 10-year timeslice. The maximum and minimum values across the 7 GCMs and 3 emissions scenarios are shown as *black horizontal lines*; the central 67 % of values are shown in the *shaded areas*; the median is the *dark grey line* (Modified from Rosenzweig et al. 2009a, p. 46, and published with kind permission of ©Rosenzweig 2009a. All Rights Reserved)

During heavy precipitation, the storm sewers overflow into the sanitary sewers, mixing stormwater and untreated sewage (combined sewage overflows, or CSOs), releasing them into local waterways.

Rain events as small as 0.15 in. can trigger such events. Each year, more than 27 billion gallons of CSOs are diverted into the harbor (Plumb 2006), causing significant eutrophication (Howarth et al. 2000) and limiting recreation. Management of CSOs is of paramount importance for safeguarding aquatic biodiversity and a wide variety of ecosystem services. Therefore, the City's Department of Environmental Protection has invested over US$9 billion in reducing CSOs since 2002. PlaNYC, discussed below, is relying heavily on a combination of updated conventional "gray" infrastructure (such as increased capacity at treatment plants) and "green" infrastructure (see Sect. 19.6.1 below) technologies to reduce CSOs.

19.3.4 Air Quality

Health Consequences of Poor Air Quality

3,000 deaths
2,000 hospital admissions
6,000 emergency room visits

(continued)

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Ozone is Responsible for

400 deaths
800 hospital admissions
4,000 emergency room visits for children and adults

Sources: Kheirbek et al. n.d.

In the U.S., the Environmental Protection Agency sets air quality standards and NYC is currently designated as in "moderate nonattainment" for ozone and "nonattainment" for small particulates ($PM_{2.5}$) (US EPA 2012). The NYC Health Department estimates that reducing current $PM_{2.5}$ levels by 10 % could significantly reduce deaths and hospitalizations (Kheirbek et al. n.d.).

Local sources of fine particulates account for almost 50 % of the city's air pollution (Kheirbek et al. n.d.; Johnson et al. 2009). The highest levels are located at convergences of major emission sources, such as high buildings and traffic. Recently, a strategy to combine a phased-in regulatory ban on the dirtiest heating oils plus a financing program to encourage improved boilers in buildings is projected to contribute to a 50 % reduction in CO_2 and a 44 % reduction in soot emissions (PlaNYC 2012; Sklerov 2011).

Planting trees is a principal, low-cost tool for addressing air pollution, as well as helping to manage stormwater and mitigate the urban heat island effect. NYC trees remove about 2,200 tons of air pollution per year, valued at US$10 million annually (Nowak et al. 2007). An extensive tree planting effort, MillionTreesNYC, has been underway since 1997 and by mid-2012 was over 50 % complete, with over 600,000 trees planted in parks, along streets, and on private property (see below).

19.3.5 Public Health and Access to Green Space

Lifestyle factors, including physical inactivity and unhealthy diet are the primary contributors to obesity, which is the second major cause of premature death in NYC (NYC Active Design Guidelines 2010). Over 43 % of NYC elementary school children are overweight. Though not addressed directly in most discussions of biodiversity and ecosystem services, there is evidence that access to green space has direct effects on public health (Green Cities, Good Health 2012), and people in communities with abundant green space generally have better health (Harrison et al. 1995). Equity in the distribution of NYC green space is one goal of PlaNYC.

19.3.6 Invasive Species

With one of the busiest international ports in North America, NYC is an epicenter for invasive species in the U.S. (Bustamante and Taylor 2011) and functions as a major pathway for invasive species to the rest of the continent. Nonnative invasive species cause environmental losses and damages in the U.S.: US$120 billion annually (Pimentel et al. 2005). Historically in NYC, after habitat destruction, the biggest threat to local flora has been invasive species. For example, Pelham Bay Park is the largest natural area in the NYC Park system. Over a 50-year period it lost 2.8 native plant species every year, while it gained 4.9 new exotic species (DeCandido 2004).

Effects of invasive species on native populations have increased in New York State over the past several decades and nonnative invasive woody-plant species are rapidly spreading in the New York region, while native species tend to generally be in decline (Clemants and Moore 2005; New York State Invasive Species Task Force 2005). In the Hudson River 33 % of fish species are suspected to be non-native (Strayer 2010).

19.4 Biodiversity and Habitats of NYC

19.4.1 Snapshot of Current Biodiversity in NYC

Snapshot of NYC Biodiversity

26 distinct ecological habitat types
1,450 plant species native to the five counties of NYC
826 native plants still have extant populations
140 plants species with some formal designation of rarity
220 native species of bees

Source: NYC Department of Parks and Recreation 2006

NYC's position on the border of New England and the Mid-Atlantic regions results in exceptional biodiversity. The range of habitats, from serpentine grasslands in Staten Island to vernal ponds in Alley Pond Park in Queens is an indication of this variety.

There are 26 distinct habitat types in NYC and 1,450 species of plants are native to the 5 counties of NYC; 57 % (826) of these are currently extant and 20 % are found in more than 5 sites (Ed Toth 2011, personal communication). However, 93 % of these species are in decline in the last 100 years, not only because of habitat loss, but because of changing environmental factors such as climate, habitat fragmentation, decreased pollinator availability, UHI, invasive species, and pollution. There are at least 140 formally designated rare animal species found in NYC (Fig. 19.5).

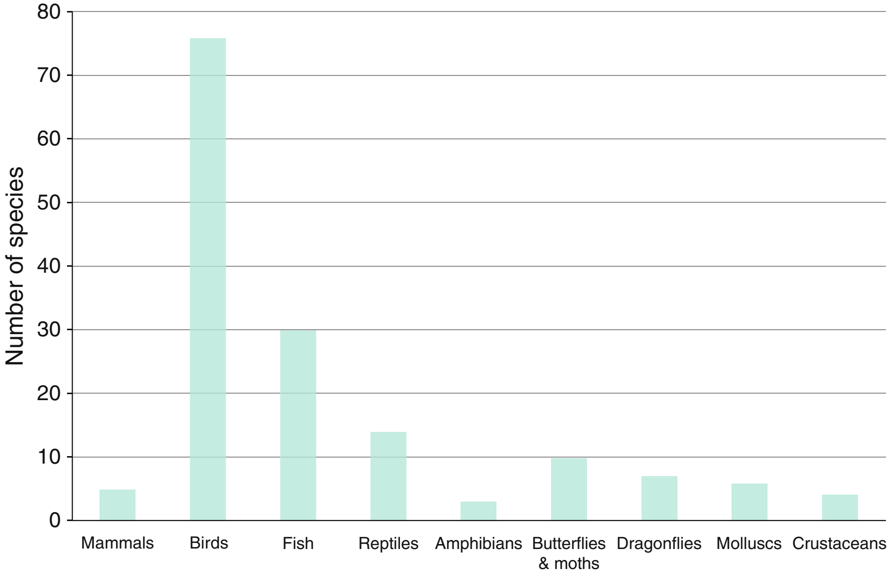

Fig. 19.5 140 formally designated rare species in NYC in their various taxa. Designations include: Federal or State listed Endangered, Threatened, Special Concern, or Species of Greatest Conservation (Prepared by and published with kind permission of ©David Maddox 2013. All Rights Reserved)

Estimated Number of Trees in NYC, as of Late 2012

Street trees: 593,132
Families: 19
Genera: 47
Species: 206
Source: NYC Parks 2006 Street Tree Census

Ca. 2 million trees in landscaped parks

Total trees in NYC: 5.2 million
Source: Nowak et al. 2007

Total canopy cover (2010): 21 % of land area
Source: NYC Urban Field Station data

5,136 acres of forest

Informal goal for canopy cover in 2030: 30 %
Source: Urban Tree Canopy Assessment (http://nrs.fs.fed.us/urban/utc/) and NYC Department of Parks and Recreation (personal communication)

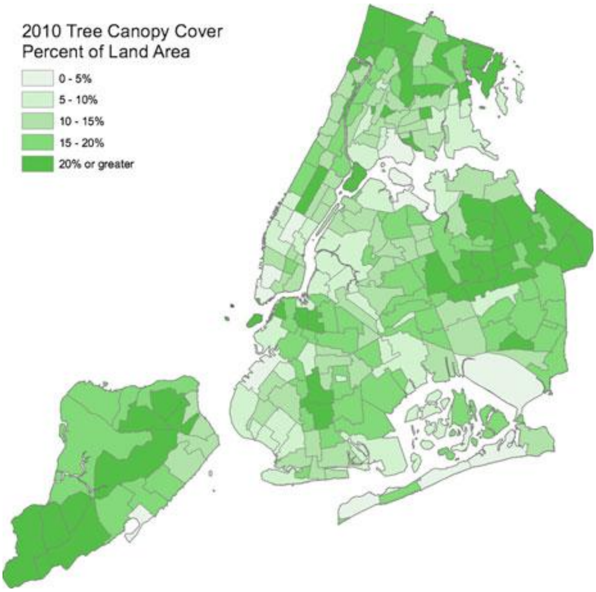

Fig. 19.6 2010 Tree canopy cover across NYC as a percent of land area (Reproduced from NYC Department of Parks & Recreation and published with kind permission of ©NYC Department of Parks and Recreation 2010. All Rights Reserved)

19.4.2 NYC's Urban Forest

Grove et al. (2006) report that tree canopy in NYC is 24 %, although more recent calculations by the City report 21 % (Fig. 19.6). An ambitious public-private partnership called MillionTreesNYC (MTNYC) is underway that will increase the tree canopy to (an informal goal of) 30 % by 2017. Current tree canopy is significantly variable across NYC neighborhoods (Fig. 19.6), and it is the intention of MTNYC to increase equitable access to green spaces across neighborhoods (Locke et al. 2010). Such a tree canopy could reduce surface temperature by a full degree or more (Grove et al. 2006; Rosenzweig et al. 2009b).

Street trees in NYC are valuable stormwater management infrastructure, intercepting almost 900 million gallons of stormwater annually, or an average of 1,500 gal per tree. The total value of this benefit to NYC is more than US$35 million each year (Peper et al. 2007). Urban trees also improve air quality by removing dust and other pollutants, including those that trigger asthma and other respiratory illnesses. A recent study found that higher street tree densities were associated with lower prevalence of asthma among children aged 4–15 in NYC (Lovasi et al. 2008).

Economically, trees provide an important return on the significant investment cities make in their care and planting. In NYC, trees provide approximately US$5.60 in benefits for every dollar spent on tree planting and care, dollars that would otherwise be spent on energy for cooling and stormwater retention services (Peper et al. 2007).

19.4.3 Wetlands in NYC

Facts and Concerns About NYC Wetlands

In the last 100 years ca.85 % of freshwater and intertidal wetlands in the NYC region have been lost
Currently there are >1,500 acres of salt marsh
Currently there are >1,600 acres of freshwater wetlands
325 species of bird have been sighted in Jamaica Bay
Over 100 species of fish occur in Jamaica Bay
Over 800,000 people live in the communities adjacent to Jamaica Bay

Source: US Fish and Wildlife National Wetland Inventory

The wetlands and riparian systems in NYC vary widely in size, type and condition, and include diverse functions from regionally critical habitat for local and migrating birds and fish to flood management and recreation for human communities. The wetland and riparian system contains approximately 1,500 acres of salt marsh and 1,600 acres of freshwater wetlands: 30 % emergent; 29 % open water; 7 % scrub/shrub; 34 % wet forest (NYSDEC and US Fish and Wildlife National Wetland Inventory).

The largest wetland complexes are found in Jamaica Bay and the Arthur Kill watersheds, where tidal wetlands dominate. The wetlands in the south and eastern sides of Staten Island, part of the Lower Bay watershed, are dominated by freshwater systems.

Roughly 85 % of the coastal wetlands and over 90 % of the freshwater wetlands have been lost in the New York-New Jersey Harbor Estuary over the last century (Regional Plan Association 2002). Hundreds of miles of riparian corridors were developed, headwater streams were filled and piped, and higher order streams were straightened and disconnected from their floodplains through typical urban development. Comparing current stream mapping to historical mapping (Eymund Deigel, personal communication) between 40 and 90 % of streams in NYC have been buried or filled at a proportion approximately equivalent to the impervious area in the landscape. The greatest intact stream length remains in Staten Island, but even there most streams have been extensively modified directly or indirectly.

Since the Clean Water Act the rate of wetlands loss has been dramatically reduced in NYC, as around the U.S. Nevertheless, incremental filling of State unregulated wetlands (wetlands smaller than 12.5 acres) and development in the wetland buffer areas has continued. One analysis comparing current development to the regulated wetlands first mapped by New York State in the 1970s suggests 3–9 % of the wetland area has been filled in Staten Island alone (Eymund Deigel, personal communication, unpublished data). Incremental loss of salt marsh in NYC has also

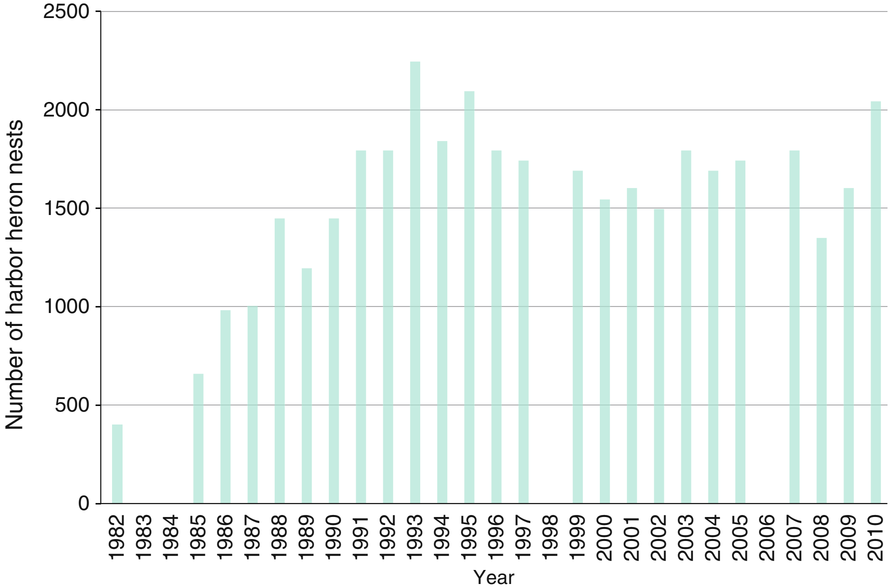

Fig. 19.7 The number of "Harbor Heron" nests in recent surveys (Modified from New York-New Jersey Harbor & Estuary Program (2012), and published with their kind permission. All rights reserved)

continued due to a variety of on-going environmental stressors and impacts. At seven wetland sites around the city, historic photo analysis from 1974 to 1999 and 2006 show loss rates of 1–2 % of the total salt marsh area per year (NYC Parks and Recreation Natural Resources Group 2010 unpublished data; Hartig et al. 2002). Today, most of the City's wetlands are smaller than 3 acres in size, an indication of how fragmented NYC wetlands are.

NYC's largest remaining wetland complexes are found in Jamaica Bay (see below), Staten Island, and the Upper East River and Western Long Island Sound. Northwest Staten Island contains a diverse array of wetland types, including salt and freshwater meadows, spring-fed ponds, forested swamps, creeks, and salt marshes.

After virtually disappearing from the New York Harbor area, "Harbor Herons" (a umbrella term including the great egret, snowy egret, black-crowned night heron, blue heron, and glossy ibis) began to appear again with the improvement in water quality over the last 30 years (Fig. 19.7). More than 100 bird species have been observed nesting or feeding in Arlington Marsh, even though large portions of this site were contaminated by industry and the ecosystem is highly disturbed.

Thousands of acres of salt marsh, tidal channels, and mud flats once characterized the Bronx shoreline, on the north side of the Upper East River. Most of these areas were filled by the 1950s. Existing tidal wetlands are concentrated in Pelham Bay Park along Goose Creek Marsh on the Hutchinson River. Most of the remaining freshwater wetlands are found in large parks, including Van Cortland Park in the

Bronx and Alley Pond Park and Forest Park in Queens. Staten Island's Greenbelt Park System and its multiple parks along the south shore contain the largest number and most diverse array of remaining freshwater wetlands in the city.

19.4.4 Jamaica Bay

Jamaica Bay, adjacent to JFK Airport, is a socially and ecologically important coastal ecosystem for the Northeast and also one of the largest and most biologically productive, housing the largest tidal wetlands in the NYC metropolitan area. Over 800,000 people live around the Bay's margins. It is important habitat for wildlife, with more than 100 species of fish, a number of endangered species (including the peregrine falcon, piping plover, and the Atlantic Ridley sea turtle), and 214 "species of special concern." More than 325 species of birds have been sighted in the Bay, which serves as an important stopover point on the Atlantic Flyway migration route for nearly 20 % of the birds on the continent (Jamaica Bay Watershed Protection Plan Advisory Committee 2007; U.S. Fish and Wildlife Service 1997).

The most significant threats facing Jamaica Bay are marsh fragmentation and loss resulting from various factors including hardening of the coastline, pollution and CSO inputs, dredging, sea level rise, and the loss of freshwater tributaries.

Aggressive plans exist to upgrade existing wastewater treatment plants; reintroduce native species; and develop green infrastructure in the surrounding areas to help reduce stormwater runoff and storm surge (NYC Department of Environmental Protection 2007, 2010). High-level agreements between U.S. Department of the Interior and the City in 2012 have committed the National Park Service (and by extension other federal agencies) to cooperatively manage public lands and waters, restoration projects, and research in the Jamaica Bay and Rockaway Peninsula areas.

19.4.5 The Big Oyster

Oysters were once an abundant resource of the Hudson River Estuary. In the nineteenth century, there were approximately 907 km^2 of oyster reefs in the region (Kurlansky 2006). By the early twentieth century, sediment, water pollution and overharvesting virtually eliminated oysters from NY harbor (Hudson River Foundation et al. 2010). The loss of oysters in the NYC Harbor has not only diminished access to a regional food, but has also resulted in the loss of valuable ecosystem services such as improving water quality and provision of physical habitats for fish and invertebrates (Nelson et al. 2004). Several organizations have been involved in oyster restoration and research, and experimental oyster reefs were seeded in Fall of 2010 throughout the New York Harbor Estuary (Grizzle et al. 2011).

19.5 Organizations and Major Initiatives in Support of Biodiversity and Ecosystem Services

Three broad features of environmental work in NYC are key to its current success in environmental protection. First, there is broad and specific support for environmental goals and action laid out in the Mayor's comprehensive PlaNYC (2011). Key to the success of this document is that it articulates 132 specific environmental initiatives, including the MillionTreesNYC effort. Other signature efforts include the Green Infrastructure Plan and wetlands assessments and restoration.

PlaNYC publications relevant to biodiversity and habitats; all are available at http://www.nyc.gov/html/planyc2030/html/home/home.shtml

Climate Resilience

- Climate Change Adaptation in NYC
- Climate Risk Information

Waterways and Stormwater

- Green Infrastructure Plan
- Wetlands Strategy
- Wetlands: Regulatory gaps and other threats
- Preliminary survey of wetlands
- Sustainable Stormwater Management Plan 2008

MillionTreesNYC

Second, a diverse constellation of public and private organizations working on various efforts directly or indirectly relating to biodiversity and environmental stewardship, from Federal and City government, to NGOs and university research. For example, StewMap, a project of the U.S. Forest Service and others, created a database of existing organizations whose missions are related to environmental stewardship and demonstrates the intensity of current environmental effort in the city (Fig. 19.8) (Connolly et al. 2012).

Third, organizations in NYC made a concerted effort to be information-driven. Although the connection between data and decision-making is not seamless, great strides have been made to make NYC a place where biodiversity protection and environmental planning can be conducted through adaptive management principles. Several high profile projects and organizations are highlighted below and many others are underway.

Fig. 19.8 Local stewardship organizations in NYC, based and working across the city (Reproduced from the Stewardship Mapping and Assessment Project, USDA Forest Service 2007, and published with their kind permission. All Rights Reserved. Svendsen et al. (2007). See also Svendsen and Campbell (2008))

19.5.1 PlaNYC 2030

In 2007 NYC Mayor Michael Bloomberg launched PlaNYC (New York City 2007), a set of long-term strategies that will cumulatively make a greener NYC by 2030. PlaNYC is an economic and environmental sustainability plan for New York City with long-term goals in ten policy areas:

1. Create homes for almost a million more New Yorkers while making housing more affordable and sustainable
2. Ensure that all New Yorkers live within a 10-min walk of a park
3. Clean up all contaminated land in New York City
4. Improve the quality of our waterways to increase opportunities for recreation and restore coastal ecosystems
5. Ensure the high quality and reliability of our water supply system
6. Expand sustainable transportation choices and ensure the reliability and high quality of our transportation network
7. Reduce energy consumption and make our energy systems cleaner and more reliable
8. Achieve the cleanest air quality of any big U.S. city
9. Divert 75 % of our solid waste from landfills
10. Reduce greenhouse gas emissions by more than 30 %

These goals are addressed through 132 specific initiatives ranging from improved bicycle and pedestrian facilities, grants and liability provisions for contaminated land remediation, enhanced waste recycling, to new regulations for energy efficiency in existing buildings. Some of the initiatives specifically address greening and habitat for biodiversity, such as MillionTreesNYC and wetlands projects.

PlaNYC has gained tremendous attention both nationally and internationally, acknowledged around the world as one of the most ambitious – and most pragmatic – sustainability plans anywhere. However, how the plan will impact biodiversity and the provisioning of ecosystem services in the city and region has yet to be comprehensively explored.

The April 2011 update to PlaNYC included for the first time a section focusing specifically on how various PlaNYC initiatives affect "natural systems" (PlaNYC Update April 2011, 166–167). Examples of the initiatives highlighted: expansion of land preservation and use of green infrastructure for stormwater management; modification of building and construction codes to increase stormwater capture; identification of coastal protective measures in the face of climate change; a regulatory and financial program to hasten boiler conversions to use cleaner fuels; and restoration of wetlands.

A majority of PlaNYC initiatives got underway shortly after publication of the plan's first iteration in 2007, and increasingly are embedded in City policy, practice and even in some cases code or law. Examples of progress include the over 600,000 trees planted, the investments being made in stormwater management and treatment, and the expansion of the parks network.

More about the plan can be found at http://www.nyc.gov/html/planyc2030.

19.5.2 New York Parks and Recreation Department

The NYC Department of Parks and Recreation (NYC Parks) division of Forestry, Horticulture, and Natural Resources is the agency's primary environmental and conservation unit. It has a capital budget to build and restore forests, wetlands, and grasslands (having restored over 1,500 acres of these habitats) and create green infrastructure citywide: greenstreets, bioswales, street trees, etc. Their relationship with the U.S. Forest Service, which led to the creation of the NYC Urban Field Station, has engendered relationships with over 60 institutions – academic, non-profit, and other municipal agencies – on research projects that span the region and with studies including bioindicator species, stormwater capture effectiveness, public design innovation, air quality, community stewardship, green roofs, tree mortality, and urban plant genetics. Science, GIS, and land mapping underscore every aspect of the division's work. Thus, the division views all plantable areas, from rights-of-way to natural areas, as interconnected components of a complex urban transect: every inch is potential space for green infrastructure and ecological engineering.

19.5.3 Greenbelt Native Plant Center

The Greenbelt Native Plant Center (GNPC) uses seed from local native flora to produce plants for ecological restoration work within the region. This effort is now expanding into the Mid-Atlantic Seed Bank, in which the GNPC will be the main conduit for seed collection and banking for restoration work throughout the region.

19.5.4 United States Forest Service Northern Research Station

The Northern Research Station of the U.S. Forest Service (USFS) has made significant and ongoing investments in urban social-ecological research across the U.S. and in several Urban Field Stations. One of these, the NYC Urban Field Station (NYCUFS), was co-founded and is co-managed by the USFS and NYC Parks. The NYCUFS is both a physical space (e.g., a lab, housing for visiting scientists, meeting rooms) and an institution to facilitate collaboration among scientists and practitioners on a variety of significant research on social-ecological systems, biodiversity and natural resource management in NYC. Particular research priorities are urban forestry, environmental literacy, resilience, health and well-being, and environmental governance and civic engagement. For information on the UFS see http://www.nrs.fs.fed.us/nyc/.

19.5.5 Natural Areas Conservancy

The Natural Areas Conservancy's (NAC) was conceived and developed to expand the work done by the NYC Park's Natural Resources Group, which has been practicing urban restoration and conservation since 1984. The mission of NAC is to restore, protect, manage and expand NYC Parks' 10,000 acres of forests, wetlands, and grasslands. Modeled on the Central Park Conservancy, NAC was founded in 2012.

NYC's natural areas are dispersed in parklands across the five boroughs. Conceptually, administratively, and fiscally, ongoing conservation and management requires looking at NYC's natural areas not as isolated patches, but as a single unified urban biosphere with an administrative whole. In addition, many of NYC's natural areas are held by other municipal, state, federal, and private entities, calling for additional channels of coordination and land management.

The NAC will unify the public identity, planning, management, and care of the more than 10,000 acres of natural areas overseen by Parks. The NAC will increase public awareness and volunteerism, and fund necessary research and development towards the implementation of advanced technology and management tools. The NAC has already begun its first signature project, a citywide ecological assessment that will be used in long-term management of the NYC's natural areas. This project is being conducted in partnership with the American Museum of Natural History, the U.S. Forest Service, The Nature Conservancy, the Wildlife Conservation Society, the Brooklyn Botanic Garden, the NY Heritage Program, and others.

19.5.6 MillionTreesNYC

MillionTreesNYC Accomplishments as of 2012

Planting

Street trees planted:97,870
Reforestation trees planted:316,585
Plantings on other public/private land:197,822

Stewardship

Number of citizen stewards trained:11,256
Number of tree care workshops given:971

Programs

Training Program graduates:104

MillionTreesNYC (MTNYC), a campaign to plant one million trees in NYC by 2017, is one of the most visible and successful initiatives in PlaNYC. To achieve this ambitious goal, NYC Parks allocated US$400 million to the MTNYC campaign over 10 years and developed a public-private partnership with the local non-profit New York Restoration Project (NYRP). Ultimately, NYC plans to add 220,000 street trees, filling every available street tree opportunity, and plant 500,000 park trees. Meanwhile, NYRP is coordinating the planting of 300,000 trees with private organizations, homeowners, and community organizations. MTNYC has planted 612,277 trees as of 2012, over 97,870 of which are street trees.

At the beginning of the campaign, NYC Parks initiated a strategy of full-block planting to rapidly green entire neighborhoods, initially targeting areas with few trees and high asthma rates in a program called Trees for Public Health, that was devised based on NYC Department of Health data showing higher incidences of childhood respiratory ailments in these communities (MillionTreesNYC 2012).

Public, private, and non-profit organizations have used the campaign to build community, encourage people-nature interactions and increase opportunities for civic ecology (Krasny and Tidball 2012) and environmental education in an unprecedented citywide environmental movement. Significant effort has been made in research (McPhearson et al. 2010a, b) and assessment (e.g., mortality studies and the relatively effectiveness of different tree pit designs) to make the MTNYC program as effective as possible.

19.5.7 New York Metropolitan Flora Project

In 1990, the Brooklyn Botanical Garden launched the New York Metropolitan Flora project (NYMF), a multi-year regional effort to document the flora in all counties within a 50-mile radius of New York City, including all of Long Island, southeastern New York State, northern New Jersey and Fairfield County, Connecticut.

19.5.8 NYC Parks and Open Space Access

One goal of PlaNYC is to ensure that every New Yorker lives within a 10-min walk of a park. Currently, more than two million residents are not within this range (PlaNYC 2011). Since 2007, more than 250,000 New Yorkers have gained 10-min walk access to a park, nearly 180 schoolyards to playgrounds sites and 260 green streets have been developed (PlaNYC 2011).

Additionally, since the first Waterfront Plan in 1992 NYC has acquired 506 ha of waterfront as parkland. Wastewater treatment initiatives, including US$6 billion allocation to upgrade the City's wastewater treatment plants and more than US$1 billion to reduce CSOs, have contributed toward making the city's waterways cleaner than they have been in a century. The 2010 Waterfront Open Space Plan calls for dozens of redevelopment sites to be completed by 2020 (NYC Comprehensive Waterfront Plan 2011).

19.5.9 Other Organizations Working in NYC

NYC contains a rich network of organizations working in areas relates to biodiversity, stewardship, and civic engagement. StewMap documents these organizations and makes them accessible through a searchable database (Connolly et al. 2012).

19.6 Biodiversity Protection Through Natural Areas Planning and Regulations

There are many regulations and programs in NYC aimed at conserving biodiverse natural areas, including development permitting mechanisms, land management by governmental entities, species data collection and environmental education. These efforts, which are particularly focused on waterfront, wetland and rare plant communities, involve cooperation among federal, state, local and nonprofit organizations.

19.6.1 Green Infrastructure Plan

NYC has adopted a sustainable infrastructure investment approach that addresses multiple goals by using "blue roofs," larger street tree pits, "green streets," porous concrete, and vacant lots to control stormwater and provide additional ecosystem services (Cohen and Ackerman 2010; McPhearson et al. 2012a, b). The plan (NYC Green Infrastructure Plan 2010), which has committed a total of US$2.4 billion over 20 years, is designed to control 10 % of stormwater runoff using green infrastructure over 20 years and is estimated to reduce CSOs by approximately 1.5 billion gallons per year (Alamarie et al. 2011). The Staten Island Bluebelt, one of the largest watershed-level stormwater management systems in the U.S. (4,856 ha of waterways and wetlands), is evidence of the importance of urban green infrastructure. The Bluebelt has proved to have 40 % removal efficiencies for nitrates and has saved the City more than US$80 million in comparison to traditional sewer construction (Gumb et al. 2008).

19.6.2 Waterfront Revitalization Program

Since 2002, City, State and Federal partners have invested over US$56 million at 16 sites to restore 59 ha of wetland. Sixteen restoration projects involving 50 ha are expected to be completed by 2013 (NYC Office of Long-Term Planning and Sustainability 2012). Additionally, the Waterfront Revitalization Program (WRP) designates Special Natural Waterfront Areas (SNWA), requiring that particular habitat features be considered in connection with development in these areas. Three SNWAs have recently been designed including Northwest Staten Island Harbor Herons Area, Jamaica Bay, and East River Long Island Sound. WRP calls for the City to prevent the net loss of wetlands. A 2012 revision of WRP to offer greater protection by designating additional sites of ecological importance is expected to be voted on by the City Planning Commission and City Council by late 2012 or early 2013, and would then be subjected to state and federal review before going into effect in 2013 or 2014.

19.6.3 Special Natural Area District

In 1975, the Special Natural Area District (SNAD) was created to improve preservation of natural features in parts of Staten Island, the Bronx and Queens. In order to guide development in a way that preserves unique natural features, the Planning Commission must review new developments and site alterations on primarily vacant land to ensure that significant natural features are preserved (NYC Department of City Planning 2012).

19.7 Recommendations for Biodiversity Management and Protection in NYC

19.7.1 Inventory and Assess Natural Areas in NYC

Knowledge of the current distribution, abundance, and status of current biodiversity and natural areas is fundamental to any protection. The new Natural Areas Conservancy (see above) has urban biodiversity inventory as part of its mission for land held by the city. A preliminary wetlands assessment has been conducted as part of PlaNYC. It is hoped that other organizations, including large conservation NGOs, will also invest in such work. Planned efforts by the National Park Service around Jamaica Bay include long-term monitoring.

19.7.2 Manage and Restore Natural Areas

The NYC Department of Parks and Recreation is actively involved in restoration of forest habitats and intertidal zones. The National Park Service and others are engaged in restoration of wetland habitats, for example in Jamaica Bay. However, land conversion is still a significant threat in NYC, including in NYC parks, where there is persistent and legitimate demand for additional recreation facilities. All natural areas would benefit from comprehensive management plans that guide restoration work and inform managers of the consequences of land conversion for biodiversity and ecosystem services.

19.7.3 More Research in the Potential for Green Infrastructure to Support Biodiversity

Trade-offs and synergies between biodiversity and ecosystem services need to be better explored in urban planning, research and modeling to provide best practices for maximizing the biodiversity potential.

19.7.4 Promote Native Species and Manage Against Invasive Species

Requiring the use of native species and locally adapted varieties in all city planting and restoration efforts would benefit local ecosystems. MillionTreesNYC, for the most part, plants native species. Other programs are needed to promote the use of native species in green infrastructure programs such as Green Streets. Increasing the capacity of the Greenbelt Native Plant Nursery to provide additional native plant material for green infrastructure is one avenue for achieving this.

19.7.5 Aggressively Promote Equitability in the Distribution and Access to Natural Areas and Biodiversity

The current distribution of green spaces, parks, and biodiversity is uneven across the city. Remedying this fact is a core goal of PlaNYC. Additional analysis of the full suite of ecosystem services provided by green infrastructure combined with an increased understanding of the public need for a wide variety of ecosystem services provided at the neighborhood level would help to prioritize green space development in underserved areas of the city (McPhearson et al. 2012a, b).

19.7.6 Engage Citizens and Promote Ecological Literacy

Connecting residents with urban nature is critical for developing nature awareness and students would benefit from increased opportunities in schools and communities to learn about nature in their backyards. This includes curriculum development, teacher trainings, and community programs to increase people-nature interactions and improve ecological literacy.

19.8 Conclusion

Planning a more sustainable and resilient NYC in the face of multiple social, economic, and environmental pressures requires an integrated social-ecological approach to urban research, planning, policymaking, and management, and biodiversity protection in particular. NYC has taken a significant step in this direction with PlaNYC, the environmental and economic sustainability plan for 2030. Major investments in green infrastructure, tree planting in target areas, ecological restoration, and habitat protection for biodiversity have helped make NYC a model global green city. Many organizations contribute to work now done in NYC, including federal and city agencies, non-profits, and community organizations working in research, planning, and stewardship. However, with nearly a million new residents expected within the city boundaries by 2030, and even more in the metropolitan region, there is still progress to be made in linking social planning with ecological planning across scales from neighborhoods to the region.

The Green Infrastructure Plan for NYC is one of the many strong examples of how NYC has focused on ecosystems as a resource for positive change within the city. The plan focuses primarily on stormwater absorption, but also represents a unique opportunity for NYC to substantially expand its already robust network of urban farms and gardens and transform vacant land into spaces that provide increased habitat for biodiversity, while developing corridors to improve connectivity among fragmented urban nature patches. The New York Metropolitan Region has historically harbored incredible biodiversity and, even with significant development, still retains strong ecological resources for ecosystem services. Open space protection in the region has proceeded aggressively in recent years especially driven by the Open Space Institute in collaboration with city and state governments, communities, farmers, and NGOs.

Ecosystems and the biodiversity in them provide critical ecosystem services to human inhabitants of NYC by supplying food, drinking water, wood and fiber, flood control, air purification, stormwater absorption, carbon sequestration and storage, temperature regulation, noise mitigation, aesthetic and recreational value, education opportunities, and a sense of well-being (for a complete review of NYC ecosystem services see CBO Scientific Assessment Chap. 4, this volume). Therefore, the human components of the NYC social-ecological system are intimately tied to the ecological components of the city and the region. Trade-offs and synergies in the provisioning of ecosystem services need to be better explored in urban planning and scenario modeling to provide best practices for maximizing the biodiversity and ecosystem services in the NYC metropolitan region. The progress NYC has made in this regard in recent years is significant, with strong plans in place and powerful motivation among community groups and city agencies to continue building the necessary improvements to the socio-ecology of the New York regional system.

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Chapter 20
Local Assessment of Melbourne: The Biodiversity and Social-Ecological Dynamics of Melbourne, Australia

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Abstract Melbourne, Australia is a city rich in biodiversity. It contains a high proportion of open space and supports a large number of flora and fauna species, both indigenous to the region and introduced from around the world. The high levels of biodiversity are partly the result of historical planning decisions that did not deliberately consider biodiversity yet inadvertently favoured many plants and animals. However, Melbourne is currently at a tipping point whereby continued urban growth is likely to result in a loss of biodiversity if it is not explicitly and carefully considered in planning, policy and management. Enhancing biodiversity into the future will be aided by a reconciliation of underlying tensions between (1) growth and conservation and (2) the management of 'native' and 'exotic' vegetation that are currently embedded in a range of governance structures and public attitudes. This would enable the implementation of urban design that promotes biodiversity across the city as a whole.

Keywords Urban sprawl • Habitat loss • Endangered species • Cultural preference • Native and exotic species

Key Findings

- Melbourne is rich in biodiversity because of its natural setting and historical land use decisions that have unintentionally favoured many species.
- Biodiversity values are threatened due to the rapid low-density expansion of the city on its fringe and the gradual degradation and loss of habitat within the urban matrix.
- Both native and introduced vegetation is valuable for ecological and social reasons, yet there are tensions around which should be prioritised in highly contested urban settings.
- Sophisticated biodiversity conservation legislation exists to curb ongoing losses of native vegetation. Although this has reduced the loss of native vegetation, declines in the extent and condition of threatened ecosystems around the city continue.
- Enhancement of Melbourne's biodiversity in the future will require (i) changes to the nature of fringing urban development to reduce impacts on critically endangered ecosystems, (ii) greater commitment to protect, maintain and restore vegetation on public and private land, and (iii) increased emphasis of co-benefits of biodiversity and human wellbeing.

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20.1 Introduction

Melbourne is a dynamic, culturally diverse and growing city located in a region of remarkable physical and biological diversity. It was built on the northern tip of Port Philip Bay (Fig. 20.1) and along the banks of the Yarra River in southeastern Australia. A young city by global standards, Melbourne was first settled by Europeans in 1835, yet has an indigenous history going back tens of thousands of years. What is now central Melbourne was once an important meeting place for many Aboriginal tribes (Presland 2010).

Melbourne was a favourable location for European settlement because of the readily available goods and services provided by natural ecosystems. These included clean water from the Yarra River, productive soils for growing food crops on the alluvial plains, and timber from nearby forests. The arrangement of the early township was planned strategically from its beginnings and was based on a grid arrangement of blocks with wide main streets interspersed by narrow laneways (Brown-May and Swain 2005). These remain characteristic features of the city centre today.

Initially, economic growth was driven by exporting natural resources such as gold and wool to markets of the British Empire. While reliance on ecosystem services provided vast wealth, withdrawal of foreign investment and a collapse in property prices led to a severe economic depression during the 1890s. By the early twentieth century, Melbourne's economy had diversified and a large manufacturing industry was being developed. By the 1970s, Australia embarked on a series of economic reforms and Melbourne's economy shifted to a more "economic rationalist" structure and an increased emphasis on services (Connolly and Lewis 2010). In the early twenty-first century, Melbourne has a vibrant, diversified and internationally competitive economy providing a wide range of goods and services.

Immigration from overseas migrants contributed to several instances of rapid population growth in Melbourne. During the Victorian gold rush of the 1850s, over 600,000 prospectors from around the world arrived in Melbourne, with half of them settling in the city afterwards (Brown-May and Swain 2005). Although immigration

Fig. 20.1 Aerial view of Melbourne (Photo courtesy of ©James Relph 2012. All Rights Reserved)

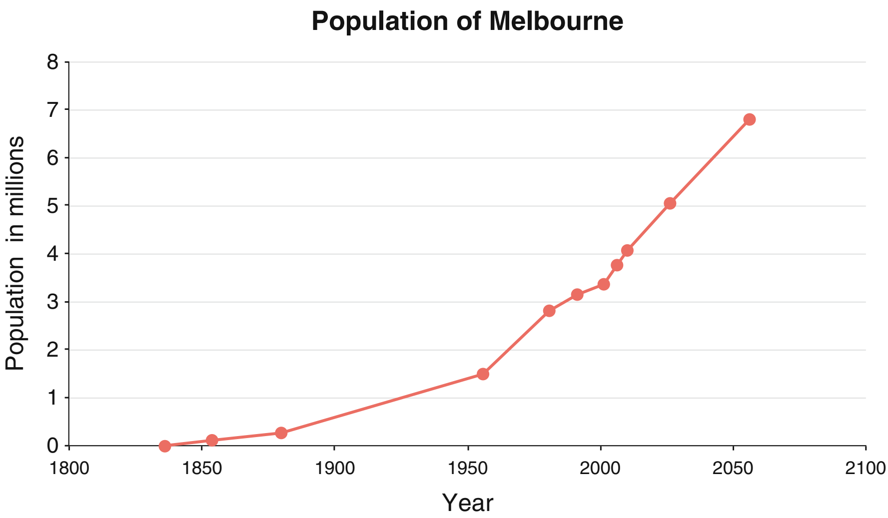

Fig. 20.2 Past and future projected population of metropolitan Melbourne (Source data from the Australian Bureau of Statistics, 2012)

from the United Kingdom was dominant, many prospectors from China and Western Europe also settled in the city. After Australia's federation in 1901, the White Australia Policy restricted immigration by non-whites. Following the Second World War, large numbers of migrants arrived from southern Europe (Greece and Italy in particular). Abandonment of the White Australia Policy by the 1970s resulted in increased immigration from East and Southeast Asia, followed more recently by immigration from South Asia, the Middle East, and Africa.

As of 2010, Melbourne has an estimated population of 4.08 million (ABS 2011) (Fig. 20.2). While average population density is low (530 people/km^2), it is highly variable, with inner Melbourne supporting 8,200 people/km^2 (ABS 2012). The city's population is projected to reach 6.5 million people by 2051 with much of the growth concentrated in the outer suburbs (ABS 2011). This has resulted in some fringing municipalities having current growth rates of over 8 % per year (ABS 2012).

A variety of geological formations have resulted in geomorphically and ecologically distinct landscapes within the Melbourne region. The western suburbs are located on flat Quaternary volcanic basalt plains, while the eastern parts of the city are located on an incised and folded platform of Silurian sedimentary rock. Much of the central and southern parts of the city are located on low elevation coastal and alluvial plains overlying Tertiary sandstones, clays and gravels. Extensive beach ridges have historically produced swamps through inhibiting drainage, however many of these have been artificially drained and the land claimed for agricultural or commercial use (Brown-May and Swain 2005).

Melbourne's climate is temperate yet variable, with a rainfall gradient ranging from less than 500 mm/year in the west of Metropolitan Melbourne to over 1,100 mm/year to the east (Bureau of Meteorology and Walsh 1993; Brown-May and Swain 2005). Temperatures range from a mean maximum of around 25 °C in summer and between 13 and 14 °C in winter. Melbourne's mean temperature has been rising over the past 50 years at a rate of 0.14 °C per decade and scientists predict it will continue to rise due to the effects of global climate change (Climate Change Task Force 2008).

20.2 Urbanization, Ecosystem Services and Biodiversity: Scenarios and Trends

20.2.1 The Ecological Character of the City Over Time

The greater Melbourne area supports some 1,864 indigenous plant species, of which 178 are considered threatened, and 520 indigenous fauna species, of which 136 of are currently considered threatened (Fig. 20.3). Melbourne also has a very diverse introduced biota. While over 1,100 taxa were recorded in a study of Melbourne's streetscapes, only 76 were indigenous (Frank et al. 2006). It is likely that many thousands more species are cultivated in Melbourne's gardens and parks. The high biological diversity of the city is due principally to three factors: the unique biodiversity of Australia, the diversity of habitats present in the greater Melbourne area, and the way in which development has historically taken place in the city.

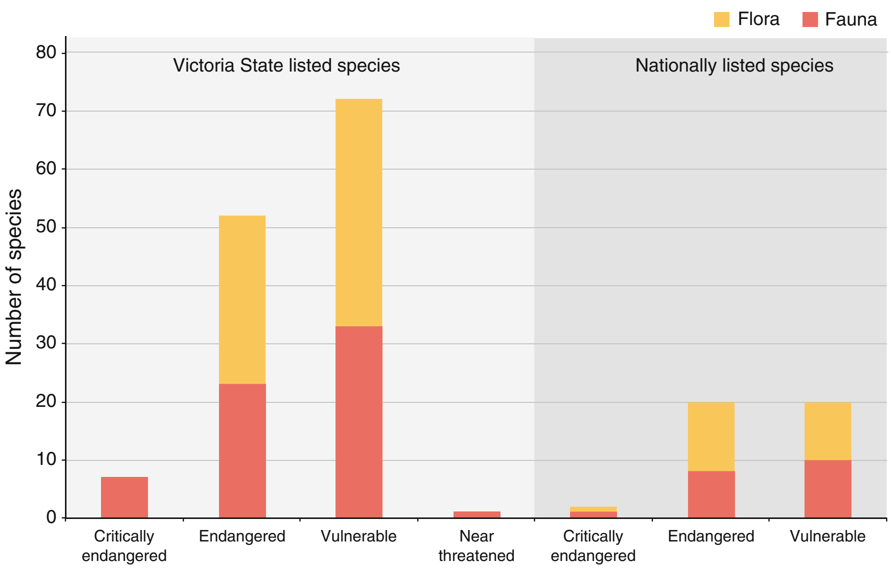

Fig. 20.3 Threatened species present within the Melbourne metropolitan region (Source data from Australian Institute of Urban Studies and City of Melbourne, 2005. "Environmental indicators for Metropolitan Melbourne: Bulletin 8")

20.2.2 Australia and Melbourne's Biodiversity

Australia is home to a diversity of plants and animals found nowhere else in the world as a result of its geographic isolation over time. It is unique floristically, because of the dominance of plant families such as the Myrtaceae, Fabaceae, Casuarinaceae and Proteaceae, and presence of many endemic plant species from the genera Acacia, Eucalyptus, Melaleuca, Grevillea and Allocasuarina. Australia also has an abundance of marsupials while lacking large native predators. Ecologically, the metropolitan area of Melbourne is positioned at the confluence of six bioregions (ARCUE 2009). These environments range from basaltic plains in the west that contain grasslands and woodlands, to low-lying, coastal and alluvial plains in the southeast featuring habitats such as dunes, floodplains and swampy flats. Aquatic, estuarine and marine habitats are also prevalent in and around the city, including the Yarra River and Port Phillip Bay.

20.2.3 Melbourne's Development Over Time

In the first half of the nineteenth century, British cultural influences dominated Melbourne's establishment as a city. Public landscapes were carefully planned and provided large areas of green space. However, as the city expanded at its fringes, many of the natural ecosystems that originally sustained the young city were either cleared or modified dramatically.

Following the discovery of the Victorian goldfields in 1851, population growth and commercial development necessitated an expansion of the city's footprint. The establishment of large parks, broad, tree lined streets and detached and semi-detached housing with front and rear gardens during the late half of the nineteenth century have fundamentally influenced the city's form and function today. After the Second World War, significant population growth, cheap housing availability outside the previously defined metropolitan area, and car ownership resulted in large numbers of people settling further from the city centre (Davison 2004) (Fig. 20.4). This trend was compounded by increasing affluence and a shift towards larger houses and smaller households (DPCD 2004). Indeed, in 1954 only 30 % of Melburnians lived further than 10 km from the General Post Office, compared with 84 % in 2001 (DPCD 2004). The spatial growth of Melbourne over time can be seen in Fig. 20.5.

20.2.4 Biodiversity Responses to City Development

The way in which Melbourne has grown in the past two centuries has enabled a range of indigenous and non-indigenous species to persist in the urban environment. Melbourne has one of the highest percentages of open green space of any city in the

Fig. 20.4 Sporadic development in outer Melbourne in the 1950s (Photograph published with kind permission of the State of Victoria through the Department of Transport, Planning and Local Infrastructure ©Melbourne Metropolitan Board of Works, 1954. All Rights Reserved)

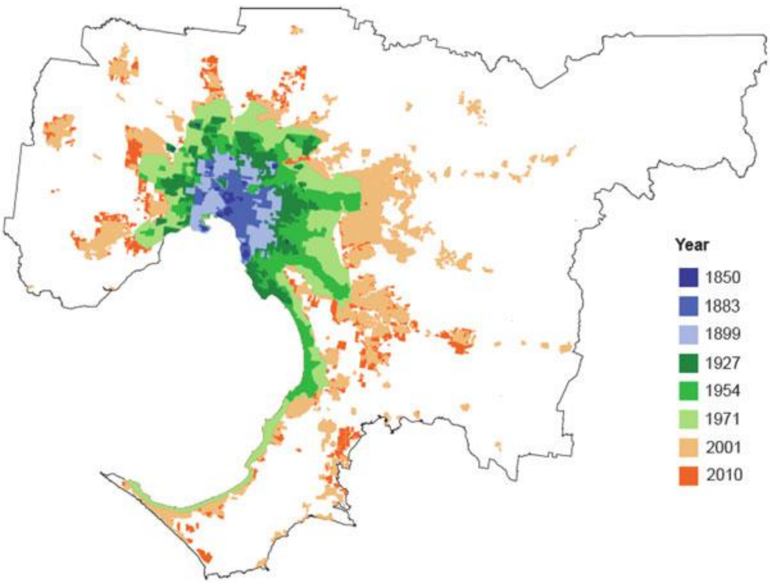

Fig. 20.5 Growth of Melbourne over time (Image courtesy of the Department of Planning and Community Development 2010. ©Department of Planning and Community Development 2010. All Rights Reserved)

Fig. 20.6 Melbourne is known for its high proportion of parks and reserves, such as Royal Park to the north of the CBD (pictured) (Photograph courtesy of Yvonne Lynch and published with kind permission of ©City of Melbourne, 2013. All Rights Reserved)

world (more than 28 %, including Crown road reserves) (VEAC 2011) (Fig. 20.6). This takes a variety of forms, including remnant patches of native vegetation, public parks, residential gardens, and recreational spaces (e.g., sports fields, golf courses) (Leary and McDonnell 2001). Incidentally, these areas have provided valuable habitat for many species as well as providing the ecosystem services characteristic of open space (e.g., recreational opportunities, psychological wellbeing, air and water filtration etc.). Moreover, the low-density, "quarter acre block" suburban development that typifies much of Melbourne allowed vegetation to exist on part of the property. Today, this vegetation helps to support large populations of certain faunal species. Those that have thrived in Melbourne however are generally urban tolerant species that can utilise resources from a wide area (Shukuroglou and McCarthy 2006; Williams et al. 2006; Harper 2005). These include Rainbow lorikeets (*Trichoglossus haematodus*), Grey-headed flying-foxes (*Pteropus poliocephalus*) and Brushtail possums (*Trichosurus vulpecular*).

Although benefiting some species, Melbourne's development has contributed to the endangerment and loss of considerable indigenous flora and fauna, the persistence of which has not been considered in the planning of the city until recently. Four of the most significant pressures impacting upon the indigenous biodiversity of Melbourne are the loss of remnant vegetation for new urban development, the fragmentation of existing patches, the presence of invasive flora and fauna species and inadequate management of native vegetation.

Fig. 20.7 Plane trees lining a busy walkway in central Melbourne (Photograph courtesy of Yvonne Lynch and published with kind permission of ©City of Melbourne, 2013. All Rights Reserved)

20.2.5 Biodiversity and City Culture

As with many other European colonies in temperate climates, British colonists brought a range of familiar plants and animals that thrived in the new environment (Crosby 1986). The soils and climate of Melbourne and the significant environmental gradients from west to east meant that a wide range of plants could be cultivated in Melbourne, from temperate and subtropical species to cold climate species in the eastern ranges. Initially, cultural landscapes were planted with fast-growing evergreen conifers and native Blue Gums, reflecting prevailing European sensibilities (Spencer 1986). However, these species were largely replaced by European deciduous broad-leaf trees such as the London Plane Tree (*Platanus × acerifolia*), and the English Elm (*Ulmus procera*). Today central Melbourne still wears its colonial heritage as a badge of honour, with many grand avenues of this period remaining (Fig. 20.7).

The current biological diversity of the city is undergoing rapid change as a function of habitat loss, population growth, cultural change, climate change pressures, and governance decisions. As the city expands at its fringes, many rare and depleted ecosystems are being placed under increasing pressure. This is evident in the west of the city where temperate native grassland communities are nearing total destruction. Despite being within a national biodiversity hotspot

(Commonwealth of Australia 2011), only approximately 0.2 % of their original extent remains, with half of this in good ecological condition (Australian Academy of Science 2011). Furthermore, these grasslands are home to a number of plant and animal species threatened with extinction such as the Golden Sun Moth (*Symenon plana*), Growling Grass Frog (*Litoria raniformis*) and the Matted Flax-Lily (*Dianella amoena*). Much of the remaining grassland occurs within the peri-urban region of Melbourne and is under serious threat from urbanisation (Commonwealth of Australia 2011).

Population growth is placing pressure not only on fringing ecosystems but also those within the existing city bounds, as infill development places constricts and degrades green space and remnant habitat patches. Recent research suggests that clearing has led to a significant extinction debt in Melbourne's indigenous flora (Hahs et al. 2009). Thus, even without additional habitat loss, future extinctions are likely unless additional effort is put into sustaining flora populations.

Many studies have highlighted the importance of management actions in protecting Melbourne's biodiversity. Indeed inappropriate management has been attributed to the degradation of habitats because of a lack of ecological knowledge of the system or socio-political constraints. For example, it is known that regular burning of grasslands is necessary for the persistence of much of their biodiversity, yet is this is often opposed by the public when it occurs close proximity to residential areas (Carter et al. 2003).

Despite these significant threats to biodiversity, there is a growing trend towards the adoption of "green infrastructure", which can promote biodiversity within the city. Driven in part by environmental regulation, city infrastructure such as roads is increasingly being designed to facilitate movement of organisms between patches of vegetation. Similarly, there are a number of notable riparian rehabilitation projects such as the Merri Creek corridor where the physical restoration of the waterway is associated with restoration of riparian vegetation and in-stream biodiversity (Bush et al. 2003) (Fig. 20.8). Water Sensitive Urban Design principles are also increasingly being adopted by municipalities, increasing biodiversity within streetscapes and benefiting in-stream biota through reducing the hydrological impact of urban development. Biodiversity is also beginning to be incorporated into urban design through features such as green roofs, walls and biodiverse public spaces.

20.3 Institutional Planning, Decision-Making and Governance

20.3.1 Urban Planning

Urban planning in Melbourne is based on a hierarchical system of governance, with the Victorian State Government setting the strategic planning direction for the city, and local governments making decisions about locally significant matters.

Fig. 20.8 Riparian restoration along Merri Creek (Photo by Luisa Macmillan, 2007 and published with kind permission of ©Merri Creek Management Committee. All Rights Reserved)

The principal planning instrument in Melbourne is the 'planning scheme', designed for each municipality. Planning schemes are developed by local governments through consultation with the state government, and integrate spatial zoning, planning policies and overlays to regulate the type and location of development (DPCD 2008). Biodiversity is typically accommodated within planning schemes through specific conservation zones or overlays to protect significant environmental assets.

In the 1970's "Green wedges" were introduced as an official planning priority by the Victorian State Government. This consisted of clearly demarking urban growth corridors and retaining large areas of farmland and bushland in between. However, during the 1990s increasing political concern that these policies were stifling growth resulted in the removal of the planning provisions, enabling new development and a gradual encroachment into the green wedges. More recently, planning strategies that adopt the new urbanism paradigm have been introduced, with an increased focus on protecting green wedges. These include the "Melbourne 2030" and "Melbourne @ 5 Million" strategies (DPCD 2011). However, in practice, many of the strategies designed to protect green areas are failing to be executed effectively due to political and economic pressures (Buxton and Goodman 2003).

The role of private enterprise in influencing the biodiversity of Melbourne is becoming increasingly important. During much of the twentieth century, residential subdivisions were developed by government authorities and many smaller private developers. However, the end of the twentieth century saw the rise of large commercial developers of residential housing that developed very large master planned estates and often developed a number of large projects simultaneously. This has led the state government to respond by undertaking centralised growth area planning, taking some responsibilities from local government authorities (Growth Areas Authority 2013).

20.3.2 Protection of Remnant Indigenous Biodiversity

In Australia, all levels of government (federal, state, and local) are responsible for protecting indigenous biodiversity resulting in a complex interplay of policies and regulations that function at different scales and with different objectives. At the federal level, the Environment Protection and Biodiversity Conservation (EPBC) Act 1999 (DSEWPaC 2012) is the Australian Government's main legislation dealing with the protection of indigenous biodiversity. It is triggered when an action (e.g., land clearing for urban development) is likely to have a significant impact on a "matter of national environmental significance" such as a listed threatened species or community (DSEWPaC 2012). In these cases the Act has the power to stop or limit activities on both public and private land.

At the state level, there are two primary pieces of legislation that regulate the clearing of native vegetation in Victoria. The first is the Flora and Fauna Guarantee Act 1988 (FFG Act; DSE 2012), which focuses on preserving particular threatened species and communities and controlling processes that threaten them. Importantly, emphasis is placed not only on the species themselves but the *habitat* that supports them and the *processes* that have contributed to their demise. In practice the FFG Act has little power to protect threatened species/communities on private land (Lawyers for Forests 2002), limiting its ability to achieve good biodiversity conservation outcomes within urbanised Melbourne.

The second piece of legislation is the Native Vegetation Framework (NVF; DSE 2002). It was introduced in 2002 by the Victorian government and takes a broader approach to managing native vegetation. Unlike the FFG Act, it is primarily focused on private land. The NVF "establishes the strategic direction for the protection, enhancement and revegetation of native vegetation across the State [of Victoria]" and has the goal of achieving a "net gain" in native vegetation, accounting for both area and condition of vegetation (DSE 2002).

The target of "net gain" in vegetation under the NVF has necessitated the development of a range of innovative instruments to implement the legislation and evaluate its outcomes. One of the main components of the NVF is the "Habitat Hectares" metric (Parkes et al. 2003). This provides a repeatable measure of vegetation condition relative to a mature and undisturbed benchmark of the same vegetation type and also incorporates information about landscape context. One of the principal uses of the Habitat Hectares metric is as a "currency" for trading losses (from permitted clearing) with gains from the implementation of biodiversity offsets.

Offsetting biodiversity losses resulting from development actions is becoming increasingly used to achieve the aims of the NVF within the context of continuing urban growth in Melbourne. The offsetting policy within the NVF is based on the mitigation hierarchy of avoid, minimise, and then offset unavoidable losses as a last resort (DSE 2002). In 2013, the Victorian Government introduced reforms to regulations governing permitted clearing of native vegetation (DEPI 2013a). These changes allow 'low impact' vegetation clearing to be exempt from site assessments, with the value of biodiversity present on a site determined via modelled maps of vegetation cover, condition and significance. They also allow proponents to purchase biodiversity offsets via an 'over the counter' fee instead of being responsible for finding the offset site(s) that meet requirement of the NVF. While the new regulations will reduce the regulatory burden for many landholders, the impacts of these changes on Melbourne's native biodiversity remain to be seen. Indeed, a 2008 government report evaluating the previous version of the NVF showed it was failing to achieve its objective of a *net gain* in "both area and condition of vegetation" (DSE 2008). Given that this overarching objective has now been revised to "no net loss" in area extend and condition of vegetation (The State of Victoria 2013), the long-term protection of native vegetation in the face of increasing development pressure from Melbourne remains dubious.

The future of native biodiversity protection in the Melbourne region will however rest to a large degree on the plans developed for proposed urban growth regions. Because of the scale of the proposed development and the presence of nationally listed threatened species and communities in the growth areas, the state government opted for a 'strategic assessment' of Melbourne's growth corridors (DSE 2009) under the EPBC Act. In this approach, impacts to nationally listed threatened species and communities are assessed alongside consideration of state vegetation regulations and plans for new housing and infrastructure in a 'strategic' manner. One of the primary strategies employed within the assessment is the establishment of new conservation reserves to offset future biodiversity losses from development (DEPI 2013b), consisting of approximately 300 ha of threatened native grassland communities (DSE 2009; Gordon et al. 2011). While biodiversity offsetting has already helped reduce the loss of native vegetation associated with recent spatial growth of the city, it appears that the future of Melbourne's native vegetation communities will rest on the efficacy of these offset schemes, especially for native grasslands – one of the most threatened ecosystems in Australia (Williams et al. 2005).

20.3.3 Management of Biodiversity Within the City Landscape

As with planning, there is a hierarchy of responsibility for the design and management of vegetated landscapes in Melbourne. Within the metropolitan region are national parks (regulated by the federal government but managed by state government), state and regional parks (managed by Parks Victoria; a state government authority) and local parks (managed by local government). Vegetation along streetscapes and other public infrastructure is largely also governed by local municipal councils.

From their early beginnings, central Melbourne and some inner suburbs retain a strong European heritage and distinct colonial character. This is evidenced by the characteristic English Elm trees that line many of the large streets. However, recent evidence suggests that traditional non-native species of street trees may be under threat from a changing climate (Kendal 2011). With expected minimum increases in mean annual temperatures of between 2 and 5 °C over the coming century (Ramanathan and Feng 2008) and associated reductions in water infiltration, many local government street tree planting schemes may need review. In contrast to inner Melbourne, some outer suburbs have retained a significant presence of Australian vegetation (McDonnell and Holland 2008; Oates and Taranto 2001). Many of these indigenous species were retained amidst spreading urban land uses and planting of non-indigenous vegetation partly because they were present on land owned and managed by the Melbourne and Metropolitan Board of Works (MMBW) (Brown-May and Swain 2005). The amalgamation of the 'parks' division of the MMBW with the conservation-focused National Parks Service in 1996 resulted in much of this urban parkland land being granted formal protection. Recently, local governments have also invested in 'bush regeneration' programs to restore patches of remnant indigenous vegetation that had become ecologically degraded as a result of processes such as weed invasion, nutrient enrichment and pollution.

20.4 Individual Decision-Making and Governance

Melbourne is comprised predominantly of private land managed by landholders who commonly cultivate plants on their properties. Around the world, cultivated landscapes have been shown to have very high levels of species diversity, often much higher than in surrounding native vegetation (e.g., Thompson et al. 2003), This is the cumulative result of many individual decision makers (Kendal et al. 2010) and is certainly true of Melbourne, where both the biophysical realities and cultural diversity present are reflected in the urban landscapes.

Historical trends in the cultural composition of Melbourne have resulted in changed public perceptions and expectations of urban landscapes. The dominance of detached and semi-detached housing containing a front and rear garden has provided ample opportunities for cultural biodiversity preferences to be expressed via gardening (Head et al. 2004). At the same time, there has been an increase in the popularity of native plants in residential gardens since the 1970s (Elliot and Elliot 2002), reflected in part by the emergence of books and nurseries in that promote indigenous species. Native trees also began to be used in public landscapes after the Second World War (Spencer 1986). Recent changes to planning practices and housing preferences are however resulting in new subdivisions and infill development with minimal private open space (Hall 2010). Consequently, the responsibility for enhancing biodiversity within developed areas is increasingly shifting from private landholders to public authorities.

When it comes to areas of remnant indigenous vegetation, the size of many private lots exempt them from state regulations and few municipal authorities include ordinances controlling the removal of trees on private land. However, private gardens are often voluntarily maintained to promote biodiversity through the cultivation of rare and threatened Australian species (e.g., the Wollemi Pine – *Wollemi nobilis*), or through planting bird-attracting species (e.g., *Callistemon* or *Banksia* spp.). Many local municipalities encourage such actions, with many known to freely give seedlings of native plants to residents.

The ownership of animals has been more tightly controlled than vegetation management, with pets generally requiring registration with restriction over movement off the property, and permits required for some forms of domestic animal ownership. This is especially important for the protection of native fauna, as cats can decimate populations of native mammals and birds and have been shown to roam many kilometres from their home. Native animals are generally protected and their ownership as pets, where permitted, is subject to licensing arrangements.

20.5 Underlying Tensions in Biodiversity Governance

The contemporary governance and institutional structures that influence biodiversity in Melbourne have developed in the context of deep ideological tensions. The two most prominent tensions are between growth and conservation, and between 'native' and 'exotic' species. The growth-conservation tension is expressed most clearly by the planning strategies imposed in Melbourne over time, while the 'native'-'exotic' tension is most clearly expressed in the formulation of conservation policies and approaches to urban landscape design.

20.5.1 Tensions Between Growth and Conservation

The maintenance and health of natural ecosystems has been at odds with city growth in Melbourne from its beginnings. Soon after the township was settled, the ecosystems that had originally made it suitable for human occupation through provision of good soils, timber and clean water were viewed as a constraint to its further development. Yet because of the abundance of resources elsewhere in the landscape, there was little attempt to preserve or integrate natural ecosystems into the city. Indeed, until recently, protection of biodiversity within formal planning instruments was incidental and ad-hoc. For example, large areas of native vegetation were originally reserved as land for freeways or retarding basins by the Melbourne and Metropolitan Board of Works (Brown-May and Swain 2005) but are now valuable for biodiversity as most surrounding native vegetation has been cleared or modified.

The planning systems that have guided Melbourne's development demonstrate the tension between urban growth and conservation. Historically, the fluctuation between growth and conservation priorities at a state level (e.g., the strengthening and weakening of the urban growth boundary) demonstrate a struggle to reconcile the two ideas politically, while the presence of clearly marked areas for biodiversity conservation and development in current planning schemes highlight the spatial separation of the two concepts. This dynamic is complicated all the more by a hierarchical planning structure whereby federal, state and local governments will very often have differing views on the relative priorities of growth and conservation. Worryingly, with the responsibility for the design of new developments largely falling to private companies, rarely are attempts made to protect or promote biodiversity outside of clearly demarked "conservation" areas.

Another area that highlights the tension between growth and conservation in Melbourne is the management of bushfire and the presence of dense eucalyptus forests on the urban fringe. Eucalypts are well adapted to fire, having evolved to possess volatile oils and decorticating bark that promote it. Periodic wildfire (bushfire in the local vernacular) has resulted in large scale loss of life on the urban fringes, the most devastating of which being the 2009 "Black Saturday" bushfires on the 7th of February, with 173 fatalities. Changes to planning schemes in response to these events have permitted much greater removal of native vegetation near housing in some areas despite its biodiversity value.

20.5.2 Tensions Between 'Native' and 'Exotic' Species Conservation

To the early settlers of Melbourne, the unique flora and fauna of Australia differed in appearance and perceived usefulness from the European plants and animals they were familiar with (Figs. 20.9 and 20.10). From this time onwards, a tension has existed about how to manage both indigenous and introduced species of plants and animals within the city. From an institutional governance perspective, the separation of "native" and "exotic" forms of biodiversity has resulted simultaneously in the development of strong and progressive legislation to conserve threatened indigenous species and ecological communities, and confusion about the role and structure of biodiversity in 'cultural' landscapes.

The strong legal protection of indigenous vegetation, while essential from an ecological perspective, suggests that 'native' biodiversity is viewed separately from 'introduced' plants and animals present in Melbourne. Indeed, this tension between native and exotic landscapes is entrenched in the management structure

Fig. 20.9 Native woodland vegetation typical of the north and west of Melbourne (Photo by ©Ascelin Gordon, 2005 and published with his kind permission. All Rights Reserved)

Fig. 20.10 An example of a European style cultivated garden, common in Melbourne (Photo by ©Dave Kendal, 2009 and published with his kind permission. All Rights Reserved)

of many local governments, with a separate "bush crew" managing areas of native vegetation while horticulture teams manage green space. Investment by local governments in "bush regeneration" programs often runs alongside street planting policies that promote non-native species. However, the retention of these species has been challenged, particularly during the recent drought, with some calling for the planting of native species that use less water. Indeed, the debate over which type of trees to plant along streets and in gardens has been picked up in the media (e.g., The Age 2006), suggesting that the 'native'-'exotic' tension is present not only in formal governance institutions but also in the culture and minds of Melburnians.

20.6 Future Directions for Melbourne's Biodiversity

The coming decades are a critical time for the future of biodiversity in Melbourne. Decisions made within the next 30 years are likely to influence biodiversity outcomes long into the future. As already discussed, the current state of biodiversity assets in the city is the result of a series of ad-hoc decisions and serendipitous events that unintentionally led to a high diversity of plants, animals and communities, and a city that is pleasant to live in. However, continuing to make decisions in this way is unlikely to achieve good biodiversity outcomes in the future, as population pressures and urban development continue to impinge upon the very factors that made it appealing for human residence from the outset. As a young city, Melbourne is positioned favourably to learn from other cities around the world and build towards a future that contains both a healthy human population and flourishing biodiversity. The degree to which this is achieved will depend largely on decisions made and actions taken in four arenas: (i) urban growth on the fringe of Melbourne, (ii) habitat management in established areas, (iii) management of green assets, and (iv) promotion of biodiversity on private land.

As mentioned above, the way in which population increases are accommodated within the city will affect the future of Melbourne's biodiversity. Although infill development may threaten the biodiversity present within parks and backyards, continued expansion of suburbs at the fringes of the city will have disproportionately large impacts on indigenous biodiversity that is not accommodated elsewhere within the city. The enforcement of a growth boundary at the fringes of Melbourne is therefore likely to result in a scenario of high biodiversity conservation values on the edge of the city, whereas a relaxing of this boundary will result in a scenario of continuing biodiversity loss in this area, regardless of biodiversity offset policies. Moreover, the style of suburban development being produced by large private development companies on the city's fringes could lead to a gradual homogenisation of biodiversity where a small number of plant species are used in street and landscape plantings in master-planned estates.

Management of extant habitat in existing suburbs is another critical arena that will influence biodiversity outcomes in the future of Melbourne. Many remnant habitat patches are at risk of serious degradation if not actively managed according to best available scientific knowledge. The looming threat of an extinction debt (Hahs et al. 2009) demonstrates clearly the challenge Melbourne faces in retaining existing levels of native species richness in the urban landscape over time. Failing to recognise and manage Melbourne's current biodiversity assets and their threats will result in continued decline in biodiversity and a future scenario of degraded ecosystem function. However, actively mitigating the impacts of urban pressures such as edge effects, weed invasion, pollutants and predation by introduced species can help create a future where the biodiversity and ecological function of remnant habitat patches are maintained and enhanced.

A third arena that will determine the future biodiversity of Melbourne is the way in which 'green infrastructure' assets are created and managed. These are anthropogenic features within the city that contribute significantly to biodiversity but do not constitute remnant native vegetation. They include features such as street trees, public parks, gardens, median strips, ponds and swales (Figs. 20.11 and 20.12). Since these features are not designed primarily for biodiversity, there is a risk that the biodiversity benefits they do provide may be degraded unintentionally over time if not carefully monitored and cared for. Often, they are managed for aesthetic and public health and wellbeing outcomes. Research linking biodiversity conservation with public health benefits and human wellbeing may therefore help to conserve and increase biodiversity in these landscapes. Melbourne has a good platform for the enhancement of biodiversity within the metropolitan region, due to the presence of large areas of parkland. However, as the population continues to grow, green assets must be valued and integrated with new urban forms.

Many of the biodiversity outcomes in a city are the result of local actions; therefore local governments in Melbourne have a strong role to play in the creation of neighbourhoods that promote biodiversity. If local governments in Melbourne adopt a holistic view of biodiversity in their legal instruments and policies (Ives et al. 2010), this will help to break down the potentially destructive dichotomy between 'native' and 'exotic' biodiversity. Similarly, governments that appreciate more fully the interrelationships between biodiversity and human wellbeing are more likely to find 'green' solutions in everyday planning and infrastructure decisions, thus helping to promote biodiversity in the city into the future. Greater integration of environmental policies with other regulatory instruments will also help to achieve this. Studies such as McConnell and Walls (2005) and Bowman et al. (2009) have demonstrated that a financial premium can be justified for housing located near to or integrated with areas of high ecological value. Therefore opportunities should be explored for integrating biodiversity and ecosystem function into residential areas.

Figs. 20.11 and 20.12 Green infrastructure such as green roofs and vegetated swales can contribute significantly to the biodiversity of urban landscapes, while simultaneously offering other environmental and social benefits (Photos courtesy of Yvonne Lynch, and published with kind permission of © City of Melbourne 2013. All Rights Reserved)

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Chapter 21
A Synthesis of Global Urbanization Projections

Michail Fragkias, Burak Güneralp, Karen C. Seto, and Julie Goodness

Abstract This chapter reviews recent literature on global projections of future urbanization, covering the population, economic and physical extent perspectives. We report on several recent findings based on studies and reports on global patterns of urbanization. Specifically, we review new literature that makes projections about the spatial pattern, rate, and magnitude of urbanization change in the next 30–50 years. While projections should be viewed and utilized with caution, the chapter synthesis reports on several major findings that will have significant socioeconomic and environmental impacts including the following:

- By 2030, world urban population is expected to increase from the current 3.4 billion to almost 5 billion;
- Urban areas dominate the global economy – urban economies currently generate more than 90 % of global Gross Value Added;

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- From 2000 to 2030, the percent increase in global urban land cover will be over 200 % whereas the global urban population will only grow by a little over 70 %. Our synthesis of recent projections suggest that between 50%–60% of the total urban land in existence in 2030 will be built in the first three decades of the 21st century.

Challenges and limitations of urban dynamic projections are discussed, as well as possible innovative applications and potential pathways towards sustainable urban futures.

21.1 Introduction

Urbanization, being a process of simultaneous demographic, economic and biophysical change (Chap. 1), is a prime area of exploration for scholars and practitioners involved in quantitative forecasting, projecting and future scenario building. Population projections for 2100 forecast total world urban population to grow by 3–5 billion (UN 2011b). Other projections suggest that most of the urban growth is expected to take place in small and medium sized cities of one million or fewer (Montgomery 2008). What all projections show is that urbanization is occurring faster and at larger volumes in locations that are at lower stages of economic development and face rapid demographic changes (Angel et al. 2005). Future urbanization will be characterized by unprecedented magnitude and high rates of change, thus distinguishing it from past urban transitions.

Global urbanization projections are needed since cities have become dominant entities in the world's social, economic, cultural, political, and environmental spheres. Urban areas dominate the global economy – the economies of cities currently generate more than 90 % of global Gross Value Added (UN 2011a). As global centers of production and consumption, urban areas rely on resources and ecosystem services, from construction materials to waste assimilation, that are distributed around the world. A better understanding of the urban growth process globally is important due to the significant and far-reaching socioeconomic and environmental impacts of urbanization. The impact on the environment comes at multiple scales including regional precipitation patterns (Kaufmann et al. 2007), loss of wildlife habitat and biodiversity (McKinney 2002), conversion of agricultural land (Seto et al. 2004), increase in air pollution coupled with increased automobile dependency and congestion (Boarnet and Crane 2001), and greater demand for water, energy, and agricultural resources (Johnson 2001); see also Chaps. 3, 25 and 26 in this volume. A better understanding of urban growth processes and urban morphology will allow us to better respond to global environmental change.

The size and scale of urban population growth, levels of income and the concomitant urban land-use changes pose major challenges to local and regional ecosystems, and ultimately the global environment. Two of these challenges stand out: (1) the location of urban development – whether in low-lying coastal zones, in agricultural areas, in forested regions, or near existing urban centers – affects the vulnerability of cities to climate change impacts such as sea level rise and storm surges (Chap. 25), the need to expand agricultural production into other areas, and the resources required to provide municipal services such as water, energy, and transportation infrastructure (see also Chaps. 3 and 22); (2) the way in which urban development occurs – whether expansive or compact, with multi-family or single family homes, automobile dependent or enabling multiple forms of transportation, with mixed-use or single-use zoning – affects transportation choices and travel behavior, determines infrastructure needs and energy consumption, and shapes the urban social fabric. Expansive urbanization leads to fragmented wildlife habitats and biodiversity loss, altered hydrological systems and local climates, and substantial changes in energy and nutrient cycles (see Chaps. 3 and 10).

Historically, researchers began exploring global urbanization through urban population employment and transportation models based on equilibrium and comparative statics (Batty 1976). We now have available a wide variety of models that attempt to capture aspects of the dynamic process of urban population, economic and land-use change; these models have been developed within various disciplines that adopt distinct methodological lenses (Batty 2005). Throughout the last six decades, models have reached a significant level of maturity, have exhibited successes in terms of their usefulness, and have become increasingly popular.

Land-use change models in the 1950s were concerned primarily with local areas or regions; the majority of the research conducted in this field remains a narrowly focused activity within specific urban regions. It was not until 2011 that the first global models – and the full picture – of urban land-use change emerged. The forecasts from these models complement the population and wealth projections. Modeling all dimensions of urban dynamics is now important for integrative research that is conducted on the (global) environment front – energy use, greenhouse gas emissions, heat island effects, and alterations in the natural nitrogen, carbon, and water cycles. For example, modeling land-use futures is now viewed as a fundamental activity for projecting the future health of the natural, human, and social systems locally and globally (Solecki et al. 2013).

This chapter synthesizes important parts of the recent literature on expected or projected patterns of change in urban population, wealth, and physical expansion that emerge from efforts of quantitative modeling of urbanization dynamics. Taken together, the projections provide a comprehensive view of possible global urban futures. Note that the implications of the reported global urban expansion projections for biodiversity and ecosystem services are reported in Chap. 22.

21.2 Population Projections for Urban Areas

Population projections exist at a variety of scales (spatial and temporal), involve a wide array of population characteristics (age, sex, etc.) and are utilized towards various goals (business planning, policy at various administrative levels, etc.)

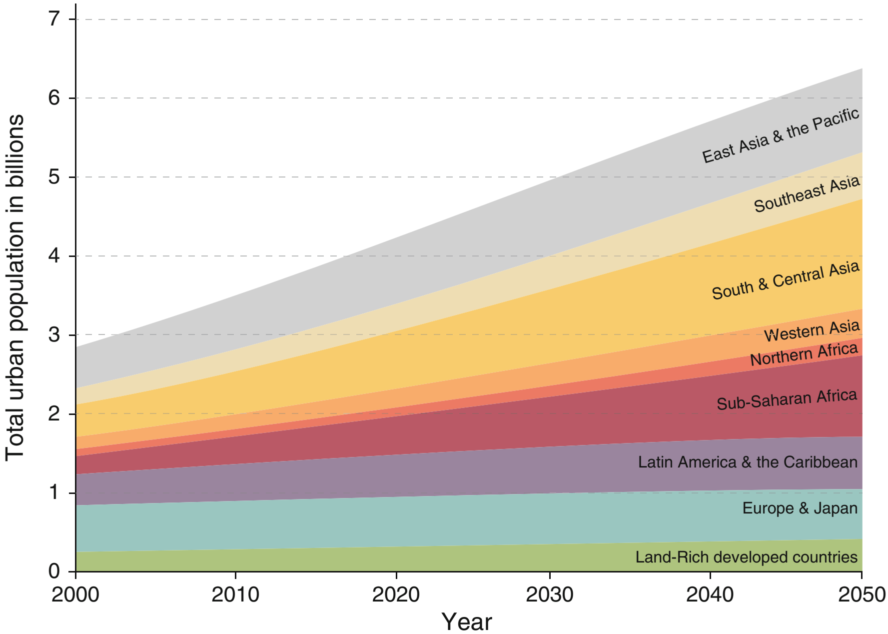


Fig. 21.1 Urban population projections for different world regions. Note that regional categories may be treated as cumulative, whereas the "land-rich developed countries" category should be regarded as separate (Data from Population Division of the Department of Economic and Social Affairs of the United Nations Secretariat 2011b, 2012b, accessed March 21, 2013)

In this subsection we focus primarily on the urban population projections provided by the United Nations, a historical leader and currently the sole provider of global urbanization projections (UN 2012b). A more comprehensive overview of global urban population projections is provided in O'Neill et al. (2001).

The most recent United Nations World Urbanization Prospects report pinpoints that more than half of the world's population now lives in cities compared to 30 % 50 years ago and 10 % 150 years ago (Fig. 21.1); see Chap. 3 for a historical and present examination of trends associated with urbanization and biodiversity. Between 2011 and 2050, the world population is expected to increase by 2.3 billion, growing from 7.0 billion to 9.3 billion (UN 2011b). At the same time, urban areas are expected to gain 2.6 billion people, rising from 3.6 billion in 2011 to 6.3 billion in 2050 (UN 2012b; Montgomery 2008). Importantly, most of the future world population growth up to 2030 is projected to occur in the rapidly growing cities of poor African and Asian nations (around 80 % of the total) as well as in Latin America. Africa and Asia today are urbanizing more quickly and at a larger magnitude respectively than the rest of the world's regions (UN 2012b). While we expect increasing numbers of megacities (i.e., cities with population of over ten million people), they are expected to contain approximately the same proportion of the world's urban population – around 15 %. The majority of future urbanites will live in rapidly growing medium-sized or small developing-world cities, subject to many present-day urban pathologies. Not only will urban areas of primarily medium size absorb the majority of future urban growth, but the majority of the new urban residents are expected to be poor (Martine et al. 2008). Poverty is increasingly becoming an urban phenomenon. While slum dwellers already constitute about 32 % of urban population in the developing regions of the world, urban growth in certain regions will come about with the formation of new slums (UN-Habitat 2012).

Much has been written about the demographic characteristics of contemporary urbanization at regional and global scales (Chap. 3). The world is currently experiencing massive demographic changes through differing rates of natural increase and net migration (Cohen 2004). These changes are crucial in terms of linking up the geopolitical realities and also prospects for rapid transitions to low carbon cities and economies in the respective contexts. In the span of the two most recent centuries, the number of cities with populations of one million or greater grew from 1 to 442 in 2010 (UN 2012b). As of 2011 there were 45 such cities in India and 88 in China. By 2025, there will be more than 600 cities of one million or more worldwide.

Over the next two decades, the combined urban population in China and India will grow by more than 700 million (UN 2012b). China's urban population is expected to increase by 400 million and India's urban population will nearly double from today's 350 million (UN 2012b). During this same period, China will create at least 30 new cities of one million; India is expected to add 26 cities of this size. The urban transitions underway in these two countries represent the largest urban transition in history. Put into a global context, by 2030, nearly one-third of the world's urban inhabitants will live in either China or India. The impacts of the growth of urban population on natural habitats are projected to be significant in both countries (see Chap. 22).

Population projections for the World Urbanization Prospects are based fundamentally on understanding the historical trajectory ratio of urban to rural population in a particular country and extrapolating that trend into the future. The UN defines the growth of this ratio as the "urban-rural growth difference" (URGD) since it is equal to the difference between the urban and rural growth rates. Historically, URGD is higher in countries that are less urbanized and declines as the level of urbanization increases. For the creation of projections, the UN has used cross-sectional data to define a global association between URGD and the level of urbanization in a country – a global norm. Following that first step, each country is modeled for the next two decades as moving from its current URGD to the global fitted value that is derived from its current level of urbanization. All countries are modeled as following the global norm after the first two decades. The methodology is designed so that the rate of increase in the proportion of urban population in each country slows down as the country becomes more urbanized. Once a sequence of rates is established, the UN uses the projections of total population for each country to produce the projections for urban and rural populations.

Urban population projections for the world and across regions provided by the UN have also been criticized and various amendments have been proposed (Cohen 2004; Bocquier 2005; Lutz and Samir 2010). Among the several complaints, the UN has consistently projected growth rates that are too high and the forecast errors are large for the 20-year- and 10-year-ahead forecasts (Montgomery 2008). Probably, the most well-known issue with the UN historical population dataset is the lack of a consistent single definition of "urban" population across countries (UN 2012b). The UN adopts the definition of the individual country for measuring sizes of urban populations – producing a rather inconsistent mosaic across the world. This problem is compounded by issues of urban/rural dual classification, the selection of methodology, and various demographic assumptions. Many researchers suggest today that a good deal of caution with projections is indeed necessary. Box 21.1 describes some of the limitations that originate in the choices that the producers of projections have to make on methodologies, assumptions, and reduced capacity for accuracy assessments.

Box 21.1 Further Discussion of UN and Other Projections

In this box, we briefly present main themes that appear in debates over data and methodologies utilized for generating global population projections.

Historical Data: UN data lack a consistent single definition of "urban" population across countries and rely on individual country definitions of "urban" (UN 2012b). Countries like China, for example, utilize administrative boundaries across distinct scales to define urban populations. But in countries like the U.S., criteria on total population and density come strongly into play for the "urban" definition. The World Urbanization Prospects website suggests that a certain attention to corrections across time for within-country definition variation does exist. Definitions used are typically the ones that are used by national statistical offices which use them for the purposes of their latest census. Changes in definitions are dealt with in data processing and adjustment with the target of consistency: "*United Nations estimates and projections are based, to the extent possible, on actual enumerations. In some cases, however, it was desirable to incorporate official or other estimates of urban population size*" (UN 2012b). The UN aims for transparency in all processes that involve adjustments or alternative sources of data. In the future, other features of urbanization may appear most appealing for an integrated picture of global urban populations where "*[…] characteristics such as the proportion of labor force employed in non-agricultural activities or the availability of urban facilities such as water or sewer systems may be used*" (O'Neill et al. 2001).

Cohen (2004) reports that the urban-rural duality still dominates classification systems even though the structures and processes of population organization into settlements requires a radically new approach:

> "While it has long been recognized that the conventional division between rural and urban is a gross oversimplification of the underlying complexity of today's human settlement systems, in reality it is still the only one that is usually available."

(continued)

Box 21.1 (continued)

New geospatial technologies "*may enable researchers to link large amounts of data of different kinds and to develop more sophisticated conceptualizations and measurements of the dimensions of settlement systems* (Hugo et al. 2001)." These observations notwithstanding, the research and policy community cannot simply rely on a single binary method of human settlement pattern measurement due to a growing complexity and shifting interests and agendas of decision-makers; "*other criteria, such as population density or the degree of accessibility (or remoteness) of a particular location may also have to be better defined and measured* (Coombes and Raybould 2001; Hugo et al. 2001)."

Methodology: Urban projections are critically dependent on total population projections and their assumptions. The World Population Prospects 2010 Revision now suggests that the past expectation of total world population stabilization at nine billion people by 2050 could be too optimistic. Armed with a new understanding of how fertility is slowly declining in particular poor countries and slightly rising in several wealthier countries such as the U.S. and the U.K., the new projections suggest that the world population could plausibly grow to 10.1 billion by the year 2100. The question of constructing reliable population projections is still very much at the forefront of demography. Godfray et al. (2010) discussing Lutz and Samir (2010) state that,

> "*[p]opulations in different countries are assumed to be composed of different age cohorts of the two sexes that vary in demographic rates such as mortality and fertility.*
>
> *Models can be extended to include differences between rural and urban populations (connected by migration) and, most importantly, educational status. There is very convincing evidence of the critical importance of female education and access to contraception in causally affecting fertility, and these are probably the chief mechanisms behind the decline in fertility as countries develop economically and go through the demographic transition. Of particular relevance here is evidence that education rates are also negatively correlated with malnutrition and food insecurity*."

The handling of uncertainty depends on a wide variety of end goals of a projection/forecasting effort. Lutz and Samir (2010) list four strategies: to ignore it, to construct scenarios, to explore a plausible range of variation, and to make fully probabilistic projections; they eventually advocate for the generation of probabilistic projections:

Specific sustainability challenges require potential adjustments in use of assumptions in projections. In some cases, such as the study of future food demands, researchers recommend the use of 'medium fertility' scenarios (1.8 children per adult female). Note though that this number overshoots

(continued)

Box 21.1 (continued)

fertility rates in China; slight variations in China can have a big impact in projections overall. According to Godfray et al. (2010),

> "*[w]ith this adjustment, global population growth is predicted to decelerate and reach just over nine billion in 2050.*
>
> *There are marked regional variations: Europe's population will decline, Africa's will double, while China will peak in about 2030 and be overtaken by India around 2020. Populations will age almost everywhere, but as the old will be healthier, rethinking age in terms of time to expected death (rather than time since birth) may give a different and more positive perspective on increased longevity.*"

These projections may mean little if they are disconnected with the overall regional and national economic growth patterns.

Accuracy: Accuracy, reliability and robustness of projections are naturally desirable properties of the work of demographers and typically in high demand by planers and policymakers (Cohen 2004). Unfortunately, on the ground, the realities are far from perfect. Most projections are never validated. O'Neill et al. (2001) reports on growth projections globally that are too high while assessing various projections of the UN and the World Bank since the early 1970s. Cohen (2004) also points out the past "fairly spectacular errors" that are made in publications that, in their majority, project an unchecked future urban change in developing countries and their largest urban agglomerations. In a systematic analysis of past projections, he finds a bias towards high projections, primarily due to the fact that the models employed missed the more rapid than expected drop in rates of fertility. Also,

> "*there has been considerable diversity in the quality of urban projections by geographic region, level of economic development, and size of country. On average, the UN urban projections have been most reliable for OECD and least reliable for countries in sub-Saharan Africa and for other high income countries, many of which are quite small. UN projections also tend to be better for larger countries than for smaller countries, probably because they receive more attention*" (Cohen 2004).

A good deal of caution with projections is indeed necessary.

21.3 Economic Projections for Urban Areas

Long-range projections for economic performance have a long history in various disciplines and are expressed in terms of national economic accounts, measuring potential growth of economies and regions (Colm 1958). Global income projections have become a bread-and-butter operation for a large number of international organizations, governance bodies, think tanks, investment firms and nonprofits

(TCB 2013; IMF 2013); not only are these reports produced annually but they are also updated frequently. As cities have become dominant entities in the world's social, economic, cultural, political, and environmental spheres, the question of measurement of economic activity at the city level has become a significant research area. Recent research reveals that urban areas dominate the global economy – they produce more than 90 % of the world's GDP (UN 2011a). It is therefore surprising that global projections for even fundamental measures of economic activity are nearly non-existent.

Academic research has shown that the level of urbanization and income per capita are highly correlated. Henderson (2010) finds that the level of urbanization explains 57 % of the variation in levels of income per capita. Additional variation can be explained by the different definitions of urbanization across countries. Others suggest that the role of urbanization is more nuanced; Bloom et al. (2008) claim that while levels of income across the globe are highly correlated with the proportion of a country's urban population, they find no evidence that the level of urbanization affects the rate of economic growth.

The McKinsey Global Institute (MGI) recently developed urban economic projections for the world's largest 600 metropolitan areas (MGI 2011). In a report titled "Urban World: mapping the economic power of cities," MGI suggests that in the near future, the list of the world's largest 600 urban economies will be substantially different since "the center of gravity of the urban world moves south and, even more decisively, east." The projections suggest that the largest 600 urban economies will maintain their 60 % share of the total (global) GDP. One third of the developing countries that are currently on the list will not be there in 15 years. About 1 out of 20 cities in emerging economies will drop out of the top 600 list. All of the new cities entering the list of the top 600 urban economies in the near future will be developing world cities and overwhelmingly (100 new cities) from China (Fig. 21.2).

Urban areas show significantly higher levels of wealth because of higher productivity levels. However, a big challenge in urban research is the lack of measures of GDP growth at the urban scale (Fig. 21.2). It is estimated that 30 % of national GDP in the United Kingdom, Sweden, Japan, and France is accounted for by London, Stockholm, Tokyo, and Paris, respectively (Seto et al. 2010). Globally, metro areas drive their national economies, but there are significant disparities in the GDP per capita between and within the world's urban areas (Seto et al. 2010). There is even a bigger disparity between the wealthy and the poor in cities, and this disparity is exacerbated by the scale and rapidity of change. Economic development and improvements in well-being are only part of the urbanization story: Worldwide, an estimated 863 million people currently live in informal settlements, with most living under life- and health-threatening conditions. Put another way, approximately one out of three urban dwellers worldwide lives in slum conditions, and this ratio is expected to increase in the future (UN 2012a; UN-Habitat 2012). In light of their importance locally and regionally, and considering their size globally, global urbanization projections need to explicitly incorporate approaches that consider informal settlements.

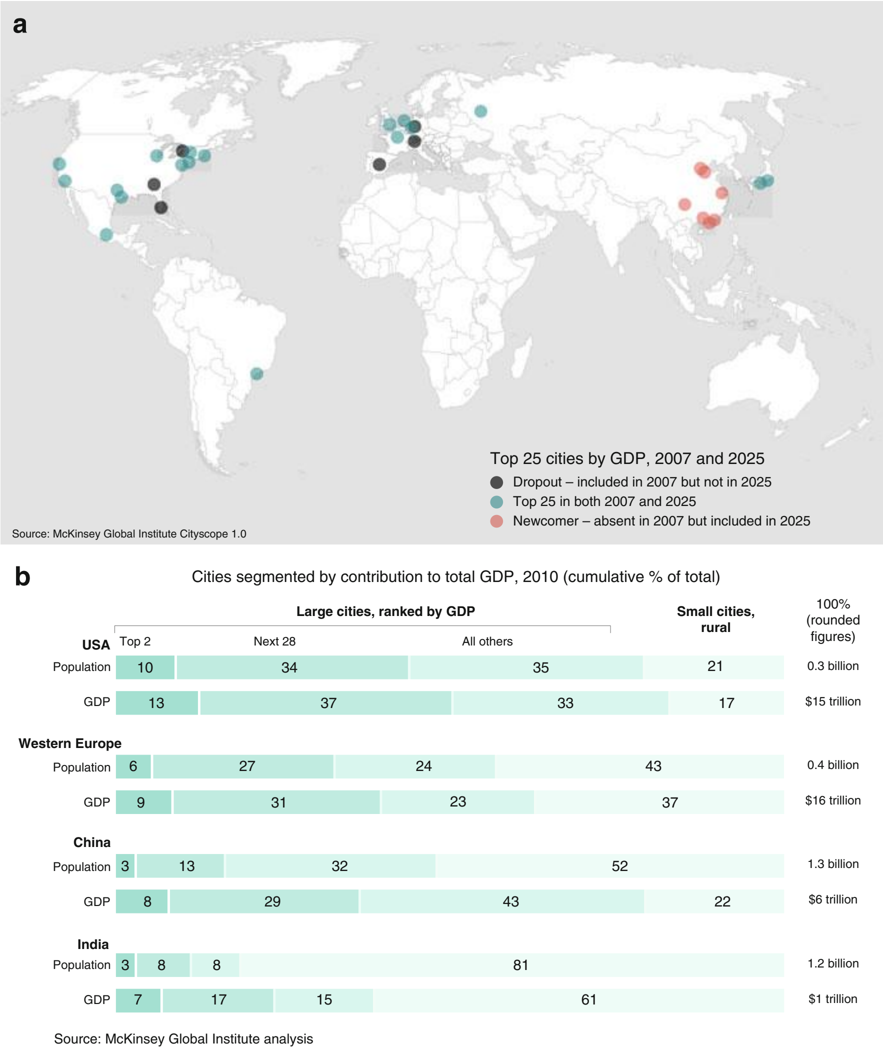


Fig. 21.2 (**a**) Shift in urban economic weight towards Asia (Data from McKinsey Global Institute 2011); (**b**) Cities segmented by contribution to total GDP, 2010, cumulative percent of total (Data from McKinsey Global Institute 2012)

21.4 Physical Expansion Projections for Urban Areas

As emphasized in the previous section, our view of global urbanization rates and magnitude has up until now been focused solely on measures of urban population. Through two major research efforts that culminated in 2011, and another in 2012, we now better understand the global patterns of actual built-up urban land rates of growth during the last four decades. The studies also provide us with a “window into the future” through scenario and projection exercises. This section synthesizes the most recent findings discussing physical expansion futures.

21.4.1 Meta-analysis Projections

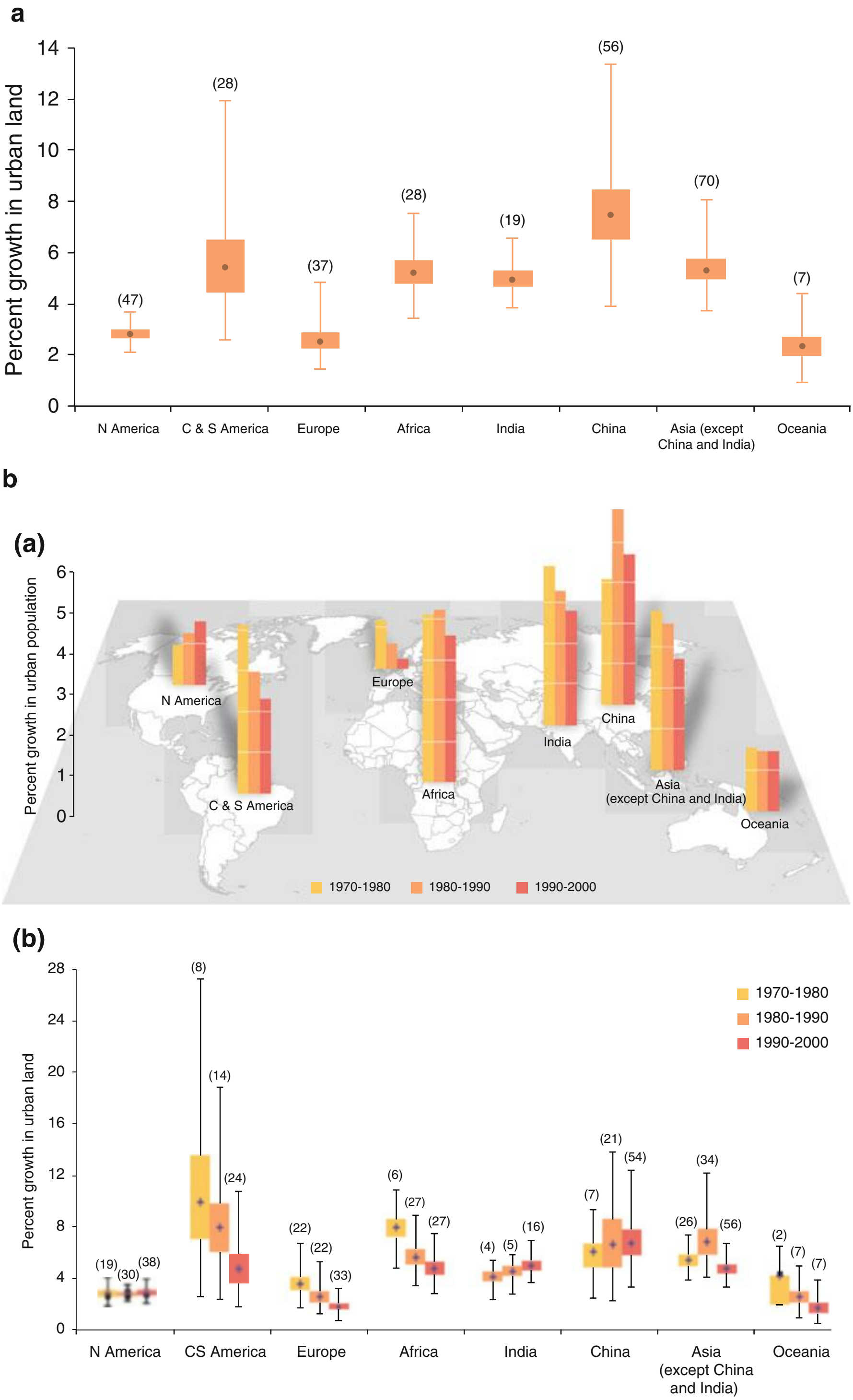
In a recent meta-analysis of 326 studies that have used remotely sensed images to map urban land conversion, Seto et al. (2011) report that between 1970 and 2000 urban areas grew by 58,000 km^2 worldwide, and by 2030, cities are expected to grow by more than another 1,527,000 km^2 – nearly the size of Mongolia. This growth in urban land area is equivalent to 1.3 times the size of the country of Denmark, or approximately 1.56–3.89 % of the global terrestrial area in 2000. India, China, and Africa have experienced the highest rates of urban land expansion, and the largest change in total urban extent has occurred in North America. Across all regions and for all three decades, urban land expansion rates are higher than or equal to urban population growth rates, suggesting that urban growth is becoming more expansive than compact.

Annual growth in GDP per capita drives approximately half of the observed urban land expansion in China but only moderately affects urban expansion in India and Africa, where urban land expansion is driven more by urban population growth. In high-income countries, rates of urban land expansion are slower and increasingly related to GDP growth. However, in North America, population growth contributes more to urban expansion than it does in Europe. Much of the observed variation in urban expansion was not captured by population, GDP, or other variables in the model. This suggests that contemporary urban expansion is related to a variety of factors difficult to observe comprehensively at the global level, including international capital flows, the informal economy, land-use policy, and generalized transport costs. Using the results from the global model, the authors developed forecasts for new urban land cover using the Special Report on Emission Scenarios (SRES) by the IPCC. Their projections show that, depending on baseline urban extent for 2000, by 2030, global urban land cover will increase between 430,000 and 12,568,000 km^2, with an estimate of 1,527,000 km^2 more likely.

The historical results of the Seto et al. (2011) study show considerable variation in the rates of urban expansion over the study period (1970–2000), with the highest rates in China followed closely by Asia and Central/South America (Fig. 21.3). Average rates of urban expansion are lowest for Europe, North America, and Oceania. Variations in urban expansion rates point to differences in national and regional socio-economic environments and political conditions. This is particularly evident in the case of China, where annual rates of urban land expansion vary from 13.3 % for coastal areas to 3.9 % for the western regions. On the other hand, the range of urban growth rates in North America is more evenly distributed, from 3.9 to 2.2 %.

Reported total urban land conversion was highest in North America, but this could reflect a sampling bias because 16 % of the urban areas in the meta-analysis were located in North America. Indeed, the geographic distribution of the meta-analysis case studies indicates that some of the largest cities worldwide are not being studied in terms of their changing urban land extent. In particular, five of the world's largest cities by population, Dhaka, Karachi, Kolkata, Jakarta, and Delhi, were not represented in the meta-analysis case studies.

About 34 % (99 out of 292) of the locations in the meta-analysis fall within 10 m of low elevation coastal zones (LECZ). For these urban areas, the average rate of


a
Percent growth in urban land
14
12
10
8
6
4
2
0
(47)
(28)
(37)
(28)
(19)
(56)
(70)
(7)
N America
C & S America
Europe
Africa
India
China
Asia (except China and India)
Oceania
b
(a)
Percent growth in urban population
6
5
4
3
2
1
0
N America
C & S America
Europe
Africa
India
China
Asia (except China and India)
Oceania
1970-1980
1980-1990
1990-2000
(b)
Percent growth in urban land
28
24
20
16
12
8
4
0
(19)
(30)
(38)
(8)
(14)
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1970-1980
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N America
CS America
Europe
Africa
India
China
Asia (except China and India)
Oceania

urban land expansion from 1970 to 2000 is greater than 5.7 %, and statistically higher than urban areas elsewhere. Given the impacts of climate change and projections of geographically uneven levels in sea level rise and storm surges, our results show that humanity has unknowingly been increasing the vulnerability of its urban populations (see also Chap. 25). Almost half of the meta-analysis case studies (47 %) are within 10 km of a terrestrial protected area with IUCN status listed in the World Database of Protected Areas. Taken together, these results show that urban land expansion is as likely to take place near protected land as elsewhere, and that being near a protected area does not necessarily slow the rate of urban land conversion (see Chaps. 3 and 22).

Across all regions and for all three decades, urban land expansion rates are higher than or equal to urban population growth rates (Fig. 21.3b). Nowhere is there evidence of a global increase in urban land-use efficiency or urban population density (as defined by the change in urban population per unit change in urban land); these trends suggest an expansive urban growth globally. Rates of urban land expansion by decade reveal three distinct urbanization trajectories: strong declining annual rates across the decades (Central and South America, Europe, Oceania, and Africa), weak positive or stable trends (China, North America, and India), and uneven trajectories (Asia - except China and India) (Fig. 21.3b). Declining rates of urban land expansion are expected for regions such as South America and Europe, which were already highly urbanized (in terms of percentage of population living in urban areas) in the 1970s, with urban population levels of 57 and 63 %, respectively. In contrast, declining rates of urban land change are surprising for Africa, where urban population levels were only 24 % in 1970. While Africa has consistently higher average rates of urban land expansion than North America, the total urban extent is greater in North America.

The authors of the Seto et al. (2011) study developed four urban land expansion scenarios based on the Special Report on Emissions and Scenarios (SRES) Scenarios available through CIESIN (http://sres.ciesin.columbia.edu/). The four SRES Scenarios, A1, A2, B1, and B2, were generated at the UN regional level for 2050 based on the global population and GDP projections. The A1 storyline is characterized by high economic growth and low population growth; the A2 storyline is characterized by lower economic development and high population growth; storyline B1 is considered a "sustainable development" scenario with moderate economic growth and

Fig. 21.3 (**a**) Average annual rates of urban expansion by region (1970–2000). *Box* plots show the median, first and third quartiles, minimum and maximum values of bootstrapped average annual rates of urban expansion by region (Modified from Seto et al. 2011, p. 4. Published with kind permission of © PLoS ONE 2011. All rights reserved.); (**b**) Comparison of two different urban growth measures by region and by decade. (*Top*) annual rates of urban population change. (*Bottom*) annual rates of urban land expansion. Population data are aggregated from individual countries to the geographic regions in the meta-analysis. Average annual rate of urban land change is based on the case studies in the meta-analysis. *Box* plots in (**b**) show the median, first and third quartiles, minimum and maximum values of bootstrapped average annual rates of urban expansion by region (Modified from Seto et al. 2011, p. 5. Published with kind permission of © PLoS ONE 2011. All rights reserved)

Table 21.1 Forecasts of additional urban land area by 2030 Using SRES Scenarios[a]

Baseline data set	Baseline urban extent (km^2)	Additional urban land area by 2030 (km^2)			
		A1	A2	B1	B2
MODIS 2001[b]	726,943	2,255,576	1,165,785	1,913,273	1,526,805
GRUMP 2000	3,524,108	12,568,323	5,734,517	9,818,872	7,619,054
GLC00 2000	307,575	857,528	429,865	719,188	586,177

Reproduced from Seto et al. (2011, p. 7). Published with kind permission of

[a]SRES Scenarios derived from http://sres.ciesin.columbia.edu/final_data.hml

[b]Based on MOD12Q1 V004 Land Cover Map (http://duckwater.bu.edu/lc/mod12q1.html). doi:10.1371/journal.pone.0023777.t003

low population growth; the B2 storyline has lower economic development than B1 and stabilizing population growth projections. For each of the four scenarios, the study created a new dataset to forecast urban land expansion. All variables other than those related to population and GDP remained constant in all four scenarios. The authors used the coefficients derived in a calibrated benchmark regression model and each of the four population/GDP scenario datasets to predict four sets of Annual Rate of Change (ARC) of urban expansion for each UN region for successive 5-year intervals up to the year 2030. This produced a range of estimates for the global urban land cover in 2030 based on the three different assumptions about the initial urban land cover in 2000/2001 (Table 21.1).[1]

Focusing on the MODIS data, which is shown to be the most accurate of all global urban extent maps (Potere et al. 2009), we find that the variation of projected urban expansion by 2050 is quite high across the four scenarios, ranging from 3,760,165 to 7,135,037 km^2 (Fig. 21.4).

21.4.2 Regression Analysis Projections

In another study on global urban expansion projections (Angel et al. 2011), the authors derived projections of urban land cover globally (across all countries and regions) up to the year 2050. The projections were made on the basis of a dataset of the population of 3,646 indexed urban agglomerations with populations over 100,000 people in the year 2000. The authors utilized information on the urban

[1]The authors' forecasts of global urban land cover for 2030 shows an increase of between 430,000 and 12,568,000 km^2 depending on assumptions about population and economic growth and on estimates of contemporary urban land cover. The primary reason for the large variance in the forecasts is the more than tenfold difference in areal estimates of contemporary urban land cover. The areal extent of urban land cover generated by GLC00, MODIS, and GRUMP are 308,007, 726,943, and 3,524,109 km^2, respectively. Using SRES scenario B2, our forecasts show additional urban land area between 587,000 and 7,619,000 km^2 by 2030. The highest estimates were generated using the GRUMP data set as the baseline for contemporary urban land extent. This data set has been shown to generate considerably higher global estimates of urban land cover than other data sets, by nearly five times the MODIS estimates, and ten times greater than the GLC00 estimates.

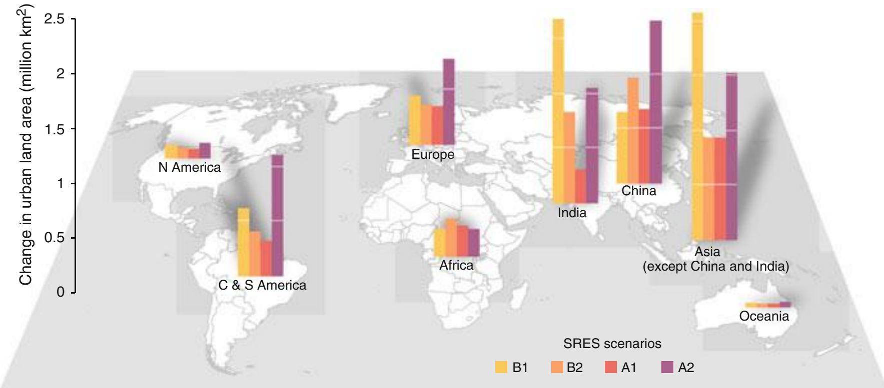

Fig. 21.4 Projected change in urban land area (2000–2050) by region and SRES scenario using MODIS 1 km data (Data from Seto et al. 2011; authors' calculations; data from Fragkias and Seto 2012. Modified from Fragkias and Seto 2012, p. 19. Published with kind permission of © The International Geosphere-Biosphere Programme/Hilarie Cutler. All rights reserved)

population sizes for these urban areas but also the (perceived) highest quality satellite-derived information existing today identifying the built-up area for each urban agglomeration for the year 2000. Assisted by the urban population projections of the United Nations and devising three scenarios of possible changes in density patterns (based on previous global and historical studies of densities), the authors used regression techniques to project land cover to the year 2050.

Based on historical observations from cities belonging in different nations and across different world regions, their high, medium and low projection scenarios assumed a 2, 1, and 0 % annual rate of density decline, respectively (Fig. 21.5). The medium projection scenario reveals urban land cover in developing countries will increase from 300,000 km^2 in 2000 to 770,000 km^2 in 2030 and to 1,200,000 km^2 in 2050. The medium projection scenario shows that globally, urban land will increase from 602,864 km^2 in 2000 to 1,267,200 km^2 in 2030 and to 1,888,936 km^2 in 2050. That is, 52.43 % of the total amount of urban land projected on the planet in 2030 was still undeveloped in 2000 (or 68.08 % considering projections for 2050).

21.4.3 *Spatially Explicit Simulation Projections*

According to one of the earliest studies on urban expansion trends into the future (Nelson et al. 2010), urban area will increase by 0.76 million km^2 (approximately the size of Turkey) from 2000 to 2015, while the cropland area – the main focus of the study–is expected to expand by 1.48–1.88 million km^2 (approximately the size of Iran and Libya, respectively). Nelson et al. (2010) do not detail spatial patterns and configurations of these changes. The two studies described in the preceding two

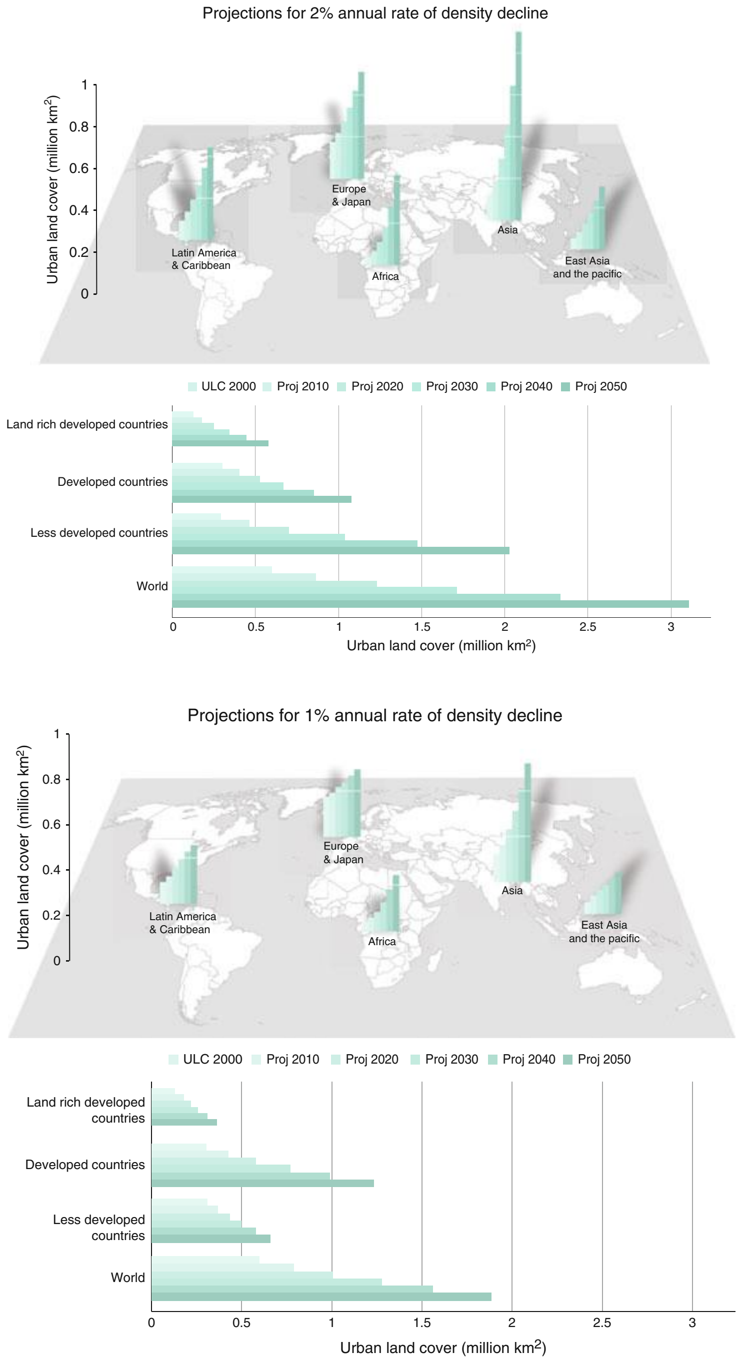

Fig. 21.5 Projections of urban land cover for world regions, 2000–2050, across three rates of density decline scenarios (Data from Angel et al. 2011, Table 6.2 and authors' plotting)

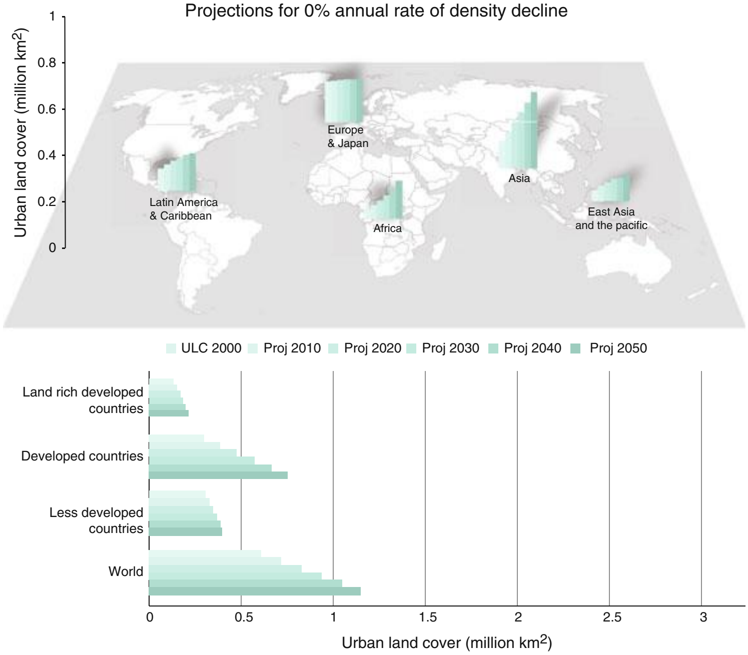

Fig. 21.5 (continued)

subsections forecasted urban growth aggregated by region. These two studies were followed by the first global, spatially explicit, probabilistic urban growth forecasts (Seto et al. 2012a). Seto et al. (2012a) used global land cover circa 2000 (Schneider et al. 2009) and projections of urban population (UN 2012b; NRC 2000) and gross domestic product (GDP) growth (Nakicenovic et al. 2000) in a probabilistic model of urban land change to develop 1,000 projections of urban expansion through to 2030. They generated the probabilistic, spatially explicit forecasts for 16 geographic regions, broadly based on the United Nations-defined world regions (see Appendix, Table A1). The forecasts show that the bulk of urban land-cover change will be concentrated in a few regions (Fig. 21.6a). Furthermore, the study suggests that there might be more urban land expansion during the first 30 years of the twenty-first century than it has been in all of history; more than 60 % of the urban land cover in 2030 is forecasted to be built in the first three decades of the twenty-first century.

Seto et al. (2012a) find that the areas with high-probability of urban expansion amount to a total area of 1.2 million km^2; half of this expansion would occur in China and India. In China, urban expansion is forecasted to create a 1,800 km coastal urban corridor from Hangzhou to Shenyang (Fig. 21.6b). In India, urban expansion is forecasted to be clustered around seven state capital cities, with large areas of low-probability growth forecasted in the Himalayan region, where many small villages and towns currently exist. In Africa, future urban expansion will be concentrated in five regions: the Nile River in Egypt, the coast of West Africa on

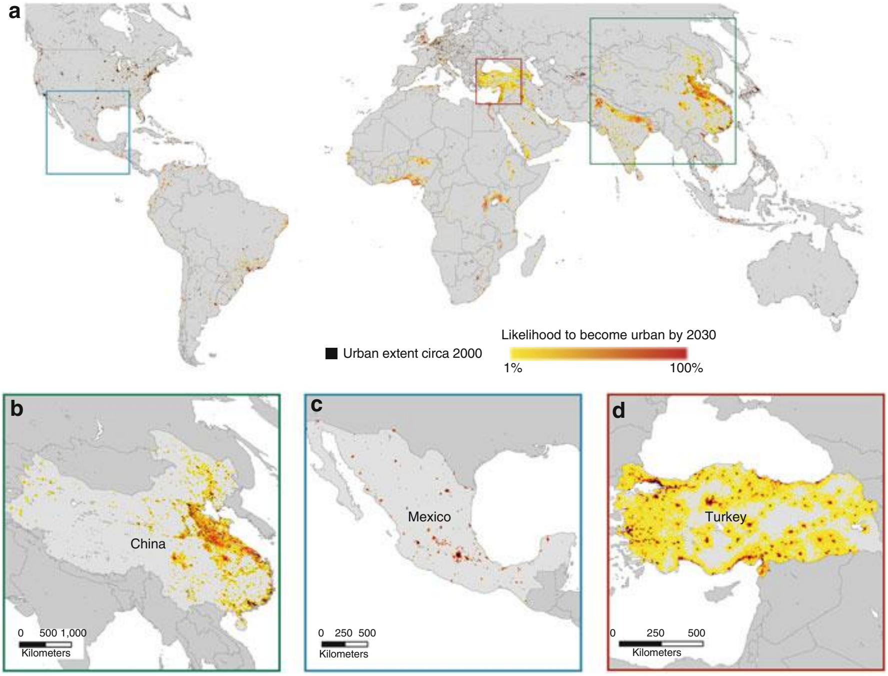

Fig. 21.6 Global forecasts of probabilities of urban expansion from 2000 to 2030 (**a**). Forecasted urban expansion in China is likely to occur along the coasts of the country (**b**). Some regions have high probability of urban expansion in a few locations (**c**) whereas others have large areas with low probability of urban expansion (**d**) (Reproduced from Seto et al. 2012a, p. 2. Published with kind permission of © PNAS 2012. All rights reserved)

the Gulf of Guinea, the northern shores of Lake Victoria in Kenya and Uganda and extending into Rwanda and Burundi, the Kano region in northern Nigeria, and greater Addis Ababa, Ethiopia. In North America, where the percentage of total population living in urban areas is already high (78 %), the forecasts show a near doubling of urban land cover by 2030. On the other hand, 48 of the 221 countries in the study are forecasted to experience little or no urban expansion. The probabilistic analysis reveals that, in many countries, there is an inverse relationship between the probability that specific geographic locations will experience urban expansion and the magnitude of predicted urban expansion. For example, total forecasted area of urban expansion in Mexico is concentrated in a few locations, whereas in Turkey, large areas have low probabilities of urban expansion (Fig. 21.6c, d).

Although the assumptions and nature of the analysis in Seto et al. (2012a) is different than either Seto et al. (2011) or Angel et al. (2011), the former reaches similar conclusions in terms of the rates and magnitudes of urban expansion. According to Seto et al. (2012a), if current trends persist, rates of urban land expansion will continue to exceed those of urban population growth everywhere around the world

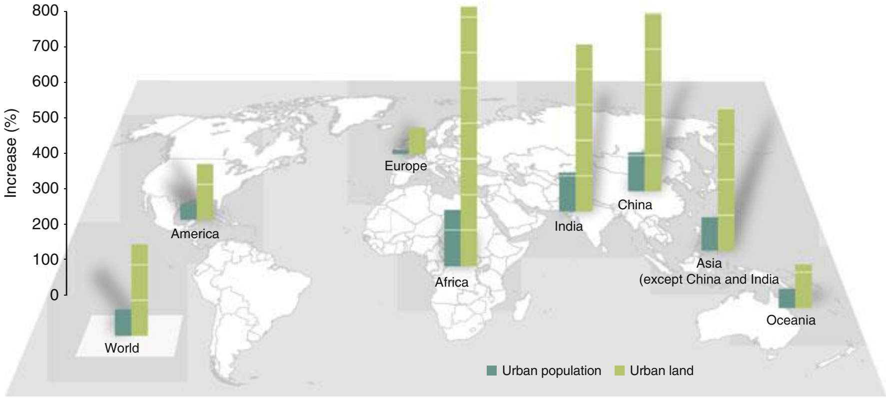

Fig. 21.7 Percent increase in urban population and urban land cover from 2000 to 2030 according to UN (2011b) and Seto et al. (2012a), respectively

(Fig. 21.7). From 2000 to 2030, the percent increase in global urban land cover will be over 200 %, whereas the global urban population will only grow by a little over 70 %. The most dramatic is Africa where urban land cover is forecasted to increase by 700 %, compared to a 160 % increase in the continent's urban population in the same period.

Box 21.2 A Comparison of Urban Physical Extent Projections Across Methodologies and Scenarios

Differences in the projections of Seto et al. (2011, 2012a), and Angel et al. (2011) reviewed in this chapter (see Table 21.2 for a summary of the estimates) arise primarily from the baseline urban extent layers used for year 2000, the choice of the methodological approach on scenario building and data analysis as well as the data use in each study. The A2 scenario of the Seto et al. (2011) study predictions fall relatively close to the 2 % annual rate of density decline scenario of the Angel et al. (2011) study, which constitutes the most "pessimistic" vision over the evolution of urban densities. Other than this convergence, the predictions of the two studies deviate significantly. Scenario-building is treated differently across the studies; the Seto et al. (2011) study employs four scenarios of possible economic and demographic futures while the Angel et al. (2011) study employs three scenarios of possible evolutions of global densities. The narratives and quantification of the former could substantially drive the difference in projections. Note that a direct and complete regional contrast is also impossible given the regional classification schemes chosen by the authors of the two studies. The Seto et al. (2012a) projections are generated utilizing a different methodological approach (a spatially explicit

(continued)

Box 21.2 (continued)

land cover change simulation that accounts for uncertainty) and still reaches similar conclusions in terms of the rates and magnitudes of urban expansion when compared to the middle of the road projections in the Seto et al. (2011) and Angel et al. (2011) studies.

Attempting a comparison of the projections, one needs to eliminate two of the baseline urban extents used in the Seto et al. (2011) study (namely, GRUMP and GLC) utilizing information only from MODIS. An important differentiation between the two studies driving projection differences is the baseline urban extent employed in each study. The Seto et al. (2011) study defines the urban land cover extent in year 2000 as 726,943 km^2 using the MODIS Urban Land Cover map at a 1 km resolution while the Angel et al. (2011) study employs an extent of 602,862 km^2 using the MODIS Urban Land Cover map at a 500 m resolution. This difference of 124,081 km^2 (about one fifth of the land cover extent used in the Angel et al. (2011) study) could explain partially the difference in the projections of the two studies. Future research is expected to explore the sensitivity of the results to the size of benchmark urban extent in 2000.

(Source: Authors' calculations)

Table 21.2 presents the collection of recent projections on urban physical expansion across all studies and scenarios utilized. It is worthwhile to note that, on average, across all studies and scenarios, 65.35 % of the total projected urban land in 2030 will be built in the first three decades of the twenty-first century. This fact constitutes an amazing challenge and opportunity for urban sustainability.

21.5 Discussion and Conclusions

Global urban projections on population, wealth, and physical expansion are increasingly becoming useful tools assisting in the governance of coupled human-natural systems. Global urban projections can help us envision if and how city systems will affect ecologically fragile areas, contribute to the loss of agricultural land, dominate coastal zones, encroach on arid ecosystems, or generally develop in areas sensitive to the effects of climate change. Global urban models can also help us identify areas where particular types of urban development may be problematic or beneficial.

Pushing the agenda of global urban projections forward, researchers will be able to explore how future urbanization hotspots can reliably incorporate functions or features such as durable housing, and access to improved water, key resources, and sanitation; they can also examine how to avoid overcrowding, high levels of unemployment and social exclusion. New scenarios and projections can also help in

Table 21.2 Urban physical expansion projections by study and scenario employed

	2000	2030	2040/2050[a]	Proportion of urban land in 2030 Not built in 2000 (%)	Proportion of urban land in 2050 Not built in 2000 (%)	Change (%) 2000–2030	Change (%) 2000–2050
Seto et al. (2011) (A1)	726,943	2,982,519	7,135,037	75.63	89.81	310.28	881.51
Seto et al. (2011) (A2)	726,943	1,892,728	3,760,165	61.59	80.67	160.37	417.26
Seto et al. (2011) (B1)	726,943	2,640,216	6,649,673	72.47	89.07	263.19	814.74
Seto et al. (2011) (B2)	726,943	2,253,748	4,736,975	67.75	84.65	210.03	551.63
Angel et al. (2011) (0 %)	602,864	938,765	1,145,698	35.78	47.38	55.72	90.04
Angel et al. (2011) (1 %)	602,864	1,267,200	1,888,936	52.43	68.08	110.20	213.33
Angel et al. (2011) (2 %)	602,864	1,710,542	3,114,330	64.76	80.64	183.74	416.59
Seto et al. (2012a) (>25 % threshold)	652,825	1,669,600	No data	60.90	No data	155.75	No data
Seto et al. (2012a) (>50 % threshold)	652,825	1,419,300	No data	54.00	No data	117.41	No data
Seto et al. (2012a) (>75 % threshold)	652,825	1,210,475	No data	46.07	No data	85.42	No data

[a]Seto et al. (2011) endpoint is 2050; Angel et al. (2011) endpoint is 2040; Seto et al. (2012a) does not produce projections for 2050

identifying institutional settings appropriate for increased prosperity through the rule of law, accountability structures and action against corruption. They can also elucidate what conditions will be critical in avoiding “stress bundles” that increase the probability of societal challenges.

Our overview suggests that the scientists that are involved in the business of projections (demographers, geographers, economists, etc.) are typically unwilling to make projections farther than a few decades into the future; this is primarily due to the uncertainty that is introduced for longer time horizons, which increases substantially beyond 30–40 years. On the foundation of this unwillingness lies the idea of a non-ergodic world, that the future is not merely a statistical reflection of the past (Davidson 1994). All scientific disciplines face methodological challenges that relate to non-ergodicity. While some of these challenges will not be easily overcome, the general argument for synergies arising from interdisciplinary collaborations also holds in the field of global urbanization projections; joint approaches in demography and remote sensing have showcased the usefulness of interdisciplinary collaborations (Donnay et al. 2001).

Projections on urban population have been and will most probably continue to be a foundation for all other global urban projections. Several dimensions of these projection activities need to be taken into serious consideration:

1. The lack of a consistent single definition of “urban” population across countries and a heavy reliance on individual country definitions of “urban” is problematic (Box 21.1 and Chap. 1). Admittedly, the task of integrating country-specific urban population is daunting and potentially forbiddingly costly, and the resulting data mosaic may be too vague and inconsistent. The “urban definition problem” is expected to continue plaguing the interpretation of global projections.
2. The urban-rural duality is an overly simplistic concept, and although it has served the research and practitioner communities well, it needs to be significantly augmented. Future research on global urban projections needs to assist in a transition towards an alternative and more context-rich approach through the flourishing of geospatial technologies in combination with more spatially sensitive governmental census and survey efforts.
3. Since urban projections are in many cases primarily dependent on total population projections and their assumptions, urban researchers need to be aware of advancements in modeling of demographic processes – including the treatment of uncertainties in the factors that lead to drops in fertility.
4. Urban and world population projections may mean little if they are disconnected with the overall regional and national economic growth patterns.

The literature on projections of urban wealth clearly shows that this is an underexplored area of research. As the world economic center of gravity moves to the South and to the East – and at progressively faster rates – businesses and governments have begun adjusting decisions and policies; this is an area that is underserved by research, and quite possibly constitutes a gap that will be filled by the sector of business intelligence. Global economic projections will be particularly important in understanding demands for particular products and services that are expressed at particular levels of income. Understandably, this is a very important consideration for sustainability trajectories and sustainable economic performance of nations and the globe. Furthermore, it is still unclear how the informal sector of economies and the interconnectivity for formal and informal economies will evolve in the future – the trajectory of the disparities introduced by the existence of the informal sector is not well understood.

Recent projections on the physical expansion of urban areas show four trends that have significant implications for climate change adaptation, biodiversity, and human well-being, among other things:

1. Physical expansion of urban areas occurs at much higher rates than population change; for example, the total urban area as reported by the meta-analysis case studies quadrupled over the 30 years while urban population at national levels doubled.
2. Urban land expansion is growing faster in low elevation coastal zones than in other areas. This is likely to put millions of people at risk to climate change impacts such as storm surges and sea level rise.
3. Rates of urban land expansion near protected areas are as high as in other regions. This will challenge conservation strategies because future urban expansion is expected to be both significant in total area extent and also as likely to occur near protected areas as other regions (Chap. 22).
4. Urban population growth and GDP explain only a percentage of urban land expansion; non-demographic factors and economic dynamics not captured by GDP also play a large role. Although global urban population is expected to increase to 5 billion by 2030 from 3.1 billion in 2010, the results indicate that many non-demographic factors, including land-use policies, transportation costs, and income will shape the size of global urban extent in the coming decades.

The physical expansion studies are also pointing out significant challenges for planning and governance in the years ahead. Excluding the case of a significant exogenous shock, the projected expansion of urban land cover is not likely to be contained and will be difficult to manage. Angel et al. (2011, p. 53) suggest that "[m]inimal preparations for accommodating it – realistic projection of urban land needs, the extension of metropolitan boundaries, acquiring the rights-of-way for an arterial road grid that can carry infrastructure and public transport, and the selective protection of open space from incursion by formal and informal land development – are now in order" (see further Chap. 27). Addressing non-myopically the massive amount of urban land increase projected globally – and especially for developing world nations – means the beginning of an era of significant investments in infrastructure development and the creation of new formal institutions as a foundation of this growth. What remains unclear is the full set of consequences for global environmental change and the wider implications of the urban responses to this change. Additional efforts from researcher and practitioner communities can offer potential pathways towards sustainable urban futures.

Although researchers have advanced the field of urban dynamic projections considerably, traditional modeling of urban dynamics is faced with challenges in its capacity to fully grasp the trajectories of urban systems globally, the planetary scale of their impacts, and the ways through which changes in global environment will affect the further development of urban systems. There are very few models that capture adequately the coupled dynamics of human-social-ecological systems and this occurs due to conceptual and methodological challenges involved in integrating human and natural systems. While urban population, wealth, and land-use dynamics are today clearly considered as major *drivers* of global environmental change, only in a few instances are they considered as direct or indirect *outcomes* of global environmental change. The final chapter of this volume (Chap. 33) discusses in more detail the importance of a new generation of integrated models that will allow researchers to better analyze the dynamic behavior of complex systems and to show the full extent of interrelations and feedbacks between human and natural systems.

Several frontiers remain open in the attempt for more accurate and precise forecasts or insightful projections for sustainability. Global urbanization projections are critical for the operationalization of a new science of urbanization (Seitzinger et al. 2012; Solecki et al. 2013). Perhaps the most important of those is the conceptualization of urban environments as closely linked to their hinterlands but also the hinterlands of urban environments far away – a concept that has been discussed as the urban land teleconnections (Seto et al. 2012b). Although cities can optimize their resource use, increase their efficiency, and minimize waste, they can never be fully self-sufficient. Therefore, individual cities cannot be “sustainable” without acknowledging and accounting for their dependence on resources and populations in other regions around the world. Cities in and of themselves cannot be sustainable. A more accurate conceptualization of sustainable cities is one that incorporates a systems perspective of urban areas and their global hinterlands, and one that considers the urbanization process and the disproportionate contribution of urbanization to the global cultural, social, and economic capital and human well-being. Global urbanization projections need to establish these links considering that sustainable urbanization is a necessary, but not sufficient, condition for a sustainable planet.

Appendix

Table A1 Composition of regions used in Seto et al. (2012a)

Regions defined in model	Abbr.	Included UN regions	Plus	Minus
Central America	CAM	Central America, Caribbean	–	–
China	CHN	–	China, Hong Kong, Macao	–
Eastern Asia	EAS	Eastern Asia	Taiwan	China, Hong Kong, Macao, Mongolia
Eastern Europe	EEU	Eastern Europe	Kazakhstan, Estonia, Lithuania, Latvia, Albania, Bosnia-Herzegovina, Croatia, Macedonia, Montenegro, Serbia	–
India	IND	–	India	–
Mid-Asia	MAS	Central Asia	Mongolia	Kazakhstan
Mid-Latitudinal Africa	MLA	Western, Middle, Eastern Africa	–	–
Northern Africa	NAF	Northern Africa	–	–
Northern America	NAM	Northern America	–	–
Oceania	OCE	Oceania	–	–
Southern Africa	SAF	Southern Africa	–	–
South America	SAM	South America	–	–
Southern Asia	SAS	Southern Asia	–	India
Southeastern Asia	SEA	Southeastern Asia	–	–
Western Asia	WAS	Western Asia	–	–
Western Europe	WSE	Western Europe, Southern Europe, Northern Europe	–	Estonia, Lithuania, Latvia, Albania, Bosnia-Herzegovina, Croatia, Macedonia, Montenegro, Serbia

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Chapter 22
Urbanization Forecasts, Effects on Land Use, Biodiversity, and Ecosystem Services

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Abstract Several studies in recent years have forecasted global urban expansion and examined its potential impacts on biodiversity and ecosystem services. The amount of urban land near protected areas (PAs) is expected to increase, on average, by more than three times between 2000 and 2030 (from 450,000 km^2 *circa* 2000) around the world. During the same time period, the urban land in biodiversity hotspots, areas with high concentrations of endemic species, will increase by about four times on average. China will likely become the nation with the most urban land within 50 km of its PAs by 2030. The largest proportional change, however, will likely be in Mid-Latitudinal Africa; its urban land near PAs will increase 20±5 times by 2030. The largest urban expansion in biodiversity hotspots, an increase of

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over 100,000 km^2, is forecasted to occur in South America. The forecasts of the amount and location of urban land expansion are subject to many uncertainties in their underlying drivers including urban population and economic growth. Nevertheless, the direct impacts of urban expansion on biodiversity and ecosystem services will likely be significant. The forecasts point to the need to reconcile urban development and biodiversity conservation strategies. Urbanization will also have impacts on food and food security. While the direct loss of cropland to urban expansion is of concern to the extent that high-yielding croplands are lost, the indirect impacts of urbanization due to dietary changes to more meat-based food products can also be substantial. Presently, regional and global studies that forecast impacts of future urban expansion on biodiversity and ecosystem services are in their infancy and more analyses are needed especially focusing on interactive effects of factors that drive urbanization. We conclude by highlighting the knowledge gaps on implications of future urbanization and suggest research directions that would help fill these gaps.

22.1 Impacts of Urbanization on Biodiversity

Urbanization impacts biodiversity both directly through physical expansion over land, and indirectly due to land use and human behaviors within urban areas. Physical expansion changes the composition of the landscape, and can eliminate organisms outright, or may alter or eliminate the conditions within a habitat that a species requires to survive. Urban expansion has the effect of decreasing, fragmenting, and isolating natural patches by altering the size, shape, and interconnectivity of the natural landscape (Ricketts 2001; Alberti 2005). In addition to physical expansion, human activity within cities can have a myriad of cascading effects that have impacts on biodiversity, including changes in biogeochemistry (Vitousek et al. 1997; Grimm et al. 2008), local temperature (Arnfield 2003; Voogt and Oke 2003), climate change (Kalnay and Cai 2003; Sanchez-Rodriguez et al. 2005; Wilby and Perry 2006) (Chap. 25), and hydrologic systems (Walsh 2000; Booth et al. 2004). Consequences for biodiversity and ecosystem services are difficult to generalize and depend on the taxonomic groups in question, spatial scale of analysis, and intensity of urbanization,

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among other factors (McKinney 2008); for example, in some urbanizing areas, local species richness may increase (albeit usually at the cost of native species) while in others it may decrease (McKinney 2002, 2006; Grimm et al. 2008) (see also Chap. 10).

Ultimately, studies attempting a detailed categorization of the impacts of current and projected urbanization on biodiversity and ecosystem services are important in further exploring, investigating, and testing the trends. A large body of work has been amassed on trends and projections of the impacts of urbanization on biodiversity and ecosystem services over time at the local scale. The local assessments included in this volume are a collection of several local studies from around the world. Local scale studies can provide useful insights; due to their limited geographical scope, they are often able to draw on rich databases concerning biodiversity (such as a detailed species records) and high-resolution data on land-use change over time. However, because of disparate approaches in methodologies and indicators, it is often difficult to merge data or results to draw aggregated conclusions. For the purpose of this chapter, we focus on scales of regional and global studies. Chapter 10 provides a deeper examination of urban impacts on biodiversity and ecosystems at the city scale. Chapter 3 provides an outlook of current global conditions in regards to how urbanization affects biodiversity conservation through impacts on global ecoregions, rare species and protected areas. In addition, Chap. 12 discusses the phenomenon of shrinking cities and its implications for biodiversity and ecosystem services. While Chap. 21 examines global projections of future urbanization, covering the population, economic and physical extent perspectives, this chapter examines research that specifically addresses impacts of forecasted urbanization on biodiversity and ecosystem services.

22.2 Impacts of Forecasted Urbanization on Biodiversity

22.2.1 Global Trends

Future urban population will increasingly reside in tropical areas (Fig. 3.6). According to the UN predictions, by 2050 there will be particularly noticeable increases in urban population in tropical moist forests, deserts and tropical grasslands. In addition, in terms of urban population per habitat area, there will be significant increases in impact in mangroves, flooded grasslands, and temperate broadleaf forests. Also worth noting are impacts to tropical conifer forests, a unique habitat type found only in a relatively small area globally. In contrast to the population dimension of global urbanization, until recently, there was little or no understanding of how urban areas grew in the past and how they will continue to grow into the future (Chap. 21). Addressing this gap in knowledge, a number of studies were recently published on global urbanization trends and their impacts on biodiversity and ecosystems services (Table 22.1).

Table 22.1 Comparison of global urban expansion studies based on data sources, methodology, impacts on biodiversity, and treatment of uncertainty

Study	Initial urban extent	Urban expansion: Drivers (Horizon)	Spatially-explicit?	Biodiversity hotspots	Protected areas	Biomes/ ecoregions	AZE sites	Uncertainty
McDonald et al. (2008)	GRUMP (urban extent and settlements data)	UN urban population forecasts (2030)	Crude	No	Yes	Yes	Yes	Scenario-based (baseline, compact, dispersed urban growth)
McDonald et al. (2009)	As in McDonald et al. (2008)	As in McDonald et al. (2008)	As in McDonald et al. (2008)	No	Yes	No	No	As in McDonald et al. (2008)
Nelson et al. (2010)	GLC2000 and GRUMP urban extent	UN urban population forecasts and slope (2015)	Yes (GEOMOD; 5 km at equator)	No	No	Yes	No	Scenario-based (country vs regional)
Seto et al. (2011)	MODIS v4, GRUMP urban extent, GLC2000[a]	Pop, economy, institutional and physical factors (2030)	Crude – regional	No	Yes[b]	No	No	Scenario-based (IPCC AR4 scenarios)
Seto et al. (2012a)	MODIS v5	Pop, economy, and physical factors (2030)	Yes (URBANMOD – descendant of GEOMOD; 5 km equal area)	Yes	No	No	Yes	Locational probability to become urban (Monte Carlo simulations) based on various sources
Güneralp and Seto (2013)	MODIS v5	As in Seto et al. (2012a)	As in Seto et al. (2012a)	Yes	Yes	No	No	Probability of urban expansion (Monte Carlo simulations) based on various sources

[a]One forecast with each
[b]Only for the past trends, not forecasted

Since urban and cropland effects are aggregated, Nelson et al. (2010) do not explore the impacts on biodiversity and ecosystem services due solely to urban expansion. However, they do specify different projections across scenarios for the following ecosystem services: provision of crops (in mass and caloric content), water availability, and carbon storage in biomass. These are fed in to the Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST) models (Tallis et al. 2010) to calculate how changes in land cover and land use will affect the global provision of crops, water availability, carbon storage in biomass (a climate regulation service), and habitat for species. Changes in undeveloped land extent (non-urban and non-cropland cover) serve as a proxy for species habitat and impacts on biodiversity, as it is characterized in the study that undeveloped land is more likely to provide species habitat than other land uses. The study uses ecoregion status (Olson et al. 2001) and describes threats to these areas based on predicted conversions of undeveloped land area. Olson et al. (2001) define an ecoregion as a relatively large area "containing a distinct assemblage of natural communities and species, with boundaries that approximate the original extent of natural communities prior to major land-use change". Critical and endangered ecoregions are predicted to retain little natural habitat, the remainder of which is highly fragmented and has highly uncertain species persistence. Those ecoregions classified as vulnerable and relatively stable are forecasted to experience fewer disturbances. According to their scenario analysis, from 2000 to 2015, between 1.2 and 1.6 million km^2 of undeveloped land in critical/endangered ecoregions is forecasted to become urban or cropland.

In their probabilistic analysis, Seto et al. (2012a) used the Alliance for Zero Extinction (AZE) dataset (Ricketts et al. 2005) to analyze the direct impact of urban expansion on highly threatened species that are confined to small areas. More than a quarter of all species in the AZE dataset will be affected by urban expansion with some probability by 2030. Africa and Europe are expected to have the highest percentages of AZE species to be affected by urban expansion: 30 and 33 %, respectively. However, it is the Americas that will have the largest number of species affected by urban expansion: 134 species, representing one-quarter of all AZE species in the region. On the other hand, in their deterministic analysis, McDonald et al. (2008) estimated that about 3 % of species in the AZE dataset will be adversely affected by urban growth by 2030; these species are mostly located along coastal areas and islands where endemism tends to be particularly high (Ricketts et al. 2005).

In the most recent publication on global forecasts of urban expansion and corresponding impacts on biodiversity, Güneralp and Seto (2013) quantify the urban extent in biodiversity hotspots and IUCN-designated protected areas (PAs) across the world by geographical region (Table A.1 in Chap. 21). The biodiversity hotspots, one of several conservation prioritization concepts (Brooks et al. 2006), are defined as regions with many endemic species facing exceptional habitat loss and degradation (Myers et al. 2000). Güneralp and Seto (2013) first quantify the amount of urban land in PAs and in three concentric buffer zones around PAs by region, around year 2000 and forecasted to year 2030. Similarly, they also quantify the

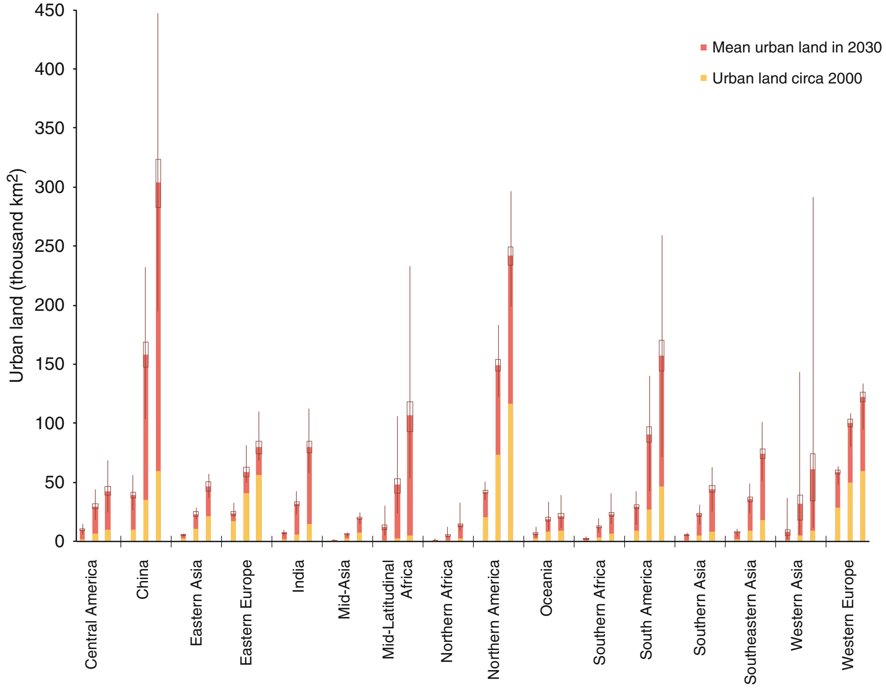

Fig. 22.1 Urban extent, within a distance of, from *left* to *right*, 10, 25, and 50 km of PAs by geographic region *circa* 2000 and as forecasted in 2030 (Modified from Güneralp and Seto 2013, p. 5. Published with kind permission of © Environmental Research Letters 2013. All rights reserved)

distribution of urban land across biodiversity hotspots by region as well as by biodiversity hotspot.

By 2030, the urban lands near PAs are predicted to increase substantially in almost all the regions (Figs. 22.1 and 22.2). Most notably, China will most likely surpass Northern America and Western Europe in urban land within 25 km and 50 km of their respective PAs. China's urban land within 25 km and 50 km distance of its PAs increase, respectively, to 160,000 ± 50,000 km^2 and 300,000 ± 93,000 km^2. These changes correspond to an increase of 4.5 ± 1.5 times in 30 years. The largest proportional change, however, will likely be in Mid-Latitudinal Africa; in that region, urban land near PAs increase 20 ± 5 times by 2030. In contrast, the rate of increase is relatively small in Northern America, South America, and Western Europe.

Across the world, between 2000 and 2030, total urban land in biodiversity hotspots is expected to increase 4 ± 0.8 times to 787,000 ± 160,000 km^2 –the average is about the same as the land area of Turkey (Güneralp and Seto 2013). Correspondingly, percentage of urban land located in biodiversity hotspots is expected to increase to 34 % (±2 %) in 2030 from 31 % *circa* 2000. By 2030, the largest increase in the amount of urban land in biodiversity hotspots is expected to be in

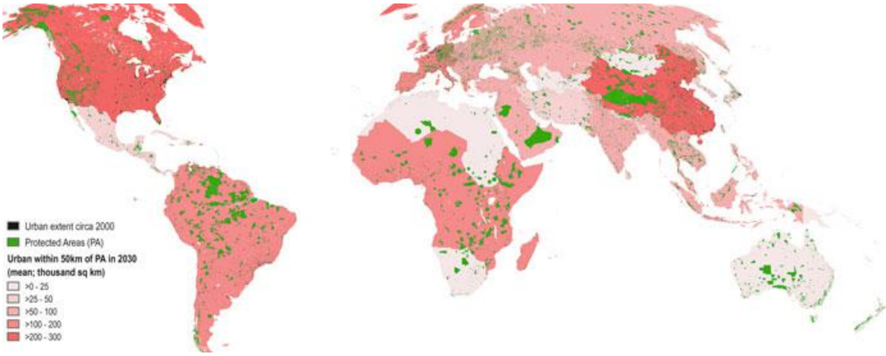

Fig. 22.2 Mean of forecasted urban extent within 50 km of PAs by geographic region in 2030. Urban extent *circa* 2000 and PAs are also shown (Modified from Güneralp and Seto 2013, p. 6. Published with kind permission of © Environmental Research Letters 2013. All rights reserved)

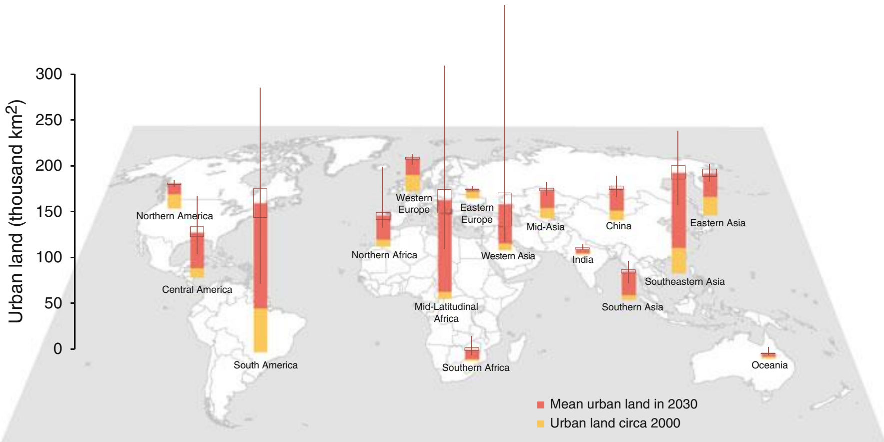

Fig. 22.3 Urban extent in biodiversity hotspots by geographic region *circa* 2000 and as forecasted in 2030 (Modified from Güneralp and Seto 2013, p. 6. Published with kind permission of © Environmental Research Letters 2013. All rights reserved)

South America (an increase by more than 100,000 ± 25,000 km^2) (Fig. 22.3). This corresponds to nearly a 3.5 ± 0.5 fold increase in urban land in the region's biodiversity hotspots. The largest proportional increase (about 14 ± 3 fold) is forecasted to be in Mid-Latitudinal Africa.

Of the 34 biodiversity hotspots (Mittermeier et al. 2004; Myers et al. 2000), seven contain more than 10,000 km^2 of urban land *circa* 2000 (Fig. 22.4). Of the seven, five are located in Asia (four wholly, one, the Mediterranean, in part);

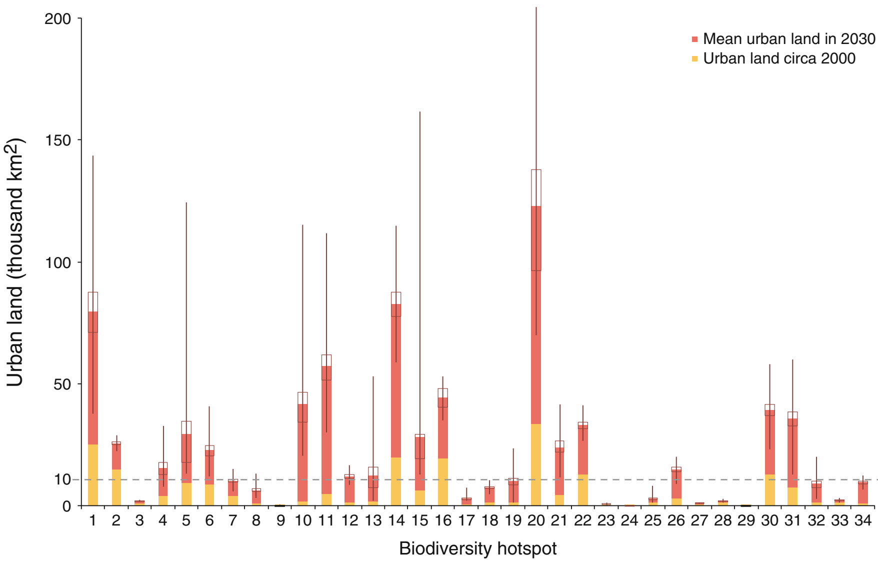

Fig. 22.4 Urban extent in biodiversity hotspots *circa* 2000 and as forecasted in 2030. *1* Atlantic Forest, *2* California Floristic Province, *3* Cape Floristic Region, *4* Caribbean Islands, *5* Caucasus, *6* Cerrado, *7* Chilean Winter Rainfall and Valdivian Forests, *8* Coastal Forests of Eastern Africa, *9* East Melanesian Islands, *10* Eastern Afromontane, *11* Guinean Forests of West Africa, *12* Himalaya, *13* Horn of Africa, *14* Indo-Burma, *15* Irano-Anatolian, *16* Japan, *17* Madagascar and the Indian Ocean Islands, *18* Madrean Pine-Oak Woodlands, *19* Maputaland-Pondoland-Albany, *20* Mediterranean Basin, *21* Mesoamerica, *22* Mountains of Central Asia, *23* Mountains of Southwest China, *24* New Caledonia, *25* New Zealand, *26* Philippines, *27* Polynesia-Micronesia, *28* Southwest Australia, *29* Succulent Karoo, *30* Sundaland, *31* Tropical Andes, *32* Tumbes-Choco-Magdalena, *33* Wallacea, *34* Western Ghats and Sri Lanka (Modified from Güneralp and Seto 2013, Figure S4, p. 6 of supplementary data. Published with kind permission of © Environmental Research Letters 2013. All rights reserved)

the remaining two are located in America and the Mediterranean along the coasts of Southern Europe and Northern Africa. The Mediterranean hotspot contains the most urban land, spread across three continents with different geographic, cultural, social, and economic characteristics. In a hotspot such as the Mediterranean that is already diminished and severely fragmented, even relatively modest decreases in habitat can cause the pressure on rare species to rise disproportionately (Tilman et al. 1994). The Mediterranean Basin may become the only hotspot containing more than 100,000 km^2 (123,000 ± 37,000 km^2) of urban land in 2030 (Fig. 22.4). Almost half of this expansion is predicted to occur in Western Asia and about a third in North Africa.

The highest rates of increase – over ten times – in urban land cover are forecasted to take place in four biodiversity hotpots that were relatively undisturbed by urban land change at the turn of this century: Eastern Afromontane, Guinean Forests of West Africa, Western Ghats and Sri Lanka, and Madagascar and the Indian Ocean Islands.

Nevertheless, these high rates imply that some of those few hotspots that remained relatively undisturbed by the turn of this century will be increasingly encroached upon by urban expansion during its first three decades.

The analysis in Güneralp and Seto (2013) complements the account of locational probability of urban expansion forecasts in biodiversity hotspots of Seto et al. (2012a). These two studies quantify the forecasted urban land expansion using a land change model; hence they extend and complement two previous studies on urbanization and biodiversity conservation (McDonald et al. 2008, 2009). These two studies report rough projections of aggregate urban land expansion based solely on forecasted urban population growth and focus on different aspects of the proximity between urban land and PAs. In particular, McDonald et al. (2008) estimate that 25 % of the world's PAs will be within 15 km of a city of at least 50,000 people by 2030. As a whole, these studies suggest that we need to find ways of coexistence between urban areas and PAs at such close proximities. The findings from Güneralp and Seto (2013) are conservative because some PAs are below the spatial resolution of their analysis (5 km). This leads to some underestimation of urban expansion in and around these areas. This is most problematic for regions in North America, Europe, and China where there are extensive networks of PAs. Most of those PAs that are below the spatial resolution of their analysis are in IUCN categories V and VI, some of which are small parks closer to cities. In addition, contrary to the conservative assumption in Güneralp and Seto (2013) of perfect enforcement of the formal regulations that do not permit urban expansion *within* PAs, the urban areas *within* PAs may very well expand at least in some parts of the world.

How urbanization will affect PAs will largely depend on the effectiveness of land use, conservation, and urbanization policies. Effective governance of land near PAs for preservation of ecosystem functioning and conservation of biodiversity can be challenging even for developed countries (Wade and Theobald 2010) (Chap. 27). This may be due to various political and cultural reasons, including fragmented jurisdictions of several bodies (Shafer 1999) and the lack of coordination between agencies responsible for governing PAs and the actors who govern the lands around PAs (Davis and Hansen 2011) (Chap. 27).

The hotspots in South and Central America as well as in Southeast Asia will experience both high rates and high amounts of urban expansion by 2030. The amount of urban land within hotspots will also increase in China, but will be relatively less than urban expansion elsewhere in the country. Some of the few hotspots that remained relatively undisturbed by the turn of this century will also be increasingly encroached upon by urban expansion –especially in the islands of Oceania and the Indian Ocean– during the first three decades of this century.

Urban expansion will also impact freshwater availability and, consequently, biodiversity (see Chap. 3 for current trends). A detailed paper modeled how population growth and climate change might affect water availability for all cities in developing countries with greater than 100,000 people (McDonald et al. 2011). These cities had 1.2 billion residents in 2000 (60 % of the urban population of developing countries). Modeled output suggests that currently 150 million people live in cities with perennial water shortage, defined as having less than 100 l/person/day of sustainable surface and groundwater flow within their urban extent. By 2050, this number is forecasted to increase to almost a billion people due to demographic growth. Climate change will cause water shortage for an additional 100 million urbanites. Cities in certain regions will struggle to find enough water for the needs of their residents, and will need significant investment if they are to secure adequate water supplies and safeguard functioning freshwater ecosystems for future generations. Of particular conservation concern is the Western Ghats of India, which will have 81 million people with insufficient water by 2050, but also houses 293 fish species, 29 % of which are endemic to this ecoregion and found nowhere else in the world.

Regardless of whether cities are investing in infrastructure to increase water supply or trying to use existing supplies more wisely, it is clear that substantial financial resources will be required to address these management challenges in the future. One study estimated that from 2003 to 2025 necessary annual investments would exceed $180 billion per year (World Panel on Financing Water Infrastructure 2003). While plenty of possible solutions to water quantity and quality problems exist, including some that are relatively less harmful to the environment, they all take money and time to implement. For the more than a billion people in cities facing water delivery challenges, both are in short supply.

Collectively, the findings of these studies suggest the need for conservation policies that consider urban growth at both regional and global scales. The threat to biodiversity comes from direct land cover change and subsequent loss of habitat, but also from indirect factors such as increased colonization by introduced species as urban areas expand. In regions with high likelihood of becoming urban, certain management practices such as establishing biodiversity corridors will require coordinated efforts among administrative bodies within and among nations. Such corridors may take on additional significance considering the migration of species in response to shifts in their ranges with climate change (Loarie et al. 2009).

Notwithstanding the differences in terms of data and methods used across these global-level studies, there are some broad agreements on the rates and magnitudes of future urban expansion and where its direct impacts are likely to be the most prominent. Urban expansion will continue near PAs at least at the same pace as elsewhere—if not faster—across most of the world. This increases the need to generate conservation and regional planning solutions to safeguard the integrity of the ecosystem processes that more often than not extend beyond PA boundaries (Hansen and DeFries 2007; McDonald et al. 2009; Güneralp and Seto 2013).

22.2.2 Regional Perspectives

There is a significant body of knowledge on urban impacts on biodiversity and ecosystem services from around the world (see Chap. 3 for current trends); however, there is yet no well-developed understanding of how these impacts will evolve into the future except those that come from the regional breakdowns in some of the global studies (Sect. 22.2.1). While global-scale analyses and projections of the effects of urbanization on biodiversity and ecosystem services are valuable for giving breadth of perspective and thus inform on broad trends, studies that focus on particular regions may allow for additional depth and insight on those regions. However, such large regional and country-level studies are also sparse.

There have been several studies forecasting the impacts of urban and ex-urban expansion on wildlife and protected areas in the United States. The wildlife-urban interface (WUI) in the United States, estimated to be about 465,614 km^2 in 2000, is likely to expand to over 500,000 km^2, with the greatest expansion expected in the inter-mountain west states (Theobald and Romme 2007). Bierwagen et al. (2010) projected growth of housing and impervious surfaces in the U.S. out to 2100 according to the IPCC 4th Assessment scenarios. According to their scenario forecasts, housing development impacts nearly one-third of wetlands under all scenarios by 2050 and nearly half by 2100 for A2. They emphasize that unless appropriate land-use and conservation policies are put in place, the vulnerability of this ecosystem type to runoff, sedimentation, and habitat loss will be high. Finally, Hamilton et al. (2013) forecasted urban land use around the protected area network in the U.S. out to 2051. They too employed a scenario-based approach to capture the uncertainty in future land change patterns. They conclude that it is unlikely for the national policies to influence the land-use change patterns in the U.S. They highlight that effective management and planning of protected lands in the country will require understanding regional land-use dynamics.

Average biodiversity appears to decline in almost all 25 EU countries across all four scenarios (combinations of lean government versus ambitious government regulation; and globalization versus regionalization) in Verboom et al. (2007). The only exceptions are Germany, Latvia, Estonia, and Malta. While this is not exclusively due to urbanization, urbanization is expected to play a significant role together with increase in nitrogen deposition and disturbance in densely populated areas. According to these projections to 2030, it is unlikely that the EU will be able to fulfill its commitment to stop biodiversity loss in the near future. In another regional study focusing on Britain, two scenarios of urbanization (densification and sprawl) are examined to study the impacts of urbanization from 2006 to 2016 on ecosystem services of flood mitigation, carbon storage, and agricultural production (Eigenbrod et al. 2011). The scenario projections suggest that how ecosystem services will be impacted will largely depend upon the patterns of urbanization. While the mean change in peak (2 year return period) flows across British rivers is rather small under both scenarios, it is more than three times higher under the densification scenario. In terms of those affected by flood mitigation services, under the densification scenario, 1.7 million people would be living in areas within 1 km of rivers for which peak flows are projected to increase by at least 10 %, while 11,000 people would be impacted under the sprawl scenario. Calculations of carbon storage and agricultural production reveal that urbanization under the sprawl scenario will result in losses that are 3.5 times higher than urbanization under the densification scenario. Vimal et al. (2012) use a land change model to predict impacts of forecasted urban expansion across the French Mediterranean region. Over one third of the high-biodiversity sites in the region will potentially be directly impacted by urban expansion by 2030. Their study also confirms the differential vulnerability of coastal habitats to urban expansion, a recurring theme across the whole Mediterranean (Médail and Quézel 1997).

The published works in this section all come from developed regions of the world. However, it is the developing regions where the need for local to region level studies is especially acute because urbanization is progressing the fastest and more of the habitats are under threat in these regions. In general, local to region level studies may be more amenable to study the processes that govern various ecosystem services and interactions among them; detecting these processes is harder at larger scale or global studies that are generally designed to detect broader trends. Consequently, the resulting process-based understanding can inform urbanization strategies that are suitable to specific regional contexts.

22.3 Future Farming in Relation to Cities

Future urbanization will also have important effects on food systems. Urban expansion, coupled with unsustainable land management practices and climate change, will likely continue to lead to loss of agricultural land (Godfray et al. 2010). A recent estimate puts the amount of cropland loss due to urban expansion between 2000 and 2015 at about 400,000 km^2 (Nelson et al. 2010). This estimate does not include pastures and rangeland. However, a more significant, indirect, impact of urbanization may be due to diet shifts among urbanizing populations towards more meat and dairy-based food products (Satterthwaite et al. 2010). These shifts in dietary preferences will undoubtedly increase the pressure on agricultural lands because more land is needed to produce meat and dairy-based foods than vegetable and grain-based diet.

With an appropriate mix of policies and technological improvements, it may be possible to feed the burgeoning world population and at the same time temper or halt agricultural expansion. These interventions include improving yield of under-performing lands, increasing cropping efficiency as well as shifting diets back to more vegetable and grain-based ones, and reducing waste (Foley et al. 2011). These strategies might double food production while greatly reducing the environmental impacts of agriculture. Nevertheless, loss of cropland to urban expansion coupled with increased demand for food from a growing and urbanizing population may increase the incentive for both extensification and intensification (Godfray et al. 2010).

There are several paradigms on the nature of food systems. These paradigms can be seen as plausible future scenarios regarding the evolving relationships between urbanization, food systems, ecosystem services and biodiversity in the twenty-first century. Chapter 26 provides an in-depth examination of these scenarios and further information on food security and ecosystem support in an urbanizing world.

22.4 Challenges and Future Research Directions

Most analyses on the implications of forecasted urbanization on biodiversity and ecosystem services have emerged in the past two decades, in particular in the past 5 years. A multitude of factors beyond those included in these forecasting studies may influence urban expansion (Seto et al. 2011). Furthermore, mapping physical expansion of urban areas is not sufficient to calculate the full range of effects of urbanization on biodiversity and ecosystem services (see Chap. 21 for a comprehensive treatment of global urbanization trends). There are many indirect effects of urbanization due to the resource demands of residential, commercial, and industrial activities in urban areas (Seto et al. 2012b). Additional insights will be needed to formulate land-use change models that better reflect the complexity, diversity, and intensity of human influence on land systems (Letourneau et al. 2012).

Alongside the challenges of understanding and describing patterns of land-use change and urbanization, there are also challenges in approaching topics of biodiversity and ecosystem services. Biodiversity and ecosystems services are flexible concepts; studies must be clear in how they define these concepts in their specific contexts and select indicators/proxies for them. For example, there are many conservation prioritization concepts based on various criteria on which there is no general consensus among the conservation community (Brooks et al. 2006). The broad nature of these concepts leaves an inevitable gap in baseline knowledge in the scientific community (such as the full range of species richness and extent across the world), and may hinder study at the global scale of the impacts of urbanization. Additional work strategies between and among scholars and practitioners may be required to expand this base and further advance biodiversity science (see Chap. 32 for further discussion on indicators for management of biodiversity and ecosystem services).

There is a need for urbanization strategies that consider conservation of biodiversity (Niemelä 1999; Puppim de Oliveira et al. 2011) (Chap. 27). This is especially so in the case of developing countries where most urban expansion near PAs and in biodiversity hotspots are expected. In these places, urbanization strategies have the potential to affect the form of urban expansion with significant consequences for biodiversity. There are two crucial aspects of these efforts: First is to ground the research on the relationship between urbanization and biodiversity on a firm theoretical foundation (see Chap. 33); the second is making findings from this research accessible and useful to those who can most benefit from them. These include citizens, community organizations, planners, and government representatives alike. This dissemination of information and connection of science to practitioners will be an important tool for formulating more robust urbanization strategies that specifically consider biodiversity.

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Chapter 23
Regional Assessment of Africa

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Although there is large spatial variation in rates of change across the 55 nations of Africa, the combined impact of high natural population growth and rural-to-urban migration means that Africa is urbanizing faster than any other continent. At a growth rate of nearly 3.4 % per annum, Africa's urban population is the fastest growing in the world. Currently nearly 40 % of Africa's inhabitants live in cities (UN Habitat 2010), which is expected to more than double from 395 million people to 1 billion in 2040. In some cases, it is projected that city populations will swell by up to 85 % in the next 15 years. The Nigerian city of Lagos, home to 8 million in 2000, is anticipated to exceed 16 million by 2015. Several other cities such as Abuja, Abidjan, Addis Ababa, Kano, Kinshasa, Luanda, Nairobi and, Ouagadougou are all expected to grow by more than one million by the end of this decade.

Population expansion and a tradition of low-density settlement mean that the rate of increase in urban land cover in Africa is predicted to be the highest in any region in the world (see Chap. 1, Fig. 1.2). Current predictions pin this at a dramatic 700 % increase over the period 2000–2030. Expansion is expected to be focused in five main areas: the Nile River, the West African urban corridor between Abidjan and Lagos, the northern shores of Lakes Victoria and Tanganyika, the Kano region in northern Nigeria, and greater Addis Ababa, Ethiopia. All except the latter are very sensitive ecological zones.

For the most part, the urbanization in Africa is taking place along the lines of past and current patterns elsewhere in the world, but becomes distinct due to its extent and its rapid development. One significant pattern is the anticipated rapid growth in smaller towns. Based on current projections for 2010–2020, 74.2 % of Africa's total population growth will occur in cities of less than one million. These are often settlements with weak governance structures, high levels of poverty, limited infrastructure and services delivery, and low scientific capacity regarding biodiversity.

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Most importantly, many of these cities suffer from simultaneous weak environmental regulation and debilitating infrastructural backlog, both of which conspire to ensure that these cities are operating beyond the carrying and regenerative capacities of the biomes of which they are a part. As a result, African urbanization is increasingly functioning as an indirect – albeit significant – driver of biodiversity loss.

More than 43 % of Africa's urban population lives below the poverty line, higher than in any other continent, making socioeconomic development a priority. This situation is particularly acute in Sub-Saharan Africa where slum dwellers account for 65 % of the urban population. Unlike some other continents where urbanization resulted from the concomitant increase in agricultural and industrial production, urbanization in Africa is mostly driven by a different set of economic processes anchored around limited natural resource exploitation and export. A cursory observation shows that the growth of most African cities has occurred in proximity to resource extraction points. However, since point source natural resources are capital intensive, their contribution to employment is extremely small compared to their share of GDP. For example in 2007, employment shares in industry in Africa were 10 % compared to 24 % for Asia (UNTACD and UNIDO 2011). The narrow focus on resource extraction for the international market and a weak manufacturing and industrial base mean there are insufficient employment opportunities for the growing urban populace. African cities are growing at a rate that is disproportional to real employment opportunities. The result is a large number of urban populations that are compelled to live in unplanned and uncontrolled urban slums and work in informal, often low paying and unregulated, sectors. In this context of informality, poverty, and lack of infrastructure, the potential role of biodiversity to serve as a source of ecological infrastructure to address numerous human needs is paramount (Schaffler and Swilling 2013).

The generally weak state control, the preponderance of feeble formal economic sectors, and the scarcity of local professional skills place constraints on handling the complex biodiversity challenges faced by rapid urbanization. In some countries there is no government authority specifically tasked with city planning and development. For example while there is a Federal Capital Development Authority (FCDA) responsible for planning and development in Nigerian capital Abuja, many states within the Federation do not have government agency that is devoted to coordinating city development. In these urban centers crucial function of city development is played to varying degrees by different ministries in a very poorly coordinated fashion. Typically officials in these ministries have little understanding of the intricate functions provided by biodiversity and how to best preserve these. It was as recently as 2008 that the government of Kenya first established a separate ministry in charge of the development of the capital city of Nairobi even though the city had grown from 0.8 million in 1989 to 3.5 million in 2010 (MoNMED 2008). A "development-first-and-anyhow" mentality is pervasive among African policy makers. This results in poor planning and a majority of large-scale developmental projects being undertaken without vital environment impact assessment. Moreover, there is often lack of clarity about lines of responsibly between the various tiers of government with regard to the process of development in sensitive areas or the general management of critical biodiversity areas. For example the construction of the Gibe III dam on the Omo River in Ethiopia is being undertaken without detailed impact assessment on the lives of indigenous communities and several important biodiversity in lake Turkana, the world's largest desert lake. The situation is very much the same for other damns constructed or planned in countries such as Sudan, Nigeria, Mozambique, Ghana, Gabon, Republic of Congo and Mozambique (McDonald et al. 2009).

Because of the high level of informality and competing governance arrangements in Africa, especially around land-use management, conventional policy and regulatory measures used successfully to promote biodiversity in cities elsewhere in the world may not be effective here. However, the wide range of custodians of the rich biophysical resources and the high level of informality may also present opportunities for local and rapid adaptation to changing conditions in the urban landscape. One of the main criticisms of current attempts at biodiversity conservation in Africa is the continued pursuit of the bureaucratic pattern set by the colonial masters rather than harnessing customary conservation practices. It is argued that top down approaches to conservation are most exemplified by the establishment of nationally managed forest reserves in countries such as Nigeria alienate the people and vital indigenous knowledge-practice complex needed to ensure sustainable management (Gbadegesin and Ayileka 2000). There are indeed notable examples of good practices especially in Southern Africa where the communities are engaged in programs seeking to link wildlife conservation with economic development and poverty alleviation. These include the *Natural Resource Management Programme* in Botswana, the *Living in a Finite Environment project* in Namibia and the *Communal Area Management Programme for Indigenous Resources* (CAMPFIRE) in Zimbabwe. At the same time, it is worth stressing that population growth, rapid soil fertility loss and the pressing demand for economic development have all come together to pressure government and people into degrading valuable ecosystems all across Africa. The case of biodiversity conservation in Africa is a complex one, mired by historical environmental injustices and currently acknowledged as critical to future sustainability. A new path needs to be forged and one such opportunity lies in the urban transition to a 'green economy'.

The effects of urbanization on land cover in Africa appear to be unique. In the neotropics and Southeast Asia, urbanization and agricultural export markets are currently the strongest drivers of deforestation. In contrast, in much of Sub-Saharan Africa, old patterns of rural consumption of wood are still the major drivers of forest loss. However, there are significant variations across the continent. For example, in several West African cities, rapid population growth has increased incentives for farmers to convert forests into fields for crops to sell in urban markets. The recent land grab to secure African fuel and food production opportunities for urban citizens in other parts of the world is a stark reminder that cities draw not only on their immediate hinterlands for ecosystem resources.

It has been suggested that increased rates of rural-urban migration in Africa would relieve sources of pressure on old-growth forests and allow marginal agricultural lands to return to forest. This is indeed being witness in places but exactly what the ecological outcomes will be remains to be seen. However, there are others that would argue that given the continued expansion of the rural population, albeit at a lower rate than urban growth, it is questionable to what extent this is a general pattern. It is likely that increased local and international demand for biofuels and other cash crops may result in a new export-driven mode of deforestation, just as in Asia and the neotropics. Some of those export demands come from (an increasingly tapped out) Asia itself. Already China has established a significant presence in many parts of Africa, offering infrastructure – e.g., superhighways, flyovers and oil refineries – in exchange for access to natural resources.

Africa has generated ambiguous settlement forms: in addition to more conventional dense urban agglomerations, there is commonly a large peri-urban population and a cyclical pattern of rural and urban migration (Cotula 2009; Zoomers 2010). While a foothold in the rural environment is retained, the shift to urban livelihoods means that rural land-use patterns no longer retain the same degree of focus on production, but instead become landscapes infused with cultural and familial significance. Low levels of formal employment in African cities put a high level of dependency on the provision of ecosystem services, such as water, fuel, and food production, from areas within cities as well as nearby natural areas. Both within cities and in adjacent rural areas, biodiversity resource harvesting feeds into an extensive economy focused on supplying cities, and many of the people who have recently migrated to them, mainly with food and agricultural products. With as much as 84 % of population in some African countries depending on firewood for cooking and heating there is enormous pressure on wood reserves with little time for regeneration (IEA 2010).

Addressing urbanization and biodiversity challenges in Africa will require governance responses across the continent. In a Cities and Biodiversity Outlook workshop that brought together African researchers, local government authorities, and planners in February 2012, participants discussed common governance challenges and identified eight key themes of specific relevance to urban biodiversity concerns on the continent:

1. Many governments are still struggling with colonial legacy and the structures (or lack thereof) that withdrawal and transition have left in the wake of new government. For example, part of this debilitating legacy was excessively rigid zoning in central urban areas, which inadvertently encouraged informal settlements in the form of slums and sprawl because residential uses were prohibited in Central Business Districts (CBDs).
2. High political instability often exists, and may be accompanied by varying levels of corruption. This can result in high informality of tenure and economy. Particularly at the city level, lack of financial and human resources, and consequently technical capacity, can prevent biodiversity and environmental issues from being recognized or addressed.
3. In many instances, biodiversity concerns are seen as independent of and less important than other urban pressures such as poverty, unemployment, and access to food, energy, water, sanitation, and housing. These pressures are principally

the ones prioritized by politicians, who must act swiftly and expediently to meet the demands of their constituencies and who are mindful to receive good press to this end.

4. Where urban biodiversity interventions are implemented, they are generally undertaken with a single ecosystem service in mind, and multiple benefits are often neglected.
5. Even in governments where environmental-management issues receive recognition and support, it may be difficult to generate continued political momentum and action.
6. Barriers to integrating the environment with other issues may also be educational. Resources to inform those in government may be inaccessible or nonexistent, and academic terms and concepts that have been developed in other parts of the world may be difficult to translate into other languages and knowledge systems.
7. There is often a disconnect between scales of government, with lack of effective communication between local and national levels, disenfranchisement or mismanagement of local government by higher levels of government, and failure of national policy to be applied and implemented properly on the local scale. Fiscal decentralization needs to match political decentralization, municipal boundaries may need to be extended for greater control over land-use change in peri-urban areas, and accompanying management tools must have area-wide (i.e., metropolitan or even regional) reach.
8. While international resources and funds exist, there is a lack of access and transparency of process on how local governments procure these opportunities.

Ultimately, how biodiversity is managed or integrated into African cities will depend on whether it is first understood holistically, then positioned institutionally and topically as a priority in governance agendas, and whether the co-benefits provided by ecosystems are integrally recognized across general policy and action. Anticipated urban growth in Africa presents a window of opportunity to forge an urban form that could acknowledge and embrace the role of biodiversity. While this can assuredly be informed and aided by experiences gleaned from the urbanized global north, it must take as its point of departure the unique nature of urbanization in Africa, and engage with the particularities and opportunities presented by this continent.

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Chapter 24
Local Assessment of Cape Town: Navigating the Management Complexities of Urbanization, Biodiversity, and Ecosystem Services in the Cape Floristic Region

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Abstract The city of Cape Town, South Africa's most southwestern city, sits on a peninsula in the heart of the geographically restricted Cape Floristic Region, which is home to exceptional biodiversity. Within the city boundary are some 3,350 plant species, 190 of which are endemic to the city itself. Like all South African cities, Cape Town continues to grapple with development discrepancies that persist from unjust apartheid governance in the past and present-day challenges of urban sprawl. The population of the city is 3.7 million. Extreme poverty, with nearly 17 % unemployment, and extensive informal settlements characterize much of the City and stand in stark contrast to wealthy suburbs with freestanding homes. For the populace of Cape Town, the natural environment presents both considerable ecosystem service advantage with, for example, a flourishing tourism industry and provisioning opportunities for the urban poor, but also a significant hazard with, for example, exposure to flooding in winter from a high water table. The value of ecosystem services is an emerging concept in the environmental management arena, and environmental and conservation issues are still seen as separate to other areas of city development, and tend to receive a lower prioritization. South Africa has good environmental legislation, but this is sometimes weakly enforced due to conflicting demands, fiscal constraints, and/or lack of implementation mechanisms. Climate change predictions for the region suggest likely biodiversity impacts, but how these will play out remain unknown. An emerging interest in the role of ecosystem services in broader City management and novel conservation approaches involving civic interests all show considerable promise for the conservation of urban biodiversity in the city of Cape Town.

Key Findings

- Cape Town is home to exceptional biodiversity. The city is located in the Cape Floristic Region, the smallest and most diverse floral kingdom on earth. The region hosts almost 9,000 plant species on 90,000 km^2, some 44 % of the flora of the subcontinent on a mere 4 % of the land area. There are approximately 3,350 indigenous plant species in the city, of which 190 are endemic to the city itself.
- Cape Town's biodiversity is under significant threat. Some 450 of the city's indigenous plant species are listed as threatened or near-threatened, and 13 are known to be extinct. Urban expansion and development is the main culprit, but invasive non-native species and suppressed natural fire regimes also play a role. Conservation targets for national vegetation types indicate that all lowland area vegetation types are poorly conserved, fall below conservation targets, and insufficient remnants remain to conserve representative diversity. Small remnant patches can still contribute to conservation of remaining biodiversity however, and restoration efforts may prove important.

(continued)

Key Findings (continued)

- Future patterns of urban development must be directed to incorporate higher density, consideration for remnant patches of biodiversity, and social justice. The city is characterized by sprawl and prevalence of free-standing single-family homes. Historical planning stratified settlements and access to resources – including green spaces – along racial lines, and these legacies persist.
- While South African environmental policies are robust in their concepts, mechanisms to promote translation of multi-scale policy into practice must be strengthened in order to achieve greater accomplishments for biodiversity. Constraints of budget and conflicting priorities present limitations, and departmental and political affiliations must be bridged. Furthermore, biodiversity must be more effectively streamlined into humanitarian and development concerns in order to receive treatment as a priority.
- Innovative collaborations between citizens, government, and other organizations to address biodiversity and environmental management have proven fruitful in Cape Town, and add a complement to formal conservation areas. Tools for biodiversity conservation and enhancement include evaluation of ecosystem services, biodiversity mapping, and environmental education.

24.1 A Brief History of Settlement

The region around present day Cape Town has been inhabited for at least the last 21,000 years (Deacon 1992), initially by San hunter gatherers, and then from about 2,000 years ago by Khoi herders. The Dutch were the first Europeans to settle in the Cape, where they established a supply station to maintain passing trade ships between Europe and the East. The natural environment, which presented numerous ecosystem services such as the provision of perennial water and abundant wildlife, was a driving factor in all these historic engagements with the region (Anderson and O'Farrell 2012). The transitory vision of the European inhabitants saw dramatic and negative environmental impacts with the formation of large sprawling farms, the systematic removal of timber, altered fire regimes and the early canalization of rivers (Anderson and O'Farrell 2012). Cape Town only really emerged as a town when the area was formally colonized by the British in the early 1800s. Population figures grew rapidly from 45,000 people in 1875, to 67,000 in 1891 and 171,000 in 1904 (Worden et al. 1998). The most dramatic period of urbanization, however, occurred following World War II, when the population of Cape Town grew rapidly to 742,400 people in 1950 (Wilkinson 2000). The repeal of apartheid spatial segregation laws including the Group Areas Act in 1991 opened up the possibility of significant spatial reconfiguration of South Africa's population and between 1996 and 2001 the number

Fig. 24.1 Cape Town, located on South Africa's southwestern coast, exhibits a diverse geography of ocean, mountains, lowland vegetated areas, and built environment. The Table Mountain range – including Devil's Peak, Lion's Head, and Signal Hill, as depicted here – has remained largely undeveloped due to its steep topography and protected status as a national park. Table Bay, visible in the background, serves as the city's main port. The city center is nestled between the mountain and the bay. As the city has expanded outward, it has covered much of the depicted lowland areas, the Cape Flats. The remaining patches of open space in this complex city matrix contain some of the most valuable biodiversity remnants. Some of these areas are conserved as nature reserves or are protected under biodiversity stewardship agreements; others remain unmanaged and their fate is yet to be determined. While insufficient remnants remain to conserve representative biodiversity and achievement of connectivity is limited, collaboration and partnerships between communities, government, and other organizations present unique and innovative opportunities to make use of and manage these spaces (Photographed by and published with kind permission of ©Robert Kautsky/Azote 2013. All rights reserved)

of people in South Africa's towns and cities increased by 17.2 % (Christopher 2005). Nevertheless, these repealed laws were not always matched by policies to ease integration and desegregation of the urban configuration (Christopher 2005). Over the last two decades, population has climbed steadily to the 3.7 million that inhabit the City of Cape Town today. The configuration of the city continues to be sprawling in nature (Fig. 24.1) and the deep social and spatial divides established through apartheid planning persist (Turok 2001) (Fig. 24.2).

24.2 Biophysical Context to the City of Cape Town

Cape Town's central business district is located at the northern tip of the Cape Peninsula and the city is Africa's most southwestern metropole. The region exhibits a Mediterranean-type climate of hot dry summers and wet winters. Rainfall varies dramatically across the city from over 1,000 mm per annum in some places to as

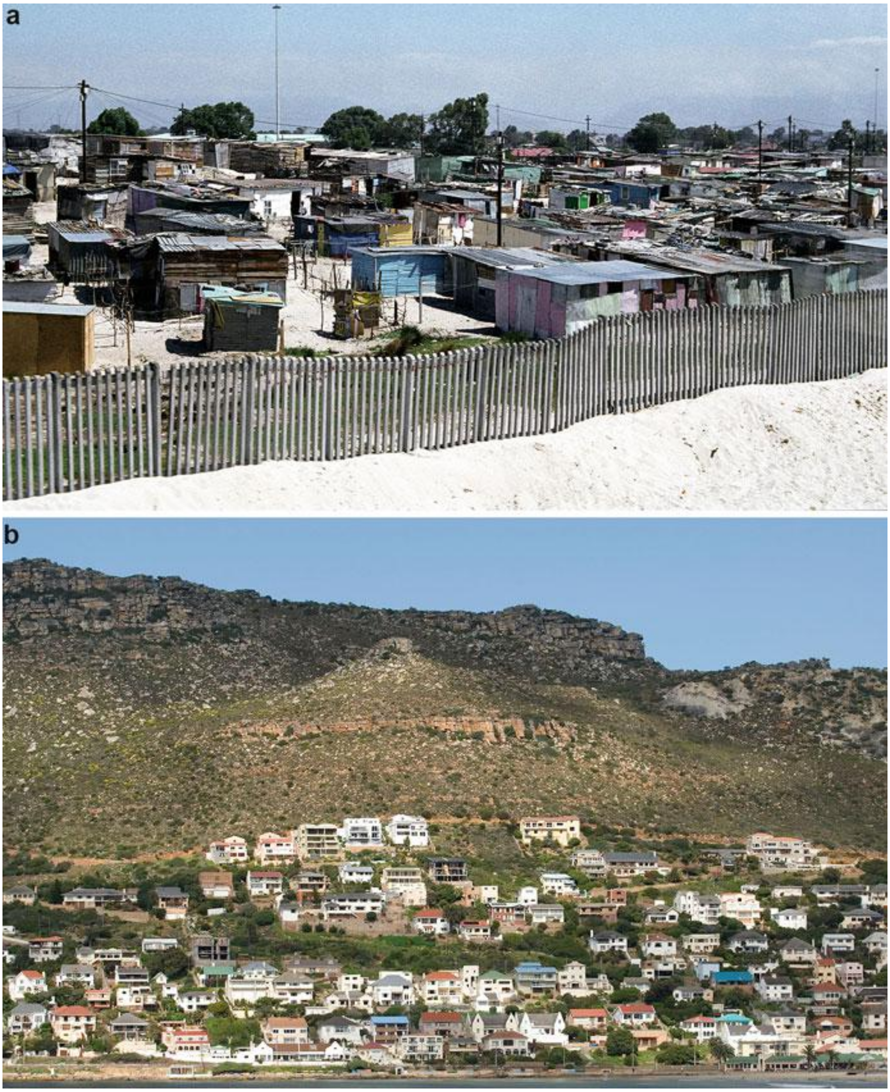

Fig. 24.2 Cape Town's built environment is characterized by a wide variety of residential structures, the patterns of which have been shaped by legacies of apartheid planning that demarcated communities along racial lines and stratified distribution of economic wealth and access to resources; this highlights persistent issues of social injustice. Across the city, in all areas and regardless of inequalities, the majority of housing consists of low-density, single-family units. As the city continues to develop, there is a need for planning policies and practices that achieve greater density, while preserving valuable biodiversity and ecosystem services as well as reducing exposure to environmental risk. (**a**) Shack dwellings are a common residential building type in Khayelitsha, located on the lowland sand dunes of the extensive Cape Flats. The dune areas experience flooding during the winter and are also subject to shifting movement, leading to unstable conditions for residents (Photographed by and published with kind permission of ©André Maslennikov/Azote 2013. All rights reserved). (**b**) Large single-family homes, embedded in the hillside between the Table Mountain range and the coast, are prevalent in Simon's Town, which is located on the southeastern side of the Cape Peninsula. Summer wildfires on the nearby mountain and human-wildlife conflict (mainly with baboons) present hazards to residents; storm surge and sea level rise also represent significant vulnerabilities (Photographed by and published with kind permission of ©Robert Kautsky/Azote 2013. All rights reserved)

little as 350 mm in others, and monthly average temperatures range between 25 °C for January and 17 °C for July (Mucina and Rutherford 2006). The region is biophysically diverse with rivers, wetlands, coastal areas, and different geological substrata. The resultant diversity of habitats supports a variety of flora and fauna and contributes to the high landscape-level diversity.

Cape Town is situated in the Cape Floristic Region, which is the smallest and most diverse of the earth's six floral kingdoms, and is noted as one of the world's biodiversity hotspots (Holmes et al. 2008; Myers et al. 2000). The region hosts almost 9,000 plant species on 90,000 km^2, some 44 % of the flora of the subcontinent on a mere 4 % of the land area (Mucina and Rutherford 2006). The process of urbanization over the last 350 years has significantly contributed to the erosion of the biodiversity of the region (Anderson and O'Farrell 2012). The vegetation is dominated by fynbos and renosterveld, both of which are types of low, shrubby, and typically fire-prone vegetation (Mucina and Rutherford 2006). The city hosts 19 of 440 National Vegetation Types. Of 21 nationally recognized critically endangered vegetation types, 11 are found in the city (Rebelo et al. 2011). Estimates place the number of indigenous plant species in the city at approximately 3,350, of which 190 are endemic to the city itself. Some 450 of these indigenous plant species are listed as threatened or near-threatened, and 13 are known to be extinct (Golding 2002; Rebelo et al. 2011). Animal numbers, while impressive in their diversity, do not share the same degree of endemism as the plant species in Cape Town (with the exception of invertebrates). Faunal counts in Cape Town yield the following: 83 mammal, 364 bird, 60 reptile, 27 amphibian, and 8 freshwater fish species (Rebelo et al. 2011). Two of the amphibian species are endemic to the city – a relatively high proportion – and ten amphibians are IUCN Red Listed as threatened. Remnant natural environments in the city are highly fragmented with little connectivity. The requirement of fire as a management tool where vegetation must burn on a 10–15 year rotation poses a further significant management challenge in the urban context.

The Table Mountain chain is situated in the heart of the city and, primarily by dint of its topography, has enjoyed a high degree of protection from development and its conservation largely is secured as a National Park. A recent land conflict involving expanding informal settlements on the border, though rapidly quashed, suggests this security may be challenged and contested in future (Fieuw 2011; Jordan 2010). Table Mountain National Park, comprised of 25,000 ha of land and surrounded on three sides by 1,000 km^2 of marine protected area, is one of the most significant conservation entities in the city. The lowland areas of the city, where the bulk of the diversity of vegetation types lie, do not share the same degree of protection and are under considerable threat from development. Conservation targets formulated for national vegetation types (Rutherford and Mucina 2006) show that all vegetation types confined to these lowland areas are poorly conserved and currently fall below their conservation targets, and insufficient remnants remain to conserve representative diversity. In these lowland areas there are a number of smaller nature reserves; most of these fall under the management of the City of Cape Town local government, and one reserve is administered by the provincial authority, CapeNature. The scale, number, and connectivity of these smaller reserves do not meet identified conservation goals. Conservation areas in the city are both enjoyed by and pose a number of challenges to local residents. Accidental fires started inadvertently by people may lead to overly frequent and uncontrolled burning, which can threaten people and property. This is particularly relevant in the hot, dry, summer months when strong winds fan the flames (van Wilgen et al. 2012). Remnant lowland areas are generally too small and fragmented to allow effective ecological burns. Animals from conservation areas, in particular baboons, are frequent visitors to adjoining neighborhoods where they scavenge for food and become problematic, leading to typical human-wildlife conflict (Hoffman and O'Riain 2012). Formal housing development is a driver of the ongoing conversion of remnant land. Informal settlement encroaches on both remnant patches and on formal conservation areas, with ensuing removals and complex associated social conflict (Fieuw 2011; Jordan 2010). The extensive network of rivers and wetlands in and around the city has been heavily impacted by the process of urbanization. For example, upper reaches have been cut off from broader systems through poor spatial planning, and lower reaches are heavily polluted. Some systems have been severely modified by inappropriate engineering interventions such as canalization in order to address problems of urban flooding (Brown and Magoba 2009). Water quality standards in terms of public health (recreation) and ecosystem health have not been met for approximately 50 % of city's river and wetland systems over the past five hydrological years, signaling that water quality is a long-term and significant issue (City of Cape Town 2013). The management challenges of meeting these multiple – and frequently conflicting – anthropogenic and conservation goals are readily apparent.

24.3 Socio-economic Context

South Africa is a young democracy and one that is dogged by the legacies of apartheid, which manifest in developmental, educational and wealth discrepancies. Cape Town has a population of approximately 3.7 million – about 70 % of the population of South Africa's Western Cape Province, in some 904,000 households (City of Cape Town 2009c). The city has exhibited an annual growth rate of 3.2 % (for figures between 2001 and 2007, City of Cape Town 2010), which is higher than the national average (Mieklejohn and le Roux 2008). The region can be described as water-scarce and urbanization places huge demands on this limited resource. Since 2000, water demand has on occasion outstripped available supplies, particularly in dry years. In light of this, publicity campaigns and water restrictions have been implemented with successful reductions in demand (Brown and Magoba 2009; City of Cape Town 2013). Despite this success, the rate of urbanization continues to be a major problem and options such as reduction of water wastage and the re-use of grey water must be considered (City of Cape Town 2013). The physical footprint of the city is extensive at 2,460 km^2 and characterized by urban sprawl and the stark contrast of middle- to upper-income areas of freestanding houses on large plots adjacent to extensive and rapidly expanding informal settlements (Rebelo et al. 2011) (Fig. 24.2). These informal settlements, and formal historic townships, established during the course of the previous century and enforced through apartheid planning, tend to be on the biodiversity-rich lowlands, also known as the Cape Flats. Following World War II there was a dramatic increase in population and the city expanded in an easterly direction, wrapping around the base of Table Mountain on remaining readily accessible land, to the detriment of the Cape Flats Sand Fynbos and Renosterveld vegetation types (Rebelo et al. 2011) (Fig. 24.1). The adjacent lowland areas to the north were not formally settled until the second half of the twentieth century, when heavy machinery allowed the extensive dune system to be flattened for housing. Much of this took place at the height of apartheid planning in the 1960s and 1970s, and this spurred the bulldozing of much of the Cape Flats Dune Strandveld vegetation; the result was that large areas of dune slack wetlands were populated with low-income housing (Rebelo et al. 2011). These neighborhoods today still experience seasonal inundation and flooding in winter due to the high dune slack water table. These areas were cleared for housing for non-white communities during apartheid and are by-and-large the same communities that occupy them today. These same regions host the city's informal settlements; collectively, this highlights significant social justice challenges faced by the city. The demand for housing continues to place a significant burden on city authorities and hence on remnant biodiversity in the city.

Poverty is characteristic of Cape Town, where as much as 38 % of households earn less than the Minimum Living Level of US$230 per month (in 2010) (City of Cape Town 2007). The city's population has a high burden of disease, in particular of HIV and tuberculosis (City of Cape Town 2007). Respiratory conditions are exacerbated by a brown haze (induced by particulate matter attributable to exhaust fumes and smoke from wood burning in informal settlements) which frequently rises in excess of World Health Organization (WHO) levels. Khayelitsha, a township of the Cape Flats and most affected by the brown-haze phenomenon, experienced 86 days in 2006 in which atmospheric particulate matter was above WHO standards (Wicking-Baird et al. 1997; City of Cape Town 2007). Education levels, while better than other areas in the country, are generally low in Cape Town with 58 % of the adult population educated to a standard lower than matriculation (Grade 12) (Statistics South Africa 2010). Unemployment, while again below the national average, is still high at 16.9 % (City of Cape Town 2007).

A recent study into illegal resource harvesting from remnant patches of natural vegetation gives a list of 448 locally occurring species (198 animals and 250 plants) that are harvested and sold. These ecosystem provisioning services support a large informal economy with significant livelihood implications (Petersen et al. 2012). With respect to the formal economy, it is conservatively estimated that for city natural assets, or green infrastructure, there is a flow of services valued at R4 billion per annum (de Wit et al. 2009). Most of this value for Cape Town is created through the tourism industry, but recreation in parks, open spaces, and beaches, as well as specific industries such as film-making also benefit substantially from the services provided by well-functioning ecosystems. Natural landscapes and biodiversity are major drivers in the tourism industry in which, for example,

Table Mountain National Park receives 4.2 million visitors a year (UNEP 2009). For additional detailed information on evaluation of ecosystem services in Cape Town, see Chap. 11.

24.4 Emerging Challenges to Biodiversity Conservation and Stewardship in Cape Town

Challenges to the conservation and stewardship of the biodiversity of Cape Town include ongoing land conversion at odds with a biodiversity conservation agenda, suppression of indigenous vegetation by invasive non-native plant species, overexploitation and degradation of natural resources, variable perceptions regarding needs for conservation of biodiversity, and inequitable access to environmental space and resources. Pressure to address development issues of unemployment, poverty, and a significant formal housing shortfall all place considerable demand on remnant vegetation patches, which are highly sought after for conversion to housing or industrial development. Administering to these important humanitarian issues frequently takes precedence over the conservation of the natural environment (Goodness 2013), and this is certainly evident during electoral campaigning. Indeed, there is very little evidence of an understanding that green space or biodiversity is linked to human wellbeing, and there is a lack of vision for finding synergies between pressing environmental and humanitarian issues (Goodness 2013). Perceptions around the validity of conservation vary, and the large, highly-visible tract of conserved land in the heart of the city, Table Mountain (which conserves only a few of the already better-conserved vegetation types), drives a misconception that biodiversity is well-protected in the city. Generally, perceptions around remnant vegetation and biodiversity conservation vary. Remnant patches do play a significant utilitarian role where people enjoy the cultural ecosystem services of these areas for recreation, ceremonial, and aesthetic purposes. While in some instances there is considerable civic support for biodiversity initiatives (Pitt and Boulle 2010; Ernstson 2013a), in other cases the indigenous vegetation is seen as unappealing and other landscape forms, for example non-native tree plantations, are viewed as preferable (Ernstson 2013b; Van Wilgen 2012) (Fig. 24.3). In a city with a high crime rate (City of Cape Town 2007), wilderness areas are frequently perceived as dangerous and perceptions around remnant patches can be negative (Holmes et al. 2008; Goodness 2013).

The high turnover in diversity both within and between vegetation types (i.e., beta diversity) means that small remnant patches can make significant contributions to biodiversity conservation. The Spatial Development Framework for the City promotes densification with a view to addressing the question of urban sprawl (Holmes et al. 2012). While this is laudable with respect to a broader landscape vision and city environmental efficiency, this restriction of development to areas within defined urban edges presents a challenge to those few remnant patches of conservation-worthy vegetation and associated biodiversity in the city. This is particularly relevant

Fig. 24.3 Varied perceptions and sentiments exist in regard to biodiversity in Cape Town, and negotiation of these issues is complex. In some cases, there is considerable support for native biodiversity, while in others, alternative landscape forms are preferred. In one example, an accidental fire in the Tokai forest (a plantation consisting of non-native pines, but historically present since 1885) triggered the sprouting of dormant fynbos seeds and reemergence of native fynbos vegetation. Cape Town residents were divided in their opinions on which landscape form should prevail. Those in favor of fynbos cited the benefits bestowed by restoration of precious native biodiversity and reduction in stress to the water table (through elimination of water-thirsty pines). Those in favor of the tree plantation countered the "native biodiversity" argument with the long-standing historical record and cultural value of the pines, and listed benefits of recreational activities such mushroom harvesting and walking beneath a shaded canopy. The debate is ongoing; this exhibits some of the tradeoffs between biodiversity conservation and ecosystem service agendas (Ernstson 2013b). In the photograph, an area of restored fynbos is visible amidst plantation pines (Photographed by and published with kind permission of ©Julie Goodness 2013. All rights reserved)

on the lowlands, where demand is highest and conservation needs most critical, compounding current biodiversity conservation and management with further fragmentation. The proposal to densify is additionally complicated by the suggestion that National Government housing grants cannot accommodate the suggested denser housing models (Holmes et al. 2012). Denser housing models need to be promoted, as well as spatial development models in which the importance of biodiversity remnants is recognized and densification is not only constrained by the outer city boundary. When land does become available to purchase for conservation, there are often fiscal constraints – either for the actual purchase or for sustainable management – that prevent the land being secured (Holmes et al. 2012). Furthermore, even where critical biodiversity has been identified, it is often difficult for City conservation officials to monitor, and delay or prevent disturbance to these patches, which are often small, fragmented, and spread over a large geographic area. Thus, remnants may often be destroyed in the wake of development (Yeld 2011). An overarching concern is the lack of a detailed understanding of the ecological functioning of species in the urban context. Species-specific studies (for example, among nectar feeding birds) that explore functioning ecology in relation to the configuration of the urban space, suggest sensitivity and complex responses that caution against generalizations (Pauw and Louw 2012). Ongoing empirical research is needed to inform future policy and plans.

Invasive non-native plant species, many of which were introduced from other colonial regions (Anderson and O'Farrell 2012), are a significant problem in the region, where they proliferate and suppress local biodiversity, and use considerably more water than indigenous plant species. The problem is extensive; for example, in 1996, 24 % of the Table Mountain National Park was invaded by non-native plant species (Richardson et al. 1996). In terms of prevention, the South African National Biodiversity Institute (SANBI) hosts an "Invasive Aliens Early Detection and Rapid Response Program"[1] that seeks to identify and address invasive non-native plants before they become a problem. In terms of intervention, in 1995, the South African Government established its Public Works Working for Water Programme (Turpie et al. 2008). This programme, a joint environment and poverty alleviation programme, is aimed at clearing invasive non-native vegetation in order to improve water catchment management, and has a simultaneous agenda of empowerment and poverty alleviation. The programme has made some noteworthy inroads and simultaneously demonstrates clear ecosystem service linkages between poverty alleviation and biodiversity support (Turpie et al. 2008). The problem of invasive non-native plants persists however, and research suggests there is still a need to expand our understanding of the ecological implications of clearing and rehabilitation methods to ensure a positive and lasting environmental impact. In the interim, the number of invasive species and areas invaded continue to expand.

Attention needs to be given to the potentially unsustainable harvest of biodiversity resources (Petersen et al. 2012). This is not just a simple issue of management and policing, but relates to poverty alleviation and livelihoods, as well as cultural practices and expectations. Historical apartheid planning has given rise to a city where the bulk of the urban poor are situated in locations far from the major conservation areas. Recent analyses reveal that that this trend persists; poor access to managed green open space (particularly larger nature reserves) is shown to be concentrated in areas of lower socio-economic status (City of Cape Town 2013). In a time when it is known that all people need and deserve access to the cultural ecosystem services provided by nature, this issue of access needs attention. It could be argued that the urban poor require greater access to nature for the additional provisioning service it provides, where even in the urban setting Cape Town residents

[1] http://www.sanbi.org/programmes/conservation/invasive-aliens-early-detection-and-rapid-response-programme/invasive-aliens-early-detection-rapid-respo

still hunt small mammals and collect wild fruits and vegetables (Petersen et al. 2012). While all urban biodiversity issues require more interdisciplinary research with greater input from social scientists, these last two areas of concern around access to and use of both provisioning and cultural ecosystem services are particularly pertinent to the sensitive social agenda of a society in the process of addressing developmental discrepancies imposed in the past.

24.5 How Are These Challenges Being Addressed in Cape Town?

A variety of governmental structures have been put into place which address biodiversity issues in Cape Town. These include legislation and agreements, institutions, and programmes that range in scale from the international level down to the local. At the international level, South Africa is a signatory to the 1982 World Charter for Nature, the 1992 Rio Declaration on Environment and Development, the Convention on Biological Diversity (CBD) (with commitments in Rio de Janeiro in 1992 and Nagoya in 2010), and the IUCN Countdown 2010. These agreements establish terms that are used in South African legislation, including sustainable development and biodiversity, but they are non-binding agreements. The City of Cape Town also became a signatory to the Durban Commitment in 2008, and is a pioneer member of the ICLEI Local Action for Biodiversity (LAB) programme, which is a network of municipalities working to categorize their biodiversity and share tools and best practices of biodiversity management.[2] The LAB programme cities also seek to call international attention to the importance of urban biodiversity and the role that local governments can play in maintaining this biodiversity. As part of LAB, the City of Cape Town has selected five "biodiversity implementation projects" to address key biodiversity challenges in the city (Chap. 30), and has produced a number of publications, including a city Biodiversity Report, a Local Biodiversity Strategy and Action Plan (LBSAP), a Biodiversity and Climate Change Assessment Report, and a Biodiversity and Communication, Education, and Public Awareness (CEPA) Report (for additional discussion of the CEPA Report and associated Evaluation Design Toolkit, see Chap. 30).

At the national level, the 1996 South African Constitution outlines and establishes basic environmental rights, and assigns powers and functions. The Constitution's Bill of Rights states that all South Africans have, "the right to an environment that is not harmful to their health and wellbeing; and to have the environment protected, for the benefit of present and future generations." There are several pieces of legislation, which have direct implications for biodiversity, that have been enacted as a result of this constitutional provision. The National Environmental Management Act 107 of 1998 (NEMA) serves as the main structure that establishes principles

[2] http://www.cbc.iclei.org/lab-about

and procedures for environmental management, assessment and governance. The Protected Areas Act 57 of 2003 (NEM:PAA) and the Biodiversity Act 10 of 2004 (NEM:BA) both address biodiversity conservation. Within the Biodiversity Act, four main tools are outlined: (1) NBSAP (National Biodiversity Strategy and Action Plan, 2005), which provides a framework and plan for conservation and sustainable use of South Africa's biodiversity, (2) NBA (National Biodiversity Assessment, 2011), which outlines the threat status and protection levels of ecosystems within the country and provides a frame for the development of provincial and local spatial biodiversity assessments and plans (preceded by the National Spatial Biodiversity Assessment 2004), (3) NPAES (National Protected Area Expansion Strategy, 2008), which provides an action plan for acquiring and aggregating land for conservation (particularly land that can be acquired economically and linked to existing areas), and (4) NBF (National Biodiversity Framework, 2008) which sets out 33 priority biodiversity actions for the country (Holmes et al. 2012). The Biodiversity Act also established the South African National Biodiversity Institute (SANBI),[3] which conducts research, monitoring, and reporting on South Africa's biodiversity, as well as manages the National Botanical Gardens (including Kirstenbosch in Cape Town). In addition to these pieces of legislation, the National Water Act 36 of 1998 (NWA) also has impacts for biodiversity, as it mandates that water resources must be managed for the protection of aquatic and associated ecosystems and their biological diversity.

The national implementing agency for the environment is the Department of Environmental Affairs (DEA).[4] The DEA also administers the South African National Parks (SANParks)[5] agency, which oversees the management of Table Mountain National Park in Cape Town. A set of flagship successful national government programmes are the Working for the Environment Programmes (including Working for the Coast, Working for Water, Working for Wetlands, Working on Fire, and Working on Waste); these are conducted under the umbrella of the national Expanded Public Works Programme (EPWP) that supplies local residents with livelihoods through training and work in environmental restoration and management. Activities include rehabilitation of coastal areas and upgrading of tourist infrastructure, clearing and removal of invasive non-native species, rehabilitation and protection of wetlands, and establishment of integrated fire management through regulating vegetation with prescribed burning, fighting unregulated fires, and educating communities about how to protect lives and property (DEA 2011).

National legislation is further implemented at the provincial and municipal levels. The provincial agency for the environment is the Department of Environmental Affairs and Development Planning (DEA&DP), Western Cape Government.[6] In addition, an entity called CapeNature was established as a DEA&DP parastatal responsible for biodiversity conservation in the Western Cape. It is governed by the

[3] http://www.sanbi.org/

[4] http://www.environment.gov.za/

[5] http://www.sanparks.org/

[6] http://www.westerncape.gov.za/eng/your_gov/406 and http://eadp.westerncape.gov.za/home

Western Cape Nature Conservation Board Act 15 of 1998 and is mandated to "promote and ensure nature conservation, render services and provide facilities for research and training, and generate income".[7] Among its conservation activities, CapeNature provides scientific services for research and evaluation, youth development programmes for education and skill building, and eco-tourism at provincial nature reserves. Two of these nature reserves, Driftsands and Hottentots Holland, overlap with the Cape Town city area. CapeNature is also involved in helping to establish biodiversity stewardship agreements with private landowners, with options for differing legal categories, including contract nature reserves, biodiversity agreements, and conservation areas.

At the municipal level, the City of Cape Town has an Integrated Metropolitan Environmental Policy (IMEP), an auxiliary Biodiversity Strategy, and a Local Biodiversity Strategy and Action Plan (LBSAP). A review of the IMEP has produced the City of Cape Town Environmental Agenda 2009–2014, which outlines 17 detailed goals and targets for environmental sustainability in Cape Town (City of Cape Town 2009a, 2013). The City agency is the Environmental Resource Management Department (ERMD), which contains a Biodiversity Management Branch (BMB).[8] The ERMD runs programmes and initiatives on a range of focus areas relating to biodiversity, including environmental education (Fig. 24.4), climate change management, coastal management, and invasive non-native species management. The BMB is responsible for managing 32 conservation areas (16 Contract Nature Reserves and 16 Biodiversity Agreements) which differ widely in size, location, and dominant landscape and vegetation type. These conservation areas have undergone proclamation for protection in perpetuity under the NEM:PAA (Holmes et al. 2012). While significant in number, currently established conservation areas do not secure a representative sample of terrestrial biodiversity in the city. To address this issue, the BMB has used a systematic biodiversity assessment to analyze minimum targets. Cape Town's first comprehensive systematic biodiversity plan, the Biodiversity Network (BioNet), was produced in 2004, and has been periodically updated to include new data and to conform to national vegetation requirements (Rebelo et al. 2011). While the BioNet does not yet have legal status to serve in the protection of land, it can serve as a flag during the Environmental Impact Assessment (EIA) process. In addition, while the BioNet cannot be in conflict with the City's Spatial Development Framework (SDF), the 2011 city-wide SDF, the eight district Spatial Development Plans (SDPs) and accompanying Environmental Management Frameworks (EMFs) have incorporated the BioNet and thus have established a path for future implementation action. Furthermore, a process begun in 2010 to produce a Bioregional Plan for the city (in line with the NBF and sanctioned under the NEM:BA) will provide legal status to the BioNet (Holmes et al. 2012). Beyond the scope of the nature reserves, the BMB works in conjunction with CapeNature to secure biodiversity stewardship agreements with

[7] http://www.capenature.co.za/

[8] www.capetown.gov.za/environment

Fig. 24.4 A number of initiatives and programs across Cape Town are working to conserve and manage biodiversity as well provide environmental education. (**a**) Nature camps run by the City of Cape Town and held in the nature reserves of the False Bay Ecology Park provide opportunities for learners in team-building, physical exercise, and nature discovery. (**b**) The Kirstenbosch Bus provides free transport for financially limited school groups to visit the National Botanical Garden located in the city. (**c**–**d**) The Biodiversity Showcase Garden, established by City of Cape Town following the 2010 World Cup, provides an experiential overview of Cape Town's native biodiversity, and serves as both a resource for citizens, and a field trip destination for students (Photographed by and published with kind permission of ©Julie Goodness and Katie M. Hawkes 2013. All rights reserved)

public and private landowners, and has also worked with communities to establish creative methods of land management (Colding et al. 2013). Groundwork has been laid to establish and fund additional mechanisms for the purchase of conservation land; the City has initiated research on sourcing and making connections with international and other funding agencies, and is also working to study, evaluate, and streamline the value of ecosystem services into City government decision making (de Witt et al. 2009).

Outside the sphere of government in Cape Town, there are a number of NGOs, nature societies, friends groups, neighborhood groups, and individuals that organize at the level of the community. A recent compilation of such organizations can be found in the City of Cape Town Environmental Resource Directory (City of Cape Town 2009b). In particular, a well-documented example of community-based

Fig. 24.5 In Cape Town, grassroots collective action at former derelict spaces outside of protected areas have expanded the possibilities and ways by which biodiversity protection can be sustained, but also how one can speak about urban nature in a post-apartheid city. Although assisted by governmental organisations and NGOs, these rehabilitation efforts are grounded in local communities moving 'biodiversity' beyond scientific discourse and into the imaginations of popular memory and action (Ernstson, 2013a). Depicted in the photograph is a resident-led initiative in Grassy Park that partnered with City of Cape Town biodiversity managers and the national Working for Water Programme to restore fynbos vegetation on the edge of the wetland of Zeekoevlei (Photographed by and published with kind permission of © Henrik Ernstson 2013. All rights reserved)

environmental stewardship is the Grassy Park neighborhood in Cape Town, where residents have partnered with city and national government entities to restore fynbos vegetation alongside the vleis (wetlands) near their homes (Ernstson 2013a; Ernstson and Sörlin 2013; Colding et al. 2013) (Fig. 24.5).

Innovative structures and entities have also been formed to incorporate action for biodiversity across spatial scales and organizational boundaries. One example is Cape Action for People and the Environment (C.A.P.E.), a partnership of government and civil society aimed at "conserving and restoring the biodiversity of the Cape Floristic Region and the adjacent marine environment, while delivering significant benefits to the people of the region." C.A.P.E. is comprised of 23 signatory entities that include government departments, municipalities, non-governmental and community-based organizations, and conservation agencies.[9] All have signed on to the Cape Action Plan for the Environment Strategy (also called the C.A.P.E. 2000 Strategy),

[9] http://www.capeaction.org.za/

a document which was designed through consultative process and outlines eight strategic objectives to conserve the biodiversity of the region (Ashwell et al. 2006; C.A.P.E. 2000). C.A.P.E. streamlines donor funding into projects that cover a variety of work areas, including landscape initiatives, conservation stewardship, business and biodiversity, fine-scale planning, catchment management, and strengthening institutions. Progress is tracked through a monitoring and evaluation system.

Another example is Cape Flats Nature, a South African National Biodiversity Institute (SANBI) partnership initiated in 2002 to explore models of practice for people-centered and community-orientated management of nature reserve sites in Cape Town.[10] Predicated on the understanding that successful nature conservation in urban areas must incorporate the social systems of the region, the programme focused on developing local leadership skills and action for conservation, and finding ways that nature sites could bring tangible benefits to communities in the form of ecosystem services, economic value, and social development. The project worked intensively with the managers of six City of Cape Town nature conservation sites located in the Cape Flats, the lowland landscapes that contain the highest levels of biodiversity in the city, but are also characterized by extensive informal settlements, where generally income is low and living conditions are poor. Through methods of experimentation in practice and peer case study learning, managers were able to explore how to best administer biodiversity sites for their surrounding communities. While the programme was concluded in 2010, participant nature reserve managers still utilize the structures, tools, and connections established. Additionally, a publication that documents experiences and case studies of the programme has been compiled as a general resource for urban conservators (Pitt and Boulle 2010). For a detailed description and analysis of Cape Flats Nature as a Cape Town "learning arena," see Chap. 30.

A further interesting development is the inclusion of a private sector component in support of urban environmental management. For example, the City of Cape Town contractually employs a private company for support in the protection and management of the city's baboon population (Hoffman and O'Riain 2012). This arrangement provides a solution to capacity shortfalls wherein the City lacks staff, expertise or resources to implement policies or plans.

24.6 A Glance to the Future: Trends and Opportunities

The Cape Town region is likely to face significant climate change related issues, with predicted increases in temperature in all seasons, reductions in rainfall, greater evaporation, more intense and frequent wind, and greater coastal erosion and storm surge with changes in the frequency and intensity of extreme weather events (City of Cape Town 2011a). This is likely to influence wild fire season, frequency

[10] http://www.capeflatsnature.co.za/

and intensity. The intensity and season of a fire has implications for biological processes of recruitment and regeneration (Cowling et al. 1996), and changes in fire regime will have repercussions for biodiversity. Plant species in the Cape may be adapted to summer drought, but prolonged periods of drought or shifts in rainfall seasonality could to lead to declining numbers or possible extinctions (Yates et al. 2010). The limited pace at which plant species can migrate, combined with edaphic restrictions may present extinction risk for some species in the face of climate change (Loarie et al. 2009; Cowling and Holmes 1992), though it is possible that species can adapt in situ, as well as move small distances to find a suitable niche. Rapid increases in the intensity of coastal storm surge will have negative implications on coastal diversity, particularly in light of development close to the coast, where there is little space for processes and biodiversity to relocate. The role of *in situ* regulatory ecosystem services will become critical (O'Farrell et al. 2012). Recent initiatives by local government to identify and spatially map ecosystem services in the city suggest a good start in seeing these elements included in development planning, though it is certainly important to remain cognizant of caveats of such methods and amenable to additional "ways of knowing" and valuation (Ernstson 2013b; Ernstson and Sörlin 2013) (Fig. 24.3).

While projections suggest the growth rate for the city of Cape Town is likely to slow down in coming years, the population of Cape Town will still continue to grow with ongoing rural-urban migration and in-migration from other countries in Africa (City of Cape Town 2006). In-migration may well be encouraged by climate change impacts elsewhere in the region. Predictions for warming in southern Africa, just north of South Africa, are particularly severe (IPCC 2007). This will in turn have implications for employment, demands on city services both infrastructural and ecological, and food security, which will all in turn have consequences for biodiversity in and beyond the city. Efforts to move towards densification are in conflict with current models of development in which, for example, there are expectations in relation to perceptions around social norms of single-family dwellings (Fig. 24.2). Pressure for more sprawling middle-income housing is frequently allowed in light of the potential revenue such developments generate both in their establishment and in on-going rates and taxes, all of which are relevant in the fiscally-constrained circumstances.

Restoration of urban biodiversity is already in many instances the only route open to meeting conservation goals (Avlonitis 2011). Restoration for biodiversity and ecosystem service delivery is likely to grow in demand, and attention will need to be given to appropriate techniques and sourcing associated costs. Establishing those areas of critical ecosystem service delivery will be important (O'Farrell et al. 2012). The identification of these areas should direct restoration efforts and guide planning. There is potential to bring together the current stewardship model with identified areas of ecosystem services delivery where remnant land can be conserved to multiple ends. Of course, biodiversity and ecosystem service delivery do not always coincide, and this is particularly relevant to consider in the biodiversity-rich Cape Town area (see Chap. 11 for additional discussion of the possible divergence between ecosystem services and biodiversity conservation agendas). Furthermore, green areas of high social value do not always correspond to biodiversity and ecosystem service priorities identified by scientific analysis or government (Fig. 24.3). Greater imagination and vision may be required in order to navigate the potential of sites, as well as negotiate possible problematic political and power dynamics inherent in "expert-based Cartesian practice of controlling space, embodied in the form of expert-managed nature reserves and biodiversity mapping techniques that calculates the 'value' of green areas by counting the number of species they contain" (Ernstson 2013a, p. 5, 2013b; Ernstson and Sörlin 2013). Citizens as well as formal institutions can drive legitimate and useful systems of environmental management (Figs. 24.4 and 24.5).

Current issues around the implementation of multi-scale environmental policies are unlikely to be resolved in the immediate future, and blockages between tiers of government could grow depending on relations between National, Provincial and City leaders. Gaps persist in finding effective mechanisms to translate well-crafted conceptual ideas of policy into pragmatic action. The exclusion of biodiversity and the environment from the realm of 'priority' in the political agenda is likely to persist as a problem into the future as growing population and climate change related impacts place greater humanitarian demands on government. There is currently a lack of vision as to how 'environment' fits into those categories earmarked as of priority (e.g., housing, planning, economic development and poverty alleviation) (Goodness 2013), and a false dichotomy between people and nature persists (see Chap. 2). Though connections between social and ecological systems may be recognized, they still receive separate treatment in reports (City of Cape Town 2013, p. 4); concomitant management presents challenges. While issues relating to the environment retain a conservation or preservation flavor, they are unlikely to be picked up as a priority by the government. Current paradigms of conservation have, in some cases, been perceived by South African officials as carrying negative connotations of being socially unjust (reinforcing apartheid racist systems in which the black population was denied access to resources held by the minority white population), disrespectful to people (especially to the poor, through blocking development), and utopian (with conservationists failing to understand more pressing issues, such as people's need for access to basic services) (Wilhelm-Rechmann and Cowling 2010). How the environment is viewed by those in government will have implications for fiscal planning. Ecosystem services have the potential to raise awareness and highlight the importance of remnant patches in and around the city, and this approach could be used to leverage funds. There is considerable potential to expand and further the proliferation of environmental education (Fig. 24.4), which has been shown to have significant benefits in growing environmental awareness (Ashwell 2010; City of Cape Town 2012), and the remnant vegetation patches in the city could be used to this end as spaces for learning. For three detailed examples of best practices and projects in environment education in Cape Town, see Chap. 30.

Ultimately, the next few years will be critical in determining the fate of Cape Town's biodiversity; if current development patterns are left unchecked, the opportunity to secure this resource will soon be lost (Holmes et al. 2012). Efforts from a variety of actors will be required, with increasing emphasis on and support for cross-cutting, collaborative, creative and innovative partnerships that bring people with different skills and powers together. There will be a significant need to strengthen connections across tiered levels of hierarchy to create action on the ground, with a flow between the rich structures, concepts, and capacity at the international and national levels, and the vibrant energy, knowledge of practice, and plasticity at the local city, community, and individual levels. Within the sound basis of a biodiversity agenda that has been established in local City government through the Environmental Resource Management Department, energy must now be directed towards transcending traditional barriers of departments and line functions, as well as political affiliation. Finding ways to streamline and integrate biodiversity concerns into core City actions and focus areas such as health, economic and social development, and service delivery will be key. Furthermore, demonstrating the tangible benefits and importance of biodiversity and ecosystem services in the city will be essential to this end. While these processes may take time, it is important, in the interim, to utilize existing legal mechanisms of land use protection and planning, and attempt to leverage funds for conservation and land management, whether locally or internationally. International designations and events also provide unique possibilities to marshal attention and resources for environmental issues in Cape Town, as in the case of the 2010 World Cup tournament (City of Cape Town 2011b) (also see Chap. 30 and Fig. 24.4), Table Mountain's title as one of the New Seven Wonders of Nature, the city's areal overlap with UNESCO World Heritage sites and Biosphere Reserves, and Cape Town's designation as the 2014 World Design Capital. In addition, dedicated ongoing efforts on the part of civil society present myriad opportunities to protect and enhance local biodiversity in communities across the city; these movements can be partnered with the work of local government and other organizations to fill action gaps and to provide some of the most creative, resilient, and self-sustaining structures of environmental management (Ernstson et al. 2010; Avlonitis 2011; Colding et al. 2013; Ernstson 2013a) (Fig. 24.5). Finally, in all of these efforts, it will be necessary to monitor and evaluate progress, and redirect action accordingly. In this, collaboration across research institutions, individuals, and agencies looking for answers on the ground will help to identify knowledge gaps and generate problem-solving research that direct the city of Cape Town onto a path towards a balance of built environment and rich biodiversity.

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Chapter 25
Climate Change and Urban Biodiversity Vulnerability

William Solecki and Peter J. Marcotullio

Abstract The objective of this chapter is to examine selected connections between ongoing global urbanization, climate change, and urban biodiversity. The direct and indirect interactions between ongoing urbanization processes and climate change have profound impacts on urban biodiversity and its capacity to provide ecosystem services for urban populations. The chapter reviews key aspects of how urbanization affects local and global climate conditions and how these conditions in turn impact urban areas. Special attention is focused on the vulnerability of urban biodiversity to these changes. Urban contexts in developing and developed countries are examined.

25.1 Introduction

Urbanization is a key driver of global environmental change and linked to urban climate and climate change (While and Whitehead 2013; Rosenzweig et al. 2011a; Huang et al. 2008; IEA 2008). Urbanization impacts the atmosphere's regulatory ecosystem services that augment climate variability at the local, regional and global scales. The accompanying climate consequences can lead to increased risk exposure for urban citizens (McGranahan et al. 2007) and vulnerabilities for urban

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biodiversity. Understanding the role of urbanization in climate change and urban climate change risk, and in vulnerability is critical for the production of effective climate change mitigation and adaptation measures (Rosenzweig et al. 2010; Romero-Lankao and Qin 2011; Seto and Satterthwaite 2010; Güneralp and Seto 2008), promotion of sustainable urban habitats and transition to increased urban resilience for sustainability (Solecki 2012; Solecki et al. 2011).

The chapter is divided into three main sections. The first presents a review of the impacts of urbanization on climate at varying spatial scales. The second section reviews specific vulnerabilities related to climate change that are important for urban biodiversity. The third section identifies several important impacts of climate change on urban biodiversity.

25.2 Urban Impacts on Climate

Urbanization is implicated in local, regional and global climate change in a number of ways (Seto and Shepherd 2009). The chapter briefly reviews the impact of urbanization on climate at each of these scales. At the local and regional level the impacts of urbanization on climate can be divided into three broad categories: urban heat island effect (UHI), impacts on precipitation and impacts to air quality. At the global scale the chapter reviews the contribution of cities to climate change through the production of greenhouse gas emissions.

25.2.1 Local and Regional Impacts of UHI

Inhabitants of urban area are subject to climatic conditions that represent a significant modification of the pre-urban climatic state including the well-known urban heat island (UHI) effect (e.g., Chen et al. 2011; Iqbal and Quamar 2011; Kolokotroni et al. 2010). UHI arises from the modification of radiation, energy and momentum exchanges resulting from the built form of the city, together with the emission of heat, moisture, and pollutants from human activities. Urban temperatures are typically 3–4 °C higher than surrounding areas due to UHI (Oke 1997), but can be as high as 11 °C warmer in urban "hot spots" (Aniello et al. 1995; Oke 1982). Dark surfaces such as asphalt roads or rooftops, however, can reach temperatures 30–40 °C higher than surrounding air (Frumkin 2002). The UHI effect is considered a significant urban environmental issue of the twenty-first century (for a review see Rizwan et al. 2008; McKendry 2003; Landsberg 1981).

The UHI effect does not contribute to global warming (Alcoforado and Andrade 2008; Parker 2004; Peterson 2003). Studies indicate that effects of urbanization and land use change on the land-based temperature records are negligible (0.006 °C per decade) as far as hemispheric- and continental-scale averages are concerned (Trenberth et al. 2007). At the same time, as cities increase in size and number, the

UHI effect may play a role in regional climate. One study, for example, presents evidence for a significant impact of urbanization on the regional climate in southeast China (Kaufmann et al. 2007). In this case, the region has experienced rapid urbanization and estimates suggest a mean surface warming temperature of 0.05 °C per decade. The spatial pattern and magnitude of these estimates also are consistent with those of urbanization characterized by changes in the percentage of urban population and in satellite-measured greenness (Zhou et al. 2004). One study which examines the trends of urban heat island effect in East China found clear connection between urbanization and surface warming over the region. Overall, UHI effects contribute 24.2 % to regional average warming trends in this region (Yang et al. 2011). These results are consistent with a recent 50 year study that found most temperature time series in China affected by UHI (Li et al. 2004).

25.2.2 Local and Regional Changes in Precipitation

Urbanization also affects humidity, clouds, storms and precipitation. Numerous studies describe shifts in precipitation amounts in and around cities compared to areas of nearly areas of lower population density (for a review see Souch and Grimmond 2006; Shepherd 2005). The exact mechanisms by which these urban precipitation patterns emerge are poorly understood (Lowry 1998). Unique aspects of urban areas that might affect precipitation levels include high surface roughness that enhances convergence, UHI effects on atmospheric boundary layers and the resulting downstream generation of convective clouds, generation of high levels of aerosols that act as cloud condensation nuclei sources, and urban canopy creation and maintenance processes that affect precipitation systems. No matter what the mechanisms, intensely urban areas and those that are directly downwind of urban areas are cloudier and wetter, with heavier precipitation and more frequent heavy rain events than those that are not, but within the same region (Lei 2011; Changnon 1979). Average increases of 28 % in monthly rainfall rates have been identified within 30–60 km downwind of cities (Shephard et al. 2002). Analysts also have examined whether urban areas are analogous to a warm lake in the winter and therefore enhance snow precipitation (Shepherd and Mote 2011).

25.2.3 Local and Regional Air Pollution

The composition of the atmosphere over urban areas differs from undeveloped nearby areas (Pataki et al. 2006). Most importantly, urban air contains high concentrations of pollutants. Ambient urban air pollution refers to gases, aerosols and particles that harm human well-being and the environment. Cities are the sources of significant air pollution, since they are the location of intense fossil fuel consumption and land use changes. Air pollution has multiple health, infrastructure, ecosystem and climate impacts (Molina and Molina 2004).

Once emitted, the dispersion and dilution of air pollutants are strongly influenced by meteorological conditions, especially by wind direction, wind speed, turbulence and atmospheric stability. Topographical conditions and urban structures like street canyons for example, have an effect on these parameters. Cities that develop in valleys often undergo atmospheric inversions, which trap pollution and enhance effects.

The quality of urban air for at least the past two decades has been defined as a worldwide problem (see Elsom 1996 as an early reference illustrating this issue). In Europe, over the period 1997–2008, 13–62 % of the urban population may have been exposed to concentrations of particulate matter, ozone or nitrogen dioxide above the EU air-quality limits (European Environment Agency 2010). In the United States, over 154 million people, approximately half the national population, suffer from breathing high levels of air pollution (American Lung Association 2011).

While urban air pollution is a ubiquitous problem, trends vary by development status. In countries which were already heavily industrialized in the twentieth century, air pollutants, such as carbon monoxide, sulfur dioxide and total suspended particulates are decreasing dramatically; at the same time nitrogen oxides and non-methane volatile organic compounds have reached a plateau or demonstrate weakly decreasing trends (Holdren and Smith 2000). In many developing world cities, air pollutants have dramatically increased in recent decades. In cities of middle income countries, however, air pollution and the requisite damage therein is greatest (McGranahan and Murray 2003). Recent predictions suggest that under business as usual conditions, urban air pollution in 2050 will increase in the developing world with significant, "disastrous" effects on citizen quality of life (OECD 2012)

Increasing motor vehicle traffic is a major air pollution source (Fenger 2009). Motor vehicles emit carbon monoxide, hydrocarbons, nitrogen oxides and toxic substances including fine particles (and lead in countries still using high lead content fuel). Secondary pollution, such as ozone, is a product of these primary pollutants, which react together in the atmosphere. Given the trends in automobile usage in both developed and developing countries, automobiles are a major source of air pollutants (Walsh 2003). In 2009, for example, Chinese sales in automobiles exceeded those in the USA (Ward's 2010).

Most urban air pollution attention has focused on mega-cities (Gurjar et al. 2008; Butler et al. 2007; Molina and Molina 2004; Gurjar et al. 2004; Mayer 1999). However, it is not the largest cities in the world that have the worst pollution levels. A recent global study that examined air pollution trends in over 8,000 cities suggests that urban nitrogen oxides, non-methane volatile organic compounds, carbon monoxide and sulfur dioxide emissions levels were highest in Asia (Sarzynski 2012). This suggests, as some have argued, that some of the smaller cities of the world are suffering from some of the worst environmental challenges (Hardoy et al. 2001).

Urban air pollution can have metropolitan regional effects. Emissions from cities may play a role in regional climate impacts, as high levels of fine particulate matter can scatter and/or absorb solar radiation (Molina and Molina 2004). The visible manifestation of this regional air pollution is a brownish layer or haze pervading many areas of Asia (UNEP and C4 2002; Ramanathan and Crutzen 2002). Hot spots for these atmospheric brown clouds include South Asia, East Asia, and the much of Southeast Asia. Through the examination of temperature records in urbanized regions of China and India affected by the haze, researchers have demonstrated a significant cooling effect since the 1950s (Kaiser and Qian 2002; Menon et al. 2002). The persistence of the haze has significant implications to regional and global water budget, agriculture and health.

25.2.4 Global Impacts of Urban Greenhouse Gas Emissions

Dense settlements also are responsible for land use change and the concentration of human activities (Seto et al. 2010). Both these factors concentrate and enhance the provision of infrastructure, energy use and socio-economic metabolism, all of which intensify and concentrate the production of greenhouse gas emissions (Grubler et al. 2012). While some argue that the concentration of population in dense settlements lowers greenhouse gas emissions through a decrease in per capita emissions (Dodman 2009; McDonald and Marcotullio 2011), few disagree that as the global urbanization level increases, cities will be increasingly key sources of emissions.

Despite the importance of urbanization and cities to environmental change, the role of cities in climate change is not well understood (Dhakal 2010). For example, a recent review of the literature suggests that cities contribute somewhere between 40 and 85 % of total anthropogenic GHG emissions (Satterthwaite 2008). This wide range is matched by the variation in figures for GHG emissions from individual cities. In this section, the literature on the role of cities as producers of GHGs is reviewed. While infrastructure such as buildings, streets, pipes, tracks and trains, have significant energy embodied in their structure (Ramaswami et al. 2008), global life-cycle estimates of urban infrastructure do not exist. At the same time, however, an increasing numbers of studies have identified urban scale GHG levels (see for example, Marcotullio et al. 2012).

Of great importance to identifying the role of cities and urbanization in climate change is the definition of the city and the definition of what urban activities are included in the GHG protocol. Currently no consensus of "urban" (Marcotullio and Solecki 2013; also see Chap. 1) exists and accounts of urban GHG emissions have used a variety of definitions ranging from including only governmental activities to including activities within a metropolitan region (e.g., Bader and Bleischwitz 2009; Chicago Climate Task Force 2008). With the inclusion of larger areas and greater "scope" (i.e., a more inclusive definition of urban residents' activity) the urban GHG emissions levels change. The example of an increase in urban area's influence on urban GHG emissions is straightforward: as analysts include larger areas, the level of total GHG emissions rises per city. For example, emissions levels within the political boundaries of Chicago are lower than within the entire Chicago metropolitan area (Chicago Climate Task Force 2008). On the other hand, the change in GHG levels per capita with the increase in geographic scale is less obvious. Suburban areas in the developed world have higher emissions per capita than urban areas.

Therefore, including metropolitan areas rather than core urban centers may increase GHG emission per capita levels.

The greater "scope" of emissions includes accounting for the emissions related to more activities, including consumption. The classic argument is that urban residents are responsible for electricity demand from electricity production plants located outside of urban or even metropolitan borders. The GHG emissions associated with this demand are arguably urban, but do not take place within the urban border. Some argue that urban GHG accounts must include these emissions, as they would otherwise be apportioned to rural areas (Kennedy et al. 2011).

Analysts have recently identified a set of standards in urban GHG protocols (Kennedy et al. 2010). Unfortunately, it still remains difficult to compare urban GHG emissions levels due to varying definitions of the city, measurement techniques and scope of analysis (Bader and Bleischwitz 2009). Urban GHG emissions accounts from individual cities vary considerably. For example, estimates of CO_2-eq emissions per capita in London range from 4.4 tonnes (Sovacool and Brown 2010) to 6.2 tonnes (Greater London Authority 2010) to 9.6 tonnes (Kennedy et al. 2011) (a more than 100 % difference). It is therefore important to review all urban GHG data carefully.

In a recent article, analysts have demonstrated the variability of urban GHG emissions (Hoornweg et al. 2011). Specifically, this study argues that average per capita GHG emissions for cities vary from more than 15 tonnes of carbon dioxide equivalent (tCO2e) (Sydney, Calgary, Stuttgart and several major U.S. cities) to less than half a tonne (various cities in Nepal, India and Bangladesh). This variation is due to a number of factors including the size of the city, the population density, the affluence and the urban growth rates (Marcotullio et al. 2012). Some have also identified climate as an important determinant of urban GHG emissions because of associated heating and/or cooling requirements (Kennedy et al. 2011).

Estimates of urban share of global GHG emissions have been presented as varying widely (Satterthwaite 2008). One relatively recent study by the International Energy Agency (IEA 2008) has become widely cited. The IEA (2008) estimates that urban areas currently account for more than 71 % of energy-related global greenhouse gases and this is expected to rise to 76 % by 2030. In another study, using the year 2000 as a baseline, urban GHG emissions range between 38 and 49 % of total emissions, or between 12.8 and 16.9 billion tonnes CO_2-eq. (Table 25.1) (Marcotullio et al. 2013). Both these studies suggest that urban emissions levels vary in different regions. Typically, African urban GHG emission shares are lowest of any region and North American urban GHG emission shares are highest. Overall, urban GHG emissions in developing countries have a much lower share of total emissions than those of the developed world (Fig. 25.1).

Amongst urban GHG emissions, the energy sector accounts for the largest share ranging from 54 to 65 % of total urban GHG emissions. As Hoornweg et al. (2011) state, energy-related emissions is the largest single source of GHG emissions from a production-based perspective and is even large from a consumption-oriented perspective. Agricultural activities typically provide the smallest share of total urban GHG emissions. Transportation accounts for a significant level; one study

Table 25.1 Top 15 highest GHG urban extent emitters in the year 2000

Urban center	Country	Population (thousands)	Total emissions (million tonnes CO_2-eq)
Tokyo	Japan	76,301	644.4
New York	USA	26,562	443.9
Los Angeles	USA	18,320	270.0
Chicago	USA	10,596	213.8
Seoul	South Korea	20,881	172.1
Essen	Germany	10,597	171.6
Taipei	Taiwan	18,229	165.6
Moscow	Russia	14,847	158.2
Shanghai	China	15,155	137.9
San Jose	USA	8,301	119.1
Boston	USA	7,077	117.7
Houston	USA	4,326	122.3
Detroit	USA	4,444	100.2
Baltimore	USA	6,572	97.6
London	UK	12,997	93.0

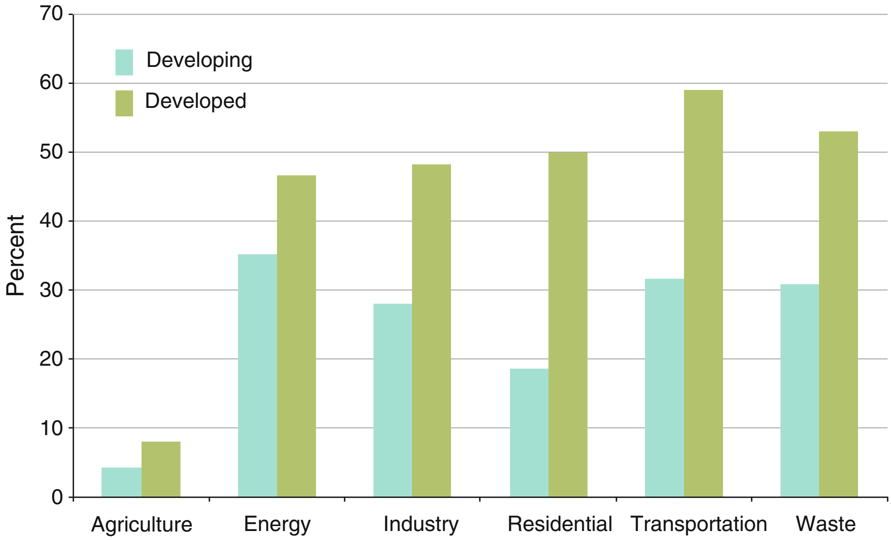

Fig. 25.1 Percentage of total GHG emissions by sector that are attributed to urban areas for developing and for developed countries, during the year 2000. Country classifications of developing and developed countries are based upon UN country categories for 2000. For examples of these categorizations see the statistical annex of the *World Economic Situation and Prospects (WESP) 2012* (UN-DESA – DPAD 2012) (Modified from Marcotullio et al. 2013, submitted)

suggests that transportation is responsible for approximately 20 % of total urban GHG emissions (Marcotullio et al. 2013).

The largest urban GHG emitters tend to be the largest urban areas, but population size is not the only determining factor in emission levels (Table 25.1).

Hoornweg et al. (2011) suggest that the sheer magnitude of some larger world cities ranks them with national emissions levels. For example, Shanghai's population and greenhouse gas emissions would place it in the world's "top 40" if it were a separate country. If all member cities of the C40 group are combined, the resultant emissions levels would be among the top four highest national GHG emissions in the world for each category. In another analysis, the top 15 largest urban GHG emitters together account for approximately 23 % of total urban GHG emissions and 8.6 % of total global GHG emissions.

The pattern of the largest urban per capita emitters follows a different pattern. The 15 largest per capita emitters are typically smaller urban centers (typically with populations under 200,000 with many under 100,000) that are locations for energy conversion and heavy- or chemical-industry, mining, or large scale livestock centers. The aggregate emissions from these 15 centers are much lower than the largest urban areas; approximately 2.6 % of total urban GHG emissions and <1.0 % of total global GHG emissions, but due to low populations they stand out as high per capita contributors (Marcotullio et al. 2013).

25.3 Climate Change Variability in Relation to Urban Areas

Climate change directly impacts urban centers. Climate change impacts include the increased occurrence of extreme weather events such as heavy rainfall, warm spells and heat events, drought, intense storm surges and sea-level rise (see Hunt and Watkiss 2011; Romero-Lankao and Dodman 2011; Rosenzweig et al. 2011b). Climate change is likely to accelerate ecological pressures, as well as interact with existing urban environmental stresses to increase vulnerability (Leichenko 2011; Wilbanks and Kates 2010), particularly those associated with urban biodiversity. For example, New Orleans' geophysical vulnerability is shaped by its low-lying location, accelerating subsidence, rising sea levels, and heightened intensity or frequency of hurricanes due to climate change (Wilbanks and Kates 2010; Ernstson et al. 2010). Alternatively, cities in arid regions already struggle with water shortages. Climate change will likely further reduce water availability because of shifts in precipitation and/or evaporation paired with rising water demand (Gober 2010). Four important climate-related issues in urban areas that increase urban biodiversity vulnerability—including flooding, temperature changes, geo-hydrology, and air pollution—are reviewed in the following sections.

25.3.1 Inland and Coastal Flooding

Heavy rainfall and storms surges could impact urban areas through flooding which in turn could lead to the destruction of properties and public infrastructure, contamination of water sources, water logging, loss of business and livelihood options, and increase in water borne diseases as noted in wide range of studies (Rosenzweig et al. 2010). Extensive studies have attempted to better model the frequency and condition of extreme precipitation events and associated flooding (i.e., Onof and Arnbjerg-Nielsen 2009; Ranger et al. 2011).

Sea-level rise represents one of the primary, if not *the* primary, shift in vulnerability in urban areas that results from climate change, given the accelerating urban growth in coastal locations (Dossou and Glehouenou-Dossou 2007; McGranahan et al. 2007). Rising sea levels and the associated coastal and riverbank erosion or flooding with storm surges could all lead to widespread vulnerability of populations, property, coastal vegetation and ecosystems, and threaten commerce, business, and livelihoods (Carbognin et al. 2010; Hanson et al. 2011; Pavri 2010). Structures built on in-filled soils in the lowlands of, for example, Lagos, Nigeria; Mumbai, India; and Shanghai, China are more exposed to risks of flood hazards than similar structures built on consolidated materials (Adelekan 2010; Revi 2008).

25.3.2 Urban Heat and Cold

In general, climate change will bring increased annual and seasonal temperatures, and declines in mean monthly, annual, and seasonal average temperatures, which will have important implications for ecosystem function in cities. Heat waves and warm spells could exacerbate urban heat island effects, including increased air pollution and heat-related health problems (Hajat et al. 2010), increased salinity of shallow aquifers in drylands due to increased evapotranspiration and the spread of some diseases, including malaria. The probability will increase for long term and spatially extensive heat waves, such as the heat wave that occurred across continental Europe in 2003. Increased warming is predicted in a wide variety of cities including sub-tropical, semi-arid, and temperate sites (Thorsson et al. 2011). Conversely, widespread reduction in cold waves will reduce heating demands (Mideksa and Kallbekken 2010). Increased climate variation resulting occasionally in more intense cold waves (such as those experienced in Ireland in recent years) also could have significant localized impacts.

25.3.3 Geo-Hydrological Hazards

Climate related hazard exposure will vary due to differences in the geomorphologic characteristics of the city. Climate change will increase the risk and vulnerability of urban ecosystems to a range of geohydrological hazards including groundwater and aquifer quality reduction (e.g., Praskievicz and Chang 2009; Taylor and Stefan 2009) and subsidence, and increased salinity intrusion. Subsidence caused by groundwater extraction has led some land in cities like Shanghai to sink by a several meters or more. This is compounded when groundwater is saline (thus eroding structures) or rainfall increases in intensity and duration. While urban areas located in lowlands will have higher risk to flooding, urban centers located in hilly areas will be exposed to landslides.

Drought will lead to food insecurity, increase in fuelwood prices, water shortages, decline in ecosystem function, and an increase in water related diseases (e.g., Farley et al. 2011; Herrfahrdt-Pahle 2010; Vairavamoorthy et al. 2008). Averaging across all climate change scenarios, recent findings suggest that nearly 100 million more city-dwellers "will live under perennial shortage under climate change conditions than under current climate" (McDonald et al. 2011, p. 2).

25.3.4 Air Pollution

Climate change has been linked to a spectrum of air pollution conditions. For example, increased temperatures will promote the increased production of secondary air pollutants such as ozone, NO_x and SO_x. In more remote stretches of metropolitan areas, climate change also will likely increase the frequency of wildfires (Moritz et al. 2012), the probability of which also has been heightened by sprawl in the urban-rural interface. Climate change may also affect the distribution, quantity, and quality of pollen, as well as altering the timing and duration of pollen seasons; the burden of asthma and allergies also could rise as a result of interactions between heavier pollen loads and increased air pollution, or as climate change promotes more frequent wildfires (Shea et al. 2008).

25.4 Key Urban Biodiversity Vulnerabilities

Cities have a surprisingly high level of biodiversity (Chap. 10) and proximity to protected areas and biodiversity hotspots (Chaps. 3 and 22) and this biodiversity improves both human well-being and the quality of life in urban areas (McGranahan et al. 2005). In order to help maintain and even enhance levels of urban biodiversity, it is important to understand the role of cities in climate change, the impacts of climate change on cities, and the vulnerability of urban biodiversity to potential impacts.

Climate change will have profound impacts on a broad spectrum of city functions, infrastructure, and services (Rosenzweig et al. 2011b; UN-Habitat 2011). It will exacerbate the general stresses already placed on urban ecosystems, and will present particular difficulty for ecosystems that exist within marginal or limited ecosystem niches, such as wetlands. These risks and vulnerabilities vary with the temporal and spatial scale and occurrence (i.e., chronic vs. acute) and are expected to increase over the next several decades. Three key aspects related to urban biodiversity are vulnerable to climate change impacts and therefore critical for policy. These include the quality and extent of urban ecosystems habitat, the provision of green infrastructure, and urban wetlands.

25.4.1 Impact on the Quality and Extent of Natural Ecosystems in Urban Areas

Urban systems will be impacted by cascading risks due to climate change (Hunt and Watkiss 2011). Climate stresses, particularly extreme events, will have effects across interconnected systems—within specific sectors and across multiple sectors (Gasper et al. 2011). The cascading effects of climate change can have both direct and indirect economic impacts (Hallegatte et al. 2011; Ranger et al. 2011), and can extend from infrastructure and built environment sectors to natural ecosystems in urban areas (Frumkin et al. 2008; Keim 2008).

Habitat for native plants and animals can hold significant value for urban residents. Wildlife appreciation activities including birding, hiking, and fishing make substantial contributions to the well-being of city dwellers. Many cities have successfully profited from their ecosystem habitats and species through a variety of passive recreation programs for students and others groups. It is important to recognize that it is difficult to isolate climate change signals from the other stressors facing the urban ecosystems. At the same time, there is literature that suggests the natural areas in cities and urban biodiversity will be affected by climate change. For example, shifts in urban system disturbance regimes (e.g., fire, wind, and drought) are mechanisms that can introduce phase changes (e.g., sudden or abrupt changes in habitat condition and quality) and pest species (e.g., invasive species/diseases/parasites) in cities. Invasive species, including both plant and animal species, could become more established with extended drought or other disturbances. Expansion or strengthening of disease pathogens could threaten locally important species such as predators of insect pests, which in turn could increase the number of the specific pest species. Winter warming and absence of cold waves will benefit certain species of insect pests and diseases that are sensitive to prolonged periods of cold; other invasive species may be able to respond more readily to warmer winter- and spring-time temperatures.

Climate shifts will impact the resources available to urban wildlife including insects, birds, and other larger animals by changing the quantity, quality, and timing of forage for animals. It also can adjust the speed of onset of emerging diseases and other pathogens and alien/invasive species entering the extended regions around cities. The shifts in forage and in species composition will result in changes in species competition and pest management regimes. Increased drought conditions will have a significant impact on ecosystem health beyond the relative strength of the drought. This is because reduced stream flows affect aquatic habitats and may cause or exacerbate chemical water quality problems such as eutrophication of already stressed urban ecosystems.

Furthermore, it is clear that some climate shifts will affect ecosystem habitats more directly than others. For example, climate change will likely result in identifiable forest tree species shifts, such as the decline in one species to be replaced by another better suited to the likely warming and moisture-limiting climate. This shift could result in an important loss of forage for one or more animal species, while the forest composition shift could have negligible impacts on a forest's net value for watershed protection and water quality (i.e., surface and sub-surface water supply recharge will still take place). Locally endangered species are particularly susceptible to climate change-related habitat shifts because they are already limited in extent and overall resilience.

25.4.2 Urban Green Infrastructure

A wide variety of ecosystem services and green infrastructure will be impacted by climate change (for a general discussion of urban ecosystem services, see Chap. 11). Climate change will alter ecosystem functions such as temperature and precipitation regimes, evaporation, humidity, soil moisture levels, vegetation growth rates, water tables and aquifer levels, and air quality. These ecosystem functions, in turn, can influence the effectiveness of a range of green infrastructure and climate adaptation strategies. These strategies include permeable surfaces used to promote storm water management, green/white/blue roofs used for urban heat island mitigation, coastal marshes that act as flood protection, food and urban agriculture and overall biomass production, disease vectors (e.g., seasonality and intensity of mosquitoes), and overall air quality (because of increase in secondary air pollutants). In the case of Mombasa, Kenya, for example, the city will likely experience more variable rainfall as a result of climate change; this variability will make initiating and expanding green infrastructure more difficult (Kithiia and Lyth 2011). Street trees in British cities will be increasingly prone to heat stress and to attacks by pests, including non-native pathogens and pests that could survive for the first time under new warmer or wetter conditions (Tubby and Webber 2010).

Some ecosystem health impacts will be intensely local in their extent. Decline and increased stress on urban forest patches represent an example of a significant local impact. These parcels could include a small grove of trees or small forest stand in a park that are quite valuable to densely-settled locations as habitat or amenity resources, even though they represent limited ecological value to the region or country in which the city is located. Loss of tree cover, habitat value, recreational value, and urban heat island mitigation value could have significant acre-by-acre costs to urban communities.

25.4.3 Urban Wetlands

An important global scale, climate-related risk and key vulnerability to urban ecosystem health is the loss of freshwater and coastal wetlands. Fresh water wetlands are especially susceptible to shifts in seasonal water flows and must also compete for water resources during times of stress. A drought can have significant impacts on the hydrology of fresh water wetlands; increased frequency of drought could lead to a phase change or tipping point (i.e., non-linear changes in ecosystem function and properties that could lead to a dramatic and potentially sudden transition in ecosystem health) that could result in a loss and significant degradation of the system.

Sea level rise will cause shifts in flooding potential on the urban coastal wetlands and beach zones, which will alter the habitat quality of these locations at rates significantly above natural baseline conditions. The amount of sea level rise could have potential large-scale impacts on the areal extent and ecosystem health of the urban coastal wetlands, including permanent inundation, accelerated inland wetland migration (if the wetlands are not blocked by bulkheads or similar structures), and shifts in salinity gradients.

The loss and degradation of urban coastal wetland ecosystems likely will be the most significant single economic consequence of climate change on urban ecosystem health. The decline of coastal wetland ecosystems will result in primary and secondary impacts. Water quality decline in coastal wetlands will result in lower productivity among fisheries. Loss of estuarine wetlands will be associated with a decline in the overall function of these areas for absorption of pollutants and nutrient removal from river water.

Loss of coastal and inland wetlands to inundation, increased flooding, and sea level rise risk threaten critical habitats in urban areas (Ehrenfeld 2008). Plants and animals at the margins have the most limited adaptive capacity and could be most negatively impacted. Upland fringes of coastal wetlands could be susceptible to storm surge which are present in the upper reaches of coastal bays and extended estuarine environments. Interior fresh water wetlands could be susceptible to extended droughts associated with groundwater declines.

In New York City and much of the extended urbanized areas of the U.S. Mid-Atlantic Coastal region, remnant coastal wetlands will be lost to sea-level rise because the wetlands will not be able to migrate inland due to bulkheading and intensive coastal development (Rosenzweig et al. 2011b). Recreational sites such as parks and playgrounds also will be affected. In New York, recreational sites are defined as critical infrastructure and often located in low elevation areas subject to storm surge flooding (Rosenzweig and Solecki 2010). Although climate change is likely to have significant impacts on traditional tourist destinations, little existing research has examined the effects upon urban tourism in particular (Gasper et al. 2011).

Finally, as highlighted in the Cities and Biodiversity Outlook Action and Policy (Secretariat of the Convention on Biological Diversity 2012), cities—and urban biodiversity and ecosystem services in particular—can play important roles in mitigating and adapting to climate change. Urban green spaces, ranging from parks to residential lawns and roof gardens, contribute to climate-change adaptation in several ways: (1) trees can contribute to adaptation by providing more shade and cooling, thereby reducing overall energy consumption. The total amount of energy savings depends on many factors, including the species, size, abundance, and location of trees. In most cities around the world, there is abundant opportunity to increase urban vegetation. (2) vegetation and green roofs can significantly reduce both peak flow rates and total runoff volume of rainwater by storing it in plants and substrate and releasing it back to the atmosphere through evapotranspiration. Functional watersheds also play a crucial role in mitigating and adapting to climate change. Watersheds provide access to safe water for drinking and irrigation, which is especially critical given how climate change is disrupting precipitation cycles and historical river flows and groundwater levels. Preserving rather than draining and paving over wetlands can allow for the absorption of excess rainfall and buffer against coastal flooding. As the effects of climate change intensify—putting unprecedented pressure on urban infrastructure such as storm drainage, seawalls, and levees—ecosystem-based adaptation is worth far more than the nominal cost of ecosystem preservation.

25.5 Summary and Conclusions

The role of cities in climate change is not well understood, and more work on this issue is urgently needed in order to generate effective mitigation strategies. Current global estimates of urban contributions to GHG emissions vary from 50 to 71 % of total GHG emissions. What seems evident is that emissions levels vary from city to city, although the largest urban centers are responsible for the more than half of the total global urban emissions. High levels of emissions per capita may not be as important as total emissions in identifying key targets for emission mitigation strategies.

Cities are also sites of climate change impacts and adaptation, and this is where direct impacts on urban biodiversity is crucially important. Research suggests that climate change may have dramatic impacts on urban biodiversity. With less biodiversity within cities, a large and growing proportion of the world's population will be cut off from daily contact with nature and options for adaptation disappear. This dislocation may result in changes in attention and interest towards biodiversity and nature in general. Maintaining urban biodiversity levels may therefore be key to not only urban residents but also global biodiversity in an urbanized future.

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Chapter 26
Feeding Cities: Food Security and Ecosystem Support in an Urbanizing World

Lisa Deutsch, Robert Dyball, and Will Steffen

Abstract In this chapter we take a complex systems approach to exploring the linkages among the phenomenon of urbanization, the changing value systems and world perspectives of urban dwellers, the sometimes distant connections to the food production systems that support cities, and the often invisible ecosystem services that support food production and in turn are affected by food production.

After we explore the relationship between a range of ecosystem services and their relationship to food production, we present three cases of economically developed cities that secure their food from global sources. The wealthy urban populations in all our three case cities adhere to the highly commoditized systems of industrial production based on energy- and material-intensive external inputs for the bulk of their food provision. Fully integrated into the global market, trade enables these cities to both consume and produce what their consumers desire without regard to the local capacity of ecosystems in the regions around the cities. Although

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each city is secure under prevailing economic and trade conditions, they are exposed to a range of socio-economic and ecosystem vulnerabilities that arise from the conventional "productivist" food production paradigm upon which they are based.

We conclude by proposing a number of scenarios describing plausible trajectories for the evolution of food systems in the twenty-first century as humanity becomes increasingly urbanized. Fundamentally, the ecologically integrated system approach, especially the urban garden component, would go a long way towards reconnecting urban dwellers with the biosphere, with potential positive effects on biodiversity.

26.1 Introduction

Humankind is now a predominantly urban species (see Chap. 1), a situation that is unique in the history of our species. We may, for the most part, no longer be agrarian societies in terms of our socio-cultural arrangements, but we remain as utterly dependent on agriculture as earlier humans were. Human ingenuity has made it possible for one to be able to dine in a restaurant that is on the 122nd floor of a skyscraper, some 422 m off the ground, in the middle of the Arabian Desert. However, the potato that one eats would not exist if the ecosystem services of the planet's surface had not been harnessed by an agricultural worker to grow it. Furthermore, agriculture would not continue to thrive if were not for an enormous amount of energy inputs organized by humanity and an even more important range of ecosystem services giving essential support that is often unseen and nearly always under-valued.

The growing trend towards urbanization is exacerbating the need to expand food production to support a growing human population. By 2050 the population is projected to grow by about another two billion people, or by about 20 %. However, food production will need to grow by more than double that, by an estimated 50 %. One primary reason for this disparity is directly related to urbanization. As people move from rural, agrarian lifestyles to urban areas, their incomes and consumption tend to rise. An essential characteristic of that trend is a shift in diet towards more protein, which in turn leads to an increasing demand, beyond the simple population growth rate, for meat and fish (Delgado et al. 2003). The increasing demand for meat, in particular, drives an increase in grain production for livestock feed and, in general, an increased use of resources associated with agricultural production.

The stocks of food available in a city do not determine adequate flows of food to consumers. Spoilage and wastage are both outflows of food that are not consumed at all. The figures vary and are difficult to estimate, but globally around 30 % of food produced is wasted. Adequate food supplies may be nominally available within the city, but not equally distributed. Available food is also not necessarily affordable, and certainly not equally so. Particular kinds of food may be culturally unacceptable or at least fail to meet preferences. Food choices are also susceptible to marketing, promotion and the influence of the buying power and retail strategies of the big food companies. Together these kinds of processes act to constrain what food gets consumed, in what volumes, by whom, and with what health outcomes. However, Earth's ecosystem services have to produce the food whether it is wasted or not, and in that sense production does come before distribution and consequent access and availability; thus, our chapter focuses mostly on production.

At the same time that urban dwellers are exerting an increasing pressure on natural resources both within urban boundaries and particularly from distant support areas (Deutsch and Folke 2005; Deutsch et al. 2007; Folke et al. 1997), urbanites have become increasingly decoupled from nature and have lost connections to very resource base that they are dependent upon for food production (Folke 1998). An example of this is the growing preference of urban dwellers in the world's wealthy countries for conservation of small plots of "pristine" ecosystems over stewardship of the agroecosystems that they are dependent on for their very existence (see case studies in Sect. 26.5 of this chapter).

Ultimately, biodiversity pays the price for the increasing demands on natural resources, and for the increasing disconnect between urban dwellers and the ecosystem services on which they depend. Biodiversity, in terms of the abundance of species, is already being lost at a rate 100–1,000 times the background rate of extinction (Millennium Ecosystem Assessment 2005). It is now estimated that the Earth's biodiversity may be approaching a critical threshold that will lead to a mass extinction event at the planetary level (Barnovsky et al. 2011). Halting the loss of biodiversity is critically important, as biodiversity provides the underpinning for well-functioning ecosystems and thus is necessary for the provision of ecosystem services, thereby supporting human well-being (Millennium Ecosystem Assessment 2005; Díaz et al. 2006; Cardinale et al. 2012).

In this chapter we take a complex systems approach to exploring the linkages among the phenomenon of urbanization, the changing value systems and world perspectives of urban dwellers, the sometimes distant connections to the food production systems that support cities, and the often invisible ecosystem services that support food production and in turn are affected by food production. We focus on the wealthy urban populations of three developed nations to explore the conventional "productivist" food production paradigm upon which they are based. We conclude the chapter by putting forward a number of scenarios describing plausible trajectories for the evolution of food systems in the twenty-first century as humanity becomes increasingly urbanized.

26.2 Impacts of Agriculture on Biodiversity

Human impacts on biodiversity (both in terms of increased rates of extinctions and reductions in abundance and distribution), which result from a quest for food, predate even the advent of agriculture. There is good evidence of human predation as an important factor in the so-called Pleistocene megafauna extinctions during the last ice age, from 30,000 to 60,000 years (Alroy 2001; Martin and Klein 1984; Roberts et al. 2001). These were widespread across the planet, ranging from the disappearance of the woolly mammoths in northern Eurasia to the giant wombats in Australia. However, by far the biggest impact of human activity on biodiversity has come in the last 10,000 years, largely through the indirect effects of agriculture and its expansion across the planet, particularly after the industrial revolution. The most important of these impacts are:

Habitat loss and fragmentation The biggest negative impact on biodiversity is coupled to habitat loss primarily due to the conversion of naturally biodiverse forests, wetlands and grasslands to less diverse agroecosystems of croplands and pastures (Pereira et al. 2010). Humans have already altered more than half of the Earth's surface (Ellis et al. 2010) and croplands and pastures occupy about 40 % of all lands (Foley et al. 2005), compared to 14 % in 1850. In the 30 years following the beginning of the Great Acceleration in 1950, more areas were cultivated than in the 150 years between 1700 and 1850 (Cassman and Wood 2005). With about 33 % of all croplands used for feed crops (Steinfeld et al. 2006), the livestock sector in total occupies more than 30 % of global land area. Today, cultivated systems need to supply cities with food, feed, fiber and fuels. However, not all increases in food production have been met by expansion of areas. 70 % of the growth in crop production in developing countries since the 1960s is due to intensification of agricultural management practices (Bruinsma 2003). These greater yields were achieved by use of irrigation, mechanization, inorganic fertilizers and new crop varieties (i.e., the Green Revolution).

Modification of the water cycle Agriculture modifies the water cycle in two ways – directly through the diversion of liquid water ("blue water") from rivers and underground aquifers, and indirectly via the conversion of forests to croplands and pastures and thus a change in evapotranspiration from the landscape ("green water flows") (Gordon et al. 2005). The diversion of blue water flows can have direct impacts on the biodiversity of freshwater ecosystems; the shrinkage of the Aral Sea due to river diversion serves as an extreme example. However, the conversion of forests – particularly very biodiversity-rich tropical forests – to agricultural systems arguably has a greater impact on biodiversity.

Application of nutrients The application of nutrients, mainly nitrogen (N) and phosphorous (P), on agricultural landscapes and consequent transport into natural ecosystems (e.g., wetland habitats and inland and coastal waters) has also had major negative impacts on biodiversity in these systems. For example, excess P can lead to the eutrophication of freshwater lakes and rivers (e.g., Schindler 2006) while transport of P and N can lead to anoxic zones in the coastal seas adjacent the mouths of large rivers whose catchments contain extensive croplands, such as the Gulf of Mexico adjacent to the mouth of the Mississippi River (Potter et al. 2010). Excess N applied to landscapes can also affect terrestrial biodiversity by favoring fast-growing generalists that then outcompete rarer species that thrive in nutrient-poor niches in the landscape (Mooney et al. 1999)

Modification of disturbance regimes Two prominent examples of this driver are: (i) changes in quantity, timing and frequency of natural flooding events on major rivers due to large-scale irrigation projects, which have consequences for the biodiversity of freshwater ecosystems (changes to the flooding regimes of the Indus, Nile, and Rio Grande rivers are all instances of vastly-modified flooding regimes mainly driven by agriculture); (ii) changes in fire regimes due to land use change such as forest conversion to agriculture (for example, naturally occurring forest fires associated with El Niño play a role in seed dispersal in Borneo's forests, but land-cover modifications related to oil palm plantations have produced changes in the intensity and extent of fire and it is now a destroyer of seeds) (Curran et al. 2004). Pervasive changes in fire regimes in dryland ecosystems – for example, the intensity, frequency and seasonality of savannah and woodland fires across much of Australia – can also lead to large impacts on biodiversity (Steffen et al. 2009).

Although the focus of this chapter is on the impacts of food production systems on biodiversity, the need to provide food for the growing urban population has other impacts on the environment, across all scales, from the local to the global level. Now that we are in the Anthropocene epoch, where human activities rival or exceed natural biogeophysical processes, we need to explicitly deal with global-level environmental challenges to resource use (Steffen et al. 2011). The planetary boundaries concept quantifies biophysical thresholds that cannot be transgressed if we wish to avoid undesirable environmental change (Rockström et al. 2009). As shown in Table 26.1, food production systems have a close and complex relationship to the planetary boundaries, including of course the boundary for biodiversity loss. They contribute to human pressure on all of the planetary boundaries, but equally are at risk themselves if many of the boundaries are transgressed.

The nine boundaries are not independent, but rather have many interconnections across clusters of them. For example, land-use change, the N and P cycles, freshwater use and terrestrial biodiversity loss are all closely interrelated around the extraction of resources and other ecosystem services, but food production is the dominant driver in each case. This strongly supports – even at the global scale – the call for a new approach to agriculture that can increase production to meet the demands of 2050 while at the same time greatly reducing the imprint on all of these planetary boundaries.

26.3 Food Systems in the Context of Ecosystem Services

Ecosystem services (ES) are both the benefits people obtain from ecosystems (Millennium Ecosystem Assessment 2005) and the capacity of natural processes and components to provide the benefits (Daily 1997). These include provisioning, regulating and cultural services that directly affect people and the supporting services needed to maintain other services. All four types of ecosystem services are associated with feeding cities. We explore the relationship between a range of ecosystem services and their relationship to food production (see Table 26.2). For a broader discussion of urban ecosystem services, see Chap. 11.

Table 26.1 Food production and planetary boundaries

Planetary boundary	Relation to food provision
1. Land-use change	Conversion of natural ecosystems to croplands and pastures is the most dominant form of land-use change in terms of the area converted. The area of "domesticated land" (crops and pastures) has increased from a low level prior to the industrial revolution to about 40 % of the ice-free land surface now
2. Phosphorous & nitrogen cycle	By far the largest perturbation of these two element cycles has been their mining (P) and their fixation from the atmosphere (N) for the production of fertilizers. In some regions, the application of manure is also a significant perturbation to the P and N cycles
3. Freshwater use	About 70 % of all freshwater diverted for human use is applied in the form of irrigation to enhance food provision
4. Rate of Biodiversity loss	The loss of habitat through the conversion of natural ecosystems to agricultural systems has been the largest driver of biodiversity loss up to the present. The conversion of mangrove forests to prawn farms plays a significant role in the loss of marine biodiversity
5. Climate change	The use of fossil fuels, especially petroleum products, is ubiquitous through the entire food system, from the tillage of soils through the processing of food to its delivery to shops and supermarkets. In addition to carbon dioxide, agricultural systems emit significant amounts of methane (livestock production and rice paddies) and nitrous oxide (fertilizer use). Destabilization of the Holocene climate has potentially very large implications for our ability to feed nine billion people
6. Ocean acidification	Agriculture affects ocean acidification through carbon dioxide emissions, although energy production is a much bigger source. Marine food systems, especially those associated with coral reefs, are affected by the increasing acidity of the ocean
7. Ozone depletion	The CFCs that are the cause of stratospheric ozone depletion were primarily used in refrigeration, a major driver for which is food storage and transport
8. Atmospheric aerosol loading	Food provision affects the production of aerosols in a number of ways. These include the burning of wood for food preparation (e.g., in south Asia), soil degradation and subsequent wind erosion resulting from overgrazing, and fires and subsequent smoke production associated with deforestation and conversion to agriculture in the tropics
9. Chemical pollution	Fertilizer, pesticides and herbicides, all directly related to intensification of agriculture, are amongst the most pervasive and toxic chemical pollutants

26.3.1 Provisioning Services Related to Feeding Cities

Provisioning services generate the products that humans use directly from ecosystems. We explore how the production of food (including crops, livestock products, fisheries and aquaculture) affects three other provisioning ES below (complementing the ES described in Chap. 11)

Table 26.2 Ecosystem services (ES) and their relationship to food production

Ecosystem service	Relation to food provision
Provisioning Services	
Fresh water	Globally, 70 % of withdrawals goes to food production. In big food-producing regions it is higher, e.g., over 90 % in the Murray Darling in Australia
Wood and fiber	In developing countries the majority of wood consumption is related to food, e.g., household cooking, commercial processing of fish and meat, etc.
Fuel	As ecosystems must provide fuels in addition to food, feed and fiber, there is a trade off between crops for biofuels and crops for food and feed
Regulating Services	
Carbon sequestration	Conversion of natural forests to croplands and grasslands for food and meat production releases carbon and crops do not sequester as much carbon as forests
Climate regulation	Deforestation for pastures is linked to rainfall decline in the Amazon through alteration of the regional moisture feedback cycle
Flood regulation	Removal of mangroves for shrimp farms has degraded natural coastal protection
Disease control	Industrial agricultural practices such as monoculture crop planting and enclosing large groups of animals in close proximity to each other and humans is conducive to pest and disease outbreaks and spreads
Water purification	Phosphorous and nitrogen are polluting aquatic ecosystems due to different activities, e.g., agricultural fertilizer runoff (Baltic Sea), livestock waste mismanagement (SE Asia)
Cultural Services	
Aesthetic	Cultural landscapes are highly valued in Europe, e.g., some Scandinavian inhabitants prefer open agricultural landscapes to native vegetation
Spiritual (cultural)	The Japanese place high cultural values on traditional food-producing *satoyama* landscapes and associated communities
Educational	Consumers can be “reconnected” to agroecosystems through urban food production and farmer’s markets
Recreational	Tourist stays by urban populations on rural farms are of significant economic value and keep agricultural communities in many developed countries viable
Supporting Services	
Nutrient cycling	Industrial systems have broken the nutrient cycle of integrated systems of animal and crops and are now dependent on purchase of chemical fertilizers for production
Soil formation	Ecosystem-oriented farming methods focus on enhancing the capacity of agricultural systems for soil maintenance as the essential prerequisite for food production
Primary Production	The biological basis for agriculture and fisheries productivity is harvested net primary productivity

Biodiversity is not considered to be an ES, but underlies ecosystem functioning and therefore production of all ES

Freshwater supply: Agricultural production can increase or decrease water supply. For example, there is a decrease in water use and an increase in stream flow associated with deforestation and subsequent conversion to cropland, but there is an increase in water use if crops replace grasslands. Agriculture can also effect where water is available, e.g., increases in animal herd density typical of industrialization of livestock production can compact soils and decrease rainwater infiltration (and resulting groundwater recharge). Changes in groundwater can be linked to drinking water availability for humans. Changes in runoff production can mean a reduction of stream flow, which can result in habitat destruction in aquatic systems. Approximately 40 % of our global food supply is dependent on irrigation, and this food production uses 70 % of global water withdrawals (Bruinsma 2003; Postel et al. 1996). In big food-producing regions the local proportion is higher, e.g., over 90 % of withdrawals in the Murray Darling in Australia go to food irrigation (Smith 1998).

Wood and fiber: The majority of wood consumption in developing countries is related to fuel for food, e.g., household cooking and commercial processing. Wood is also used for housing materials, utensils, containers and much more related to storage and consumption of food. Fibers such as cotton and flax are key raw materials for manufacture of items such as textiles, cords, ropes, and baskets which are used to store and transport food to cities.

Fuel: Recent interest in biofuels has increased demand for oil crops such as soybeans and corn as evidenced by the diversion of corn to ethanol production in the USA. These are not only food crops, but also major feed inputs for farmed pigs, chickens and salmon. Price increases in oil crops can not only effect food crop availability, but also encourage increased use of fishmeal as an alternative feed input (Deutsch et al. 2011; FAO 2009).

26.3.2 Regulating Services Related to Feeding Cities

Regulating services are the benefits obtained by humans from the regulation of ecosystem processes. Following, we describe five key regulating services and how they are directly affected by food production.

Carbon sequestration: The regulation of atmospheric carbon is fundamentally effected by agriculture (Lal 2008). The vast 770 Gt stock of atmospheric carbon is constantly being regulated by the biosphere as photosynthesis fixes carbon dioxide into carbohydrates. These stocks are then sequestered in the bodies of plants, and the animals that eat them, thus forming a second stock of 600 Gt. Through various processes, the biosphere exchanges carbon with the soil, which at 2,300 Gt is the largest stock of all. By manipulating the co-evolved terrestrial ecosystems of the planet, farming dramatically affects these stocks. Deforestation and the disturbance of the soil through tillage releases vast amounts of carbon into the atmosphere. Crops replenish carbon in the soil at a much lower rate and for lower residency periods than forests (Cederberg et al. 2011; Rockström et al. 2009). However, human focus on food production results in a trade-off between biomass for food and carbon sequestration. On one hand, although the GPP (gross primary production) of a forest is much higher than grassland, its NPP (net primary production) is low, and the NPP of food edible by humans in a forest is lower still. On the other hand, however, clearing woodlands to create a field of wheat massively reduces GPP but it massively increases the harvestable edible NPP.

Climate regulation: Deforestation driven by agricultural expansion can not only cause changes to local microclimates, but can also be tied to changes at regional scales. Extensive deforestation in the Amazon has greatly reduced transpiration and broken the regional moisture feedback cycle from the land, leaving only vapor flow from the ocean to contribute to moisture generation, which has reduced regional precipitation levels (Oyama and Nobre 2003).

Flood regulation: When natural mangrove swamps are deforested to produce food such as jumbo shrimp in aquaculture ponds, this ecosystem structure is removed, and with it the mitigating function protecting coastal areas from natural storm surges and floods disappears (Rönnback 1999).

Disease control: Industrial food production has simplified ecosystems and uses strategies that are in conflict with the natural mechanisms of disease control, e.g., in regards to diversity and population density. The monocultures of the Green Revolution have seen the spread of a few particular species of crops and animals, and the loss of native varieties (Millennium Ecosystem Assessment 2005). Herd density of industrial husbandry leads to disease spread and massively increases the use of antibiotics to maintain production, which results in drug resistance. Urbanization is mainly related to emergence of new diseases as the demand for more meat in cities means the increase of livestock as well as humans in cities (Perry et al. 2011); it is particularly concerning because growth in meat production in cities is mostly taking place where the capacity to invest in proper facilities is lower.

Water purification: More than 80 % of the P and N used globally in agriculture is not taken up by vegetation; instead, it leaks out and effects other systems, i.e., terrestrial and aquatic ecosystems (Cordell et al. 2011; Galloway et al. 2010). There are even links to water quality related to overfishing of oysters in Chesapeake Bay, USA, where these filter feeders maintained water clarity (Deutsch et al. 2011).

26.3.3 Cultural Services Related to Feeding Cities

Cultural services are the non-material benefits people obtain from ecosystems. In the context of food production, cultural ecosystem services are associated with socio-economic values (e.g. prizing the rural agrarian lifestyle or the production of culturally preferred foodstuffs) as well as with educational and aesthetic values as described below.

Aesthetic: In Sweden, the heavily managed landscape in agricultural production not only has high levels of biodiversity, but the open landscape has a high cultural value compared to the forests that inevitably grow back when agriculture is removed (Björklund et al. 1999; Pykälä 2000). The open coastal views along the Swedish Island of Gotland are some of the most valued in the country, and are maintained by grazing sheep.

Spiritual: The Japanese place strong values on traditional food producing *satoyama* landscapes and associated communities (Ichikawa et al. 2006). In addition, there are strong cultural attachments to the flavor and appearance of Japanese-grown rice as well as to the idea of having traditional Japanese rice farmers farming Japanese landscapes. Many people attach a strong identity to livelihoods, such as ranchers, farmers or fishers.

Educational: There is an educational value related to the reconnection of urban residents to ecosystems through food, as demonstrated through both the growing interest in urban agriculture in Stockholm, Sweden (e.g., allotment gardens) and the increasing number of and popularity of farmer's markets (Milestad et al. 2010).

Recreational: In Europe, Canada, USA and Australia there is an established farm tourism industry whereby urban tourists vacation at rural farms as a way to escape busy cities and allow their children to experience food production first-hand. The importance of this industry to farming incomes and as a tourism resource is increasing (Fennell and Weaver 1997).

26.3.4 Supporting Services Related to Feeding Cities

Supporting services are those necessary for the production of other ecosystem services. They are the underlying capacity of natural processes and components to provide the benefits people obtain from ecosystems, namely food production. We examine how three key supporting services are affected by different production system choices.

Nutrient cycling
Industrial animal production systems are large contributors to nutrient leakages. The current practice of specialized production systems in which crops and livestock are no longer integrated has broken the nutrient cycle; it contributes to excessive concentrations of nutrients in areas close to livestock and deficits in areas with crops. This imbalance results in the need to produce and trade industrial fertilizers globally (Galloway et al. 2010). Thus, manure needs to be treated as a valuable resource (Menzi et al. 2010).

Soil formation
Soil formation is affected by management practices in several ways: (1) livestock systems in which nutrients are not returned to the same geographical location where crops are grown break the nutrient cycle and farmers become dependent on fertilizers,

(2) mining soils by using crop management practices that do not assure that sufficient levels of soil organic material (SOM) or nutrients are generated by crop rotations of N-fixing legumes or SOM from pastures, (3) tillage or residue cover practices can prevent physical erosion by rain or wind (Robertson and Swinton 2005).

Primary production
Gross primary productivity (GPP) of the landscape, is the measure of the rate of conversion of solar energy to biomass. The balance of the energy from GPP left after a plant has satisfied its own metabolic requirements is Net Primary Productivity (NPP) and is available as biomass. Only NPP is available to enter the food chain of all non-producing species, including humans. At its core, farming is the manipulation of ecosystems into states in which the NPP of biomass that is edible for humans or their livestock is highest. In marine systems, intensive aquaculture of carnivorous species such as salmon is characterized by large inputs of high quality wild fish catch in livestock feeds and a net loss of fish NPP (protein) (Naylor et al. 1998, 2000). While progress has been made in decreasing feed conversion ratios, salmon require at least 3 kg of wild fish as feed for each kilogram of farmed salmon eventually produced, and tuna consume 12–20 kg of sardines and mackerel for each harvested kilogram (Tacon and Metian 2009). Increasingly, marine and terrestrial primary production capacity is globally scarce (Erb et al. 2009; FAO 2010; Lambin and Meyfroidt 2011).

26.3.5 Valuing Synergies and Multi-functionality Among Ecosystem Services

Recent development of agricultural production follows the economic model of specialization and focuses on one provisioning service at a time, e.g., meat production. Some main ways that intensification of livestock production can negatively impact biodiversity are through land-use changes, and pesticides and fertilizer misuse. Land-use changes such as continuous cultivation of feed crops like soybeans and conversion of tropical rainforest to grazing lands simplify agricultural systems, which results in major biodiversity losses (Donald 2004). Heavy application of pesticides and fertilizers can result in losses of both plant and animal species as well as in secondary cascading effects on a larger scale. A focus on single products and simplification of landscapes is in opposition to natural multifunctionality and diversity. Agricultural ecosystems, of which livestock are often an integral part, are multifunctional and can generate a whole bundle of ecosystem services simultaneously (see Fig. 26.1). Depending on the production methods chosen, the relative abundance of different ecosystem services can change. For example, livestock not only produce meat and milk, but can be used as a tool for maintaining and increasing biodiversity (e.g., grazing lands can be used to protect wildlife in savannas) (Reid et al. 2010) as well as storing significantly greater stocks of carbon than intensive cropping systems. The concept of ecosystem bundles is a more relevant way to look at food production systems and ecosystems services. The approach is to use

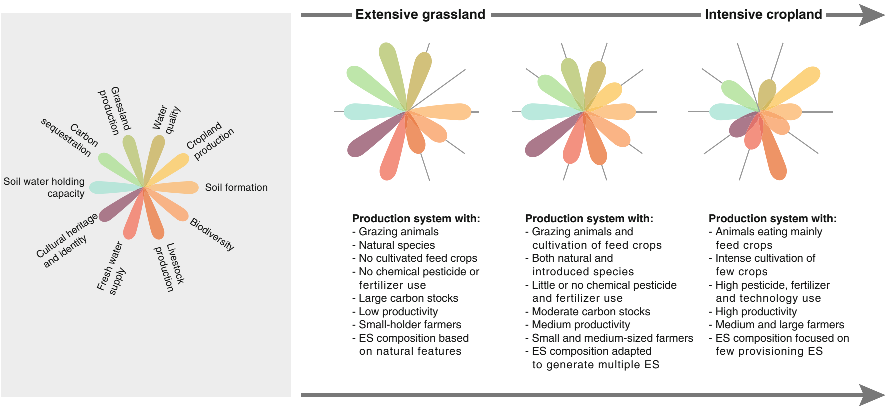


Fig. 26.1 Ecosystem services bundles associated with different levels of intensification of livestock production systems

nature to look for synergies and multifunctionality in food production (in concordance with scenario three in Sect. 26.6.3 of this chapter).

Until recently, of the four ecosystem service types associated with feeding cities, only provisioning services were economically valued. Conventional market economics has begun attempts to place value on the regulating services of agricultural landscapes, for example, through attempts to include measures like biodiversity and natural capital in national budgets (Kumar 2010). Recent debates have discussed the possibility of farmers being rewarded financially for farming in ways that provide higher levels of carbon sequestration (Lal 2010). In some systems, such as livestock grazing on native perennial grasses, much higher soil carbon stocks could indeed be achieved, while still allowing for food production (Robertson and Swinton 2005). However, consideration needs to be given to the food security implications of policy initiatives that take very large land areas out of food production altogether in the interest of promoting carbon regulation.

In addition to the above problems related to simplification and valuation, the percentage of the price paid by the consumer that reaches the farmer is typically very small. Furthermore, the price paid for basic carbohydrates bulk staples is less than for products, such as fruits, vegetables and meats. The more elaborately processed and transformed the product is by the food system, the higher the value. These economic pressures mean that farmers who continue to produce staple commodities have to produce more and more for an equivalent financial return. This 'efficiency' driver leads to increases in on-farm inputs, higher levels of mechanization, and a fewer number of farmers on farms that are larger in area. Estimates are that one third of all food produced is now dependent on fossil-fuel-derived nitrogenous fertilizers (Smil 2002). High levels of such inputs have negative consequences on the landscapes that are expected to yield more, and result in environmental harm through processes such as excessive nutrient runoff. Furthermore, above a certain point, additional inputs produce an ever smaller additional yield.

In China, for example, grain yields are in the order of 50 % more than they were 25 years ago, but it has taken almost 275 % increases in application rates to achieve this (Ingram et al. 2010). There are also negative impacts on rural communities as incomes and jobs are lost and younger generations look to cities for more secure and attractive lifestyles. These biophysical and social-economic forces combine to endanger bulk commodity production.

One way out of the trap of is to get out of bulk staple commodities and into higher value produce. Thai rice farmers can earn considerably more if they convert their rice paddies to aquaculture ponds and farm shrimp in them. The economic rationale is that, with the greater return earned from shrimp farming, they can purchase the rice that they no longer grow. This assumes that the carbohydrate producing activity can be displaced to some other landscape and that the farmer in that location is willing and able to grow it. Brazil is one such landscape. However, at a global scale we cannot indefinitely displace from one place to another the location of the ecosystems services required for the production of primary foodstuffs. Basic carbohydrates are the bedrock of adequate diets and there will be serious implications for urban food security and health if every farmer abandons carbohydrate provisioning in the quest for higher value returns from more exotic and fancy goods.

There are also implications for human health and well-being throughout this process. Displacing the point of production of low-value basic commodities from wealthy to poorer communities results in a range of health and well-being issues in marginal rural areas. Conversely, the process of commodification of food into highly processed consumables promoted through marketing and advertising changes dietary intake. Typically, these processed items are higher in salts, fats and sugars, and overconsumption of them has resulted in a global epidemic of obesity (McMichael 2001).

Finally, the further a product moves along the food chain, the greater its value. Large industrial agri-businesses now secure food provisions from across the globe and elaborately process, package and distribute them as end products. This practice increases the environmental impact of the entire food system. For example, almost half the greenhouse gas emissions associated with the consumption of food in high-income countries like the U.K. and USA are released after the produce leaves the farm (Garnett 2011; Ingram et al. 2010). A consequence of this is that the energy ratio of our food, measured as the total amount of energy required to produce the food against the total amount of energy we get out when we consume it, is now strongly negative (Pelletier 2010; Smil 2011). That is, we put more energy in than we get out.

In the next section, we present three cases of economically-developed, first-world cities that secure their food from global sources. The cases illustrate different approaches to achieving food systems security, which have resulted from changes in the human-environment relationship partly arising from a shift from agrarian to urban societies (Mazoyer and Roudart 2006). Although each city is secure under prevailing economic and trade conditions, they are exposed to a range of socio-economic and ecosystem vulnerabilities that arise from the conventional "productivist" food production paradigm upon which they are based (Lang and Heasman 2004).

26.4 Urbanization, Globalization, and the Changing Relationship Between People and the Rest of Nature

First, we examine five key changes in the human-environment relationship and then illustrate these changes in the capital city regions of Copenhagen, Tokyo, and Canberra; we then explore how this relates to food security and biodiversity.

The first change is that urban dwellers display an increasing lack of understanding of the realities of agricultural production and the social and economic processes that result in their food becoming available to them. Urban dwellers have a "textbook" understanding of ecology and tend to focus on conservation, while rural dwellers have a more practical understanding and focus on managing ecosystems (Hibbard et al. 2007). This can result in political pressure from urban dwellers, who are increasingly more politically powerful than their rural counterparts, to support conservation policy measures that can reduce agricultural output from landscapes. Any consumer, whether local or in distant global markets, who is dependent on those landscapes for their food provision is potentially vulnerable to this change.

Second, as urban populations grow relative to their rural counterparts, they tend to have increased wealth and increased consumption expectations. This typically changes the nature of their diet, as determined by 'Bennett's Law,' which states that as income increases, diets diversify from a narrow range of starch-based staples to a broader range of meat, fruit and vegetables (Cirera and Masset 2010; Timmer et al. 1983). This can have positive health outcomes for consumers, but excessive consumption of meats and highly processed foods can result in negative health outcomes, such as obesity. Furthermore, the higher economic value placed on these food types provides an economic incentive to farmers to produce them, typically at the expense of basic, low value carbohydrate staples. This is economically rational, but someone, somewhere needs to be producing carbohydrate staples in order to maintain food security (Porter et al. 2011).

Third, conditions have shifted from the historical situation of predominantly local production feeding a local population (Evans 1998) to one where food may be sourced from any of the planet's farmlands, rivers and oceans and is transported large distances (often across the globe) to be delivered to consumers in distant cities. Trade has removed a nation's limits on production and consumption, but the ecological limitations and repercussions still remain in the ecosystems of producing countries. Trade plays an ever-increasing role in the provision of biomass such as fish and crops (Erb et al. 2009). In fact, the "landless" livestock (Naylor et al. 2005) and "sea-free" aquaculture industries could not exist without the international commodities market enabling exchange of feed inputs. Although in some circumstances land areas within cities may be able to produce significant volumes of food, particularly in Africa (e.g., 90 % of vegetables in Dar es Salaam (Jacobi et al. 2000)) and Asia (58 % of rice in Hanoi (Anh et al. 2004)), the amount of food that is or could be produced within the urban environment varies widely on the basis both of ecological limits (e.g., land area available and key limits to its productive capacity) and social limits (e.g., the residents' ability and willingness to work it to produce food).

The contribution of urban agriculture to global production is at most one-third (Smit et al. 2001). Thus, in the future, urban demands for food will have to be met by the increasingly globally scarce terrestrial and marine primary production capacity in hinterlands areas outside of cities (FAO 2010; Lambin and Meyfroidt 2011).

Fourth, consumers in urban areas are highly networked into global information, communication and trade networks. This is driving rapid changes in regional cultural values, which are, in general, becoming more homogenized and modeled after Western, high-consumption lifestyles. A consequence of this is food consumption patterns across the globe becoming more Western in their profile. Not only are there health implications (both good and bad) stemming from this diet shift, but there are implications for regions that have historical mechanisms in place that are designed to maintain a degree of local food security. Trade measures to protect regional production of traditional regional cultural staples are vulnerable to this cultural change as those traditional staples decline as a percentage of regional consumption (again, as cultural preferences shift towards increased consumption of Western-style products).

Fifth, urban areas are economic engine rooms and drive free-market systems, which have transformed the food system into highly sophisticated, highly commoditized systems of industrial production. The logic of economic valuation of commodity chains is that producers at the primary production end of the chain receive least value for their product. This tends to either cause them to switch to high value primary products or to value add to what they produce, transforming it into higher value. More broadly, primary producers are driven to increase production by volume for an equivalent income. This increased production is good in the sense that it results in an increased total volume of food available. However, it often comes at an ecological cost as landscapes are driven to produce more from the same area either through large increases in inputs or by eroding natural capital such as soil nutrients (and leading to their exhaustion). Socially, it tends to depress income for agricultural producers, therefore providing an incentive for those who can to switch to other (often urban) employment, and trapping in poverty those who cannot. A shrinking and aging local agricultural workforce is one consequence of this decline in income, and an economically 'colonized' overseas workforce another. In food security terms, it makes urban consumers vulnerable as production is driven away from basic carbohydrate staples upon which they ultimately depend.

Landscapes and seascapes cannot provide food to urban consumers if farmers are not willing to manage the land, sea and coasts for food output. As the following case studies demonstrate, in many places aging populations of farmers are not being replaced by younger generations as the attractiveness of farming as a career declines. Elsewhere, farmers continue to produce, but are switching from basic carbohydrate staples, such as wheat or rice, to higher value products such as wine grapes and farmed shrimp. Furthermore, the economic opportunities of cities are a driving force for migration from rural areas, which also reduces the labor force available to produce food. Although economically rational, these changes to the value placed on ecosystem services can imperil food systems and urban food and nutrition security. We examine how Copenhagen, Tokyo and Canberra are affected by these changes and how this relates to their own food security and biodiversity.

26.5 Three Case Studies of the Changing Human-Environment Relationship as a Result of Urbanization

Urban consumers of food are linked with agricultural landscapes and workers across complex, globalized, food supply networks. The boundary of the urban food system is not the urban municipal limit, but wherever the key variables driving change in the food system occur. This complexity, coupled with the remoteness of these sites of production, can conceal their interconnectedness. However, a systems approach helps reveal how a change in one variable drives changes in the others (Proust et al. 2012). A central focus is on the nature of the feedback loops in the system, and whether they either maintain the value of key variables of concern at roughly constant levels, or rather are driving them exponentially higher or lower. If key variables are being changed, then the vulnerability of urban consumers to that change needs to be considered. Thus, we use a systems approach to understand some of the changing relationships due to urbanization in regards to food security and biodiversity in three major cities.

26.5.1 Danish Food System – Import and Transform

Denmark is a small nation in northern Europe around 43,000 km^2 in size. Over half of the population of 5.5 million is concentrated in and around the national capital, Copenhagen. The landscapes are flat and mostly deep and fertile. The climate is northern temperate, with a cold-temperature limited growing season lasting from April to October. In the absence of human intervention much of the country would be tree-covered, but generations of agricultural labor have worked to keep these forests at bay. With ample and reliable groundwater, the landscape is well suited to agriculture, as wheat yields averaging 7.2 t/ha/year indicate.

Denmark's islands give it an extensive coastline and a history of maritime trade. With few resources other than its natural fertility, agricultural export trade came to dominate Denmark's economy. However, the limitations of available land area meant that Denmark could not be a major raw commodity producer in a global economy. As the innovation of the railroad opened the vast American prairies and Russian steppes for grain production, Denmark's grain exports could not compete in terms of scale. Consequently, from the early nineteenth century, Denmark has had to add commercial value to its primary grain commodities by transforming them into livestock-based products – originally principally dairy and processed meats – to secure export niche markets.

Denmark's agricultural profile today is characteristic of this manufacturing approach. Historically, attempts to maximize agricultural output drove wetland draining and woodland clearing in order to increase total available land area. Forest coverage fell to a low of about 4 % in the 1800s, but it now stands at about 11 %. Land area devoted to agriculture has declined from around 74 % in the 1920s to stabilize at around 60 % today. However, as mechanization increased, the number of

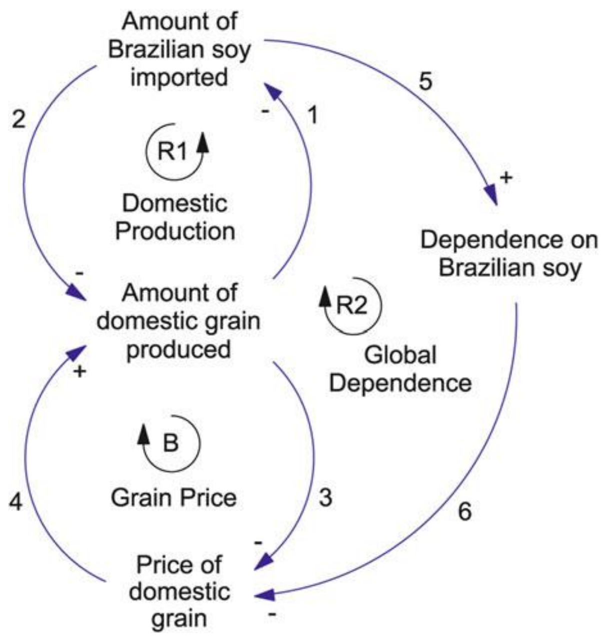
Danes employed in agriculture declined, from a quarter of the workforce in the 1950s to less than three percent today. The number of farms also declined across this time period, from around 200,000 to some 20,000 but increased in size and extent of mechanization (Jespersen 2004). Livestock farming, notably piggeries, holds a dominant position. However, a significant and growing proportion of the grains fed to these animals is not from Denmark, but imported. South American soybeans, much of them Brazilian, form about 60 % of this imported feed mix, with various grains from elsewhere in Europe making up the balance (Deutsch et al. 2009). In all, about 20 % of the land area devoted to growing the feed for Denmark's pig production is located outside of the country. One consequence of this is that Denmark can embark on reforestation and wetland restoration projects without the biological productivity of the land areas withdrawn from agriculture compromising the country's ability to produce pork for domestic consumption and export.

By outsourcing the location of primary production, Denmark retains an economically viable farming sector that produces a culturally valued food staple, while removing land areas from lower-value grain production. The land relieved from agricultural production provides other important services, such as carbon regulation, higher biodiversity and amenity. Value adding in this way is often presented as a model for improving the economies of many developing nations, e.g., the conversion of low value rice paddies into shrimp farms in Thailand. The economic rationale for these substitutions is that the higher-value product generates income with which the forgone lower-valued commodity can be purchased. This requires economically colonizing some remote landscape and harnessing its biological productivity, with all the associated impacts on biodiversity, water availability, nutrient loading, and carbon sequestration capacity displaced to that landscape (Fig. 26.2).

26.5.2 Japanese – Rice Security and Reducing Food Sovereignty

The nation of Japan is located in the East China Sea and is formed from a chain of islands with a total land area of some 378,000 km^2. Its population of 125 million is largely concentrated at very high densities in its cities, with the greater Tokyo-Yokohama region forming a megacity of some 8,550 km^2 at a density of 4,300 per km^2. This crowding is in part a consequence of the geologically young and mountainous nature of much of Japan. It also results in only some 20 % of land area being cultivatable. Climate varies across the islands due to their extensive North-south latitude. However, where they are temperate, and combined with rich volcanic soils and reliable rainfall, they can be extremely biologically productive. Consequently, for much of its history Japan was self-sufficient in a diet based around rice, fish and vegetables.

Post-World War II, the Japanese economy has grown significantly. During this period, the Japanese diet has moved away from its traditional diet to a more 'Western' diet, with greater meat, wheat and oil intake. In 1960, the Japanese



Link Number	Description
Loop R1: Domestic Grain Production	
1	If the amount of Danish domestically produced grains increased, the need for Brazilian grain would decrease assuming the total amount of grain required is constant.
2	If the amount of Brazilian grain imported increased, the amount of Danish produced grain would decrease.
Loop B: Domestic Grain Price	
3	If domestic grain production were to increase it would drive down the price paid for grain as the available supply of commodity closed in on demand for the commodity.
4	If the price of domestic grain went up it would stimulate more farmers to produce it as the economic return on grain production increased. This increased production would then feedback via link 3 to depress prices and choke further production. Production and price would stabilize around equilibrium.
Loop R2: Dependence on Brazilian Soy	
5	As the amount of Brazilian soy imports increase the Danish pork production system becomes more dependent upon them. The price of Danish pork represents the low cost of Brazilian commodity inputs and structural adjustments in the Danish agriculture system combine to make it hard to stop importing from Brazil and to return to Danish grain consumption.
6	Dependence on Brazilian soy as the primary commodity input that is transformed by the Danish pork industry into higher-valued pork products further depresses the price paid for domestic grain.

(continued)

(continued)

Link Number	Description
What is actually happening? Danish farmers are moving out of grain production and into pig production as they can earn more money from pork. It is cheaper for them to buy imported Brazilian soybeans than to produce pig feed themselves. Denmark's agriculture system now transforms globally sourced feed inputs (produced in remote ecosystems) into high-quality, high-value pork products for domestic consumption and export. As a result domestic grain production has fallen, driving imports up and further reducing domestic production. The ready availability of cheap Brazilian feed means that farmers have little incentive to increase domestic grain production. Thus, Denmark is becoming more and more dependent on non-Danish primary commodity inputs and so dependent on the social and environmental conditions in other countries. Denmark can increase biodiversity in its own landscapes, e.g., restoring wetlands, by importing soybeans. Any negative effects on biodiversity then occur in the ecosystems where soybean production occurs.	

Fig. 26.2 Denmark's growing dependence on imported soybeans for pork production. Global and local dynamics interact to effect Danish food security and local and global biodiversity

population of 92.5 m consumed 126 kg of rice per capita. By 2010 the population had climbed to 127.5 m but per capita consumption had fallen by half, to 67.4 kg (Yamashita 2008).

Despite this change in cultural preferences, Japan maintains its sovereign self-sufficiency in rice production due to a complex mix of domestic policy initiatives. These protect Japanese rice farmers through import restrictions, tariffs and subsidies, and ensure national rice consumption can be met by national production, even if food imports are disrupted. Only around 4 % of the rice in Japan is imported in one form or another.

About 85 % of Japanese farmers produce at least some rice. Rice yields per hectare are high by world standards, which is a reflection of the suitability of Japanese landscapes and climate to this crop. However, Japanese rice farmers are not efficient by most measures. Farms are extremely small, averaging around 1.8 ha. Levels of mechanization are high but this is largely to subsidize on-farm labor time and energy. Rice farming is a part-time occupation for most farmers, who earn the majority of their income from more profitable activities outside of agriculture. Government policy instruments designed to maintain sufficient sovereign food production of rice are blamed for artificially high prices that encourage micro-farming and prevent the production efficiencies that could be gained from up-scaling.

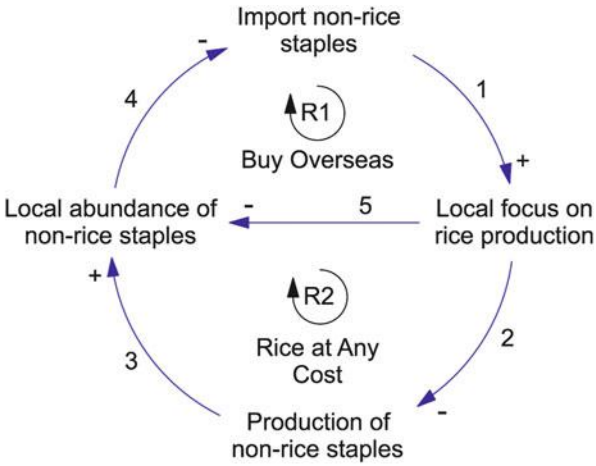
Because Japanese consumption of rice per capita has fallen so dramatically, halving since 1960, domestic production required to secure domestic demand is much less today than it was 50 years ago. Consequently, although Japanese rice production meets Japanese rice consumption, is not meeting changing food preferences, so overall sovereign food security is declining. If Japanese food production tracked Japanese food consumption patterns then it would need to shift to produce the basic commodities required for the 'Western' style eating habits that are coming to dominate. This would require switching to products such as grains to feed livestock, wheat for bread and canola for oils. It is these commodities that are responsible for Japan's import-dependency, which, based on total calorific value, is currently at around 60 % of food consumed. However, many barriers to changing production output exist, including the suitability of Japanese landscapes and climate for these 'exotic' products, the entrenched skill sets of farmers, the low levels of willingness of young people to farm, the exposure of these commodities to cheap world market prices, and the small size of rice farms with their capitalization in rice-production-specific mechanization.

Profitable Japanese farming does exist in non-intensive fields, such as some fruits, vegetables and flowers, but this is not what makes up the carbohydrate staples. Orthodox economic rationalism could argue that there is no problem here and Japan should abandon farming altogether and rely on its non-agricultural sector's earning-capacity to purchase the food it needs from world markets. In ecosystem terms, this is to suggest that Japan should cease trying to harness the provisioning service capacities of its own landscapes and instead appropriate the provisioning services of landscapes elsewhere in the world. Demographic changes and young people's perception that they can earn far better incomes and more enviable lifestyles by working in big cities may simply deliver this outcome anyway. However, it would seem that Japan would then be following a pathway to a future it does not want. In addition to being vulnerable to disruptions to imports, it would lose the cultural ecosystem services that it claims to value, exemplified by traditional '*satoyama*' landscapes, their iconic farming communities and their quality rice output (Takeuchi 2010). It is also highly likely that it will start to experience levels of obesity prevalent in the West as a consequence of increased adoption of the highly processed Western diet (Fig. 26.3).

26.5.3 Australia – Net Food Exporter

The Australian Capital Region (ACR) is in the South East of the continent, and includes the Australian Capital Territory and surrounding regional local government areas. The ACR has a population of 550,000 in a land area of 5.86 million ha (a population density of 0.1 persons/ha). The ACR landscape is dominated by 2.4 million ha grazing lands for sheep and cattle, much of which is on unimproved native perennial grasslands, which are mostly unsuitable for cropping. Croplands cover approximately 187,000 ha, including extensive wheat growing in the northwest of the ACR. Significant forestry activity occurs to the east. Using wheat yields as an indicator of biological productivity, ACR yields are approximately 2.0 t/ha, but this figure is extremely variable depending on highly fluctuating rainfall patterns and other factors inherent to Australian climate and landscape conditions.

Although Australian soils are of low productivity per ha, given the very large land area it commands and its very low population density, the ACR could meet regional demands for the staple foodstuffs: beef, sheep meat, cheese, apples and wheat. As an overall average, and for a diet restricted to these products, the ACR is food sovereign. However, regional consumers would probably not be willing to



Link Number	Description
Loop R2: Rice at Any Cost	
1	The Japanese government's concern about vulnerability to interruptions in the importation of its traditional carbohydrate staple, rice, results in policy initiatives that protect Japanese rice farmers, which increases rice production and dampens rice import demand
2	The more emphasis on domestic rice production, the less willing and able farmers are to economically produce other staples
3	If local production were to occur it would positively affect their local abundance, although this is not in fact occurring
4	The higher the levels of local abundance lower the amount of imports that are required to make up the difference
Loop R1: Buy Overseas	
1	As described above, the Japanese government protects Japanese rice production to ensure Japanese domestic consumption demands are met
5	Focusing on local rice production negatively affects the production of non-rice staples.
4	Because local levels of abundance of non-rice staples are low, imports are high to make up the difference between actual levels and demand

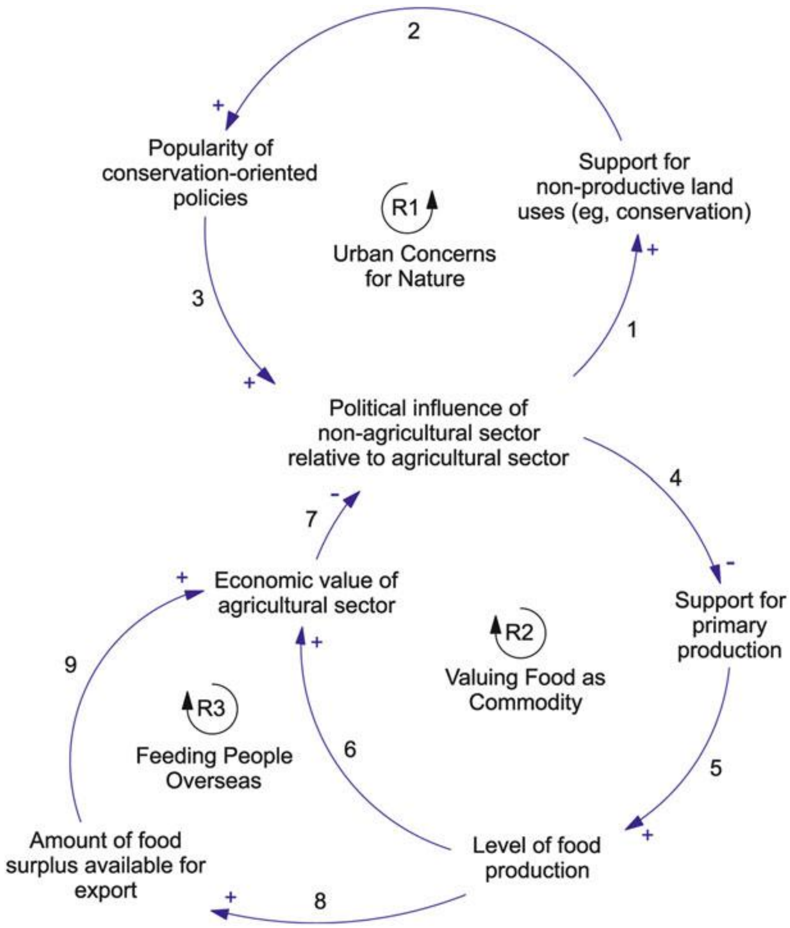
What is actually happening? Japanese government support keeps Japanese rice farmers viable so that Japanese domestic rice production can satisfy Tokyo's rice demand. Over time, Japanese dietary preferences are changing towards a more 'Western' diet and rice consumption (as a percentage of total food consumption) is going down. Government-supported small farms geared to rice production cannot viably track these changes in preferences. With imports making up the difference between local production and local demand this reinforcing loop is driving Japan's food self-sufficiency downwards

Fig. 26.3 Japan's growing dependence on imported non-rice staples due to government supports for domestic rice production limit their intake to these local, seasonally available goods and there may be health implications for such a restricted diet. As a relatively wealthy country the population is under no pressure to restrict its consumption to local production, and so food is traded in and out of the region for reasons of cultural preferences and economic efficiency with little concern for ecological capacity.

The surplus provisioning capacity of the landscape of places like the ACR are exported to make up the productive shortfalls of cities like Tokyo. However, for each hectare of Japanese ecosystem taken out of production, for the various reasons described above, a greater number of hectares of Australian landscape is required for equivalent volume of provision. This is a consequence of Japanese landscapes being at least twice as agriculturally productive, capable of yielding at least six tons of rice per hectare to Australia's two tons of wheat. They are also much less vulnerable to annual climatic variation, such as drought, which is endemic to Australia. If, as discussed above, Japanese policy is to depend on the provisioning capacity of non-sovereign landscapes, then domestic food policy needs to adapt to reflect the vulnerability and variability of the local conditions. This vulnerability includes both local ecological as well as local policy changes that sit largely outside of Japanese political influence.

One example of this is the growing level of environmental concerns, predominantly in the politically-influential and numerically-dominant urban electorates in Canberra. Concern for the cost (in terms of river ecological health) of large volumes of water abstracted for irrigation has seen the growth of political pressure for environmental flow restoration. Despite the merits of such arguments (from an environmental perspective), the consequence is, by and large, that their success means less water is available for irrigation. Rice growers, for example, are particularly susceptible to any reduction in water allocation or increase in its value due to the very high volumes that they require per ton of output and the relatively low value of their product compared to a product such as wine grapes. The observation that consumers actually need rice more than they need wine in order to subsist does not reflect the economic driver pushing in the opposite direction. Consumers dependent on the food produced, including overseas, are consequently vulnerable to this shift in local land-use priorities (Fig. 26.4).

The wealthy urban populations in all three of our cases show a typical highly diversified diet, although the composition in Tokyo is slowly Westernizing. All three cities adhere to the highly commoditized systems of industrial production based on energy- and material-intensive external inputs for the bulk of their food provision. Fully integrated in the global market, trade enables these cities to both consume and produce what its consumers desire without regard to the local capacity of ecosystems in capital city regions. Strong government support policies in Tokyo struggle to maintain local rice production due to cultural values, but Tokyo must import the vast majority of its food due to limited farm areas. Meanwhile, the Copenhagen and Canberra regions could be much more self-sufficient in their food provision. Copenhagen has chosen to focus on large-scale commodity production of pork to supply the world's increasing demand for meat using the industrial productionist system, which imports the majority of feed inputs from other systems through



Link Number	Description
Loop R1: Urban Concerns for Nature	
1	A growing urban population that is increasingly distant from and unaware of its dependencies on ecosystem productive services tends to favor 'conserving nature,' e.g., river health
2	Popular interest in conservation initiatives leads to policy interventions to deliver conservation programs
3	The popular success of conservation programs leads to a political will to enact further such initiatives and wariness to support agricultural-production orientated policies, which are seen as in opposition
Loop R2: Valuing Food as a Commodity	
4	As the political influence of the non-agricultural sector increases, measures supportive of encouraging primary production decrease
5	The less primary production is supported, the less food is produced
6	Decreasing food output decreases the economic value of the sector
7	The less the economic strength of its agricultural sector, the lesser its political influence

(continued)

(continued)

Link Number	Description
Loop R3: Feeding People Overseas	
8	As less food is produced, less surplus food is available for export
9	International income derived from the export of food commodities further decreases the value of the sector and its political influence
What is actually happening? Australia's urban population is far larger than its rural population and this imbalance continues to grow. Wealthy and educated, relative to its rural counterpart, the urban population is largely unaware of its dependency on agricultural output. They support government policies that tend to favor 'nature conservation' over agriculture, perceiving little direct cost to themselves. Like most commodity sectors, agriculture in Australia suffers from worsening terms of trade. The resultant economic decline in the sector weakens its political influence. Consequently there is a political drift away from support for agriculture. Declining agricultural productivity erodes Australia's food exporting capacity The willingness of recipient nations to pay high prices for these commodities has the potential to increase the economic value of Australian agriculture sector. However, currently the dependency of these recipient nations on Australian output is either not readily recognized (in the case of wealthy nations, like Japan, who can afford to import from elsewhere) or is recognized, but beyond the receiver's ability to do anything about it (in the case of poor nations, like Bangladesh, who cannot afford to pay more)	

Fig. 26.4 Urbanization in Australia is weakening the political influence of the agricultural sector. Conservation efforts to maintain riparian areas can benefit biodiversity, but reduce Australia's food exporting capacity

the global market. This approach can increase biodiversity in its capital city region if it releases areas previously under cultivation for restoration, but it presently comes at the expense of biodiversity in the countries supplying the feed. We see a similar disconnect between urban populations and food production in Canberra as urban residents push politically for reductions in irrigation to restore riverine habitats, benefitting riparian biodiversity, but presently at the expense of food production and even local rural livelihoods and food security.

26.6 Urbanization, Food Systems, Ecosystem Services and Biodiversity in the Twenty-First Century: Three Possible Futures

The linkages between cities and the production, processing, transport and access systems that provide them with their food are obviously multi-scale, complex and continually evolving. Predicting how the continuing trend of urbanization and its connection to food systems will evolve in the twenty-first century is a daunting task. But understanding these possible future trajectories is crucial to understand how urbanization will continue to affect the ecosystem services on which we all depend, and the future of biodiversity, which underpins the provision of all ecosystem services. Here we use a scenarios approach, based on the work of Lang and Heasman (2004), to explore three plausible futures for the urban food system, and the implications for biodiversity.

26.6.1 Scenario 1: Industrialized Productionist System

This scenario is basically a higher-tech version of business-as-usual, with a continuation of the food production systems that have developed and come to dominate in most developed nations through the second half of the twentieth century. It is technology- and energy-input dependent and is enabled by a range of revolutions in land use, land ownership and agrarian social relations.

The industrialized productionist paradigm has been hugely successful when assessed by the key indicators that it values. The overwhelming variable of central concern to this paradigm is total volume of food output driven by both intensification and extensification. Consequently, the key food sector that it addresses is the global commodity markets where demand is met through high-input agriculture that channels mass production into mass markets. The efficiency of the industrialized agri-businesses that have come to dominate is measured largely in terms of quantity of produce, with limited choice, variation and quality, other than marginal marketing-dependent brand-based perceptions of product range at point of sale. The knowledge inputs into this food system are narrowly focused around direct application of chemical, pharmaceutical and genetic interventions to raise yields and minimize losses, with the agroeconomic extension officer the primary authority. Often, the productionist paradigm has first looked to secure national markets through local subsidies and market protection, although tensions emerge between larger scale concerns attracted to international markets and more local concerns stressing national security.

Overwhelmingly, the consumer focus is on cheapness, choice and convenience of supply, with the prime purchaser of household food assumed to be a time-stressed female. As post-war food shortages fade from memory, the consumer's gratefulness for adequate supply becomes expectation that all ingredients for any world-cuisine recipe will be constantly available. In many cases these exotic dishes, or their primary inputs, are prepared in tinned, frozen, or even fresh, ready-made forms. The assumption regarding the ecosystem services required to support these levels of provision is that they are cheaply and – increasingly – globally available. The underlying natural resources to sustain large-scale, homogenized, bulk commodity output are assumed to be either inexhaustible or indefinitely relocatable. Key fossil fuel energy inputs for transporting and processing are likewise assumed to be cheap and inexhaustible. Food wastage and pollution along all steps in the food system is not seen as being of pressing concern and may be an insignificant cost relative to the cheapness of the primary input. The health consequences of consuming food are very narrowly considered, the main argument being that food's health dimension is primarily concerned with freeing the world's population from starvation. Broader issues, such as obesity, are regarded as a consequence of consumers' free choice.

Overwhelmingly, food systems early in the industrialized productionist mode are the sole concern of agricultural ministries. Over time, the political support of domestic agricultural departments is eroded as foreign affairs and trade departments seek to open up markets globally.

This model will continue and intensify the current relationship between urban dwellers and food production. That is, urban dwellers are physically and conceptually separated from the places and processes involved in producing the food that they consume. This disconnect means that most urban dwellers do not have an understanding of the ecosystem services on which their food supplies are based, nor on the impacts of their food provision systems for biodiversity. Thus, the separation of urban dwellers from the rest of nature in general not only continues, but becomes even more pronounced.

Although it has tremendous momentum, there are signs that the global dominance of this industrialized productionist paradigm may be coming to an end. The mantra that more food will end global food shortages has not been born out, as large numbers of people regularly go without adequate supplies despite the vast output. In some cases cheap food imports can erode local self-sufficiency, rendering communities aid-dependent. Elsewhere, the abundance of produce fails to reach end consumers in adequate volumes, being lost or spoilt en route or simply because they do not have the means to acquire it. Concerns over the globalized food system's vulnerability to rising energy costs, water shortages, fertilizer input ceilings, and land use and other planetary boundaries all belie the paradigm's conviction that more can always be produced. The further perturbation of climate variation and its effect on productive output of landscapes across the global is an additional risk of largely unknown seriousness (for more on climate change and urban vulnerability, see Chap. 25).

26.6.2 Scenario 2: Life Sciences Integrated System

This approach, which Lang and Heasman (2004) postulate as one of two possible alternate pathways, emphasizes the combination of biotechnology and information/communication technologies to revolutionize the current system. Here, science in the hands of globally integrated food corporations comes to play a dominant role. Biotechnology and computer logistics combine to increase yields and optimize input regimes that are tailored to local conditions, through computer monitored water management and fertilizer regimes adjusted to local soil nutrient profiles – in short, precision farming. Total distribution systems would track produce across the entire food systems from production to retail and consumption. More attention would be paid to losses and wastage through controlled environments and "just-in-time" delivery systems. The old industrialized productionist approach of flooding markets with large volumes of inputs in the hope some would be consumed would be replaced by hi-tech control over the right produce being in the right place at the right time to meet market requirements.

GMOs are the archetype product of the life-science paradigm as scientists try to engineer plants to yield ever more of what humans value under ever more stressful growing conditions. In such a future paradigm, the health needs of consumers could be met by highly personalized provisioning requirements in an information-rich product environment. In terms of ecosystem services, this paradigm would point to lower but more effective input regimes and perhaps the ability to take stressed ecosystems out of production. Overall though, the natural capacity of ecosystems to yield services only partly constrains what human ingenuity can do with those systems. In this paradigm the challenges of producing sufficient food within planetary boundaries would be considered to be yet another laboratory challenge.

The life sciences integrated paradigm can be viewed as the least challenging transition away from the current industrialized productionist paradigm. Its promise is that human ingenuity can continue to overcome the limits that nature temporarily imposes on human behavior. It seems highly likely that some aspects of the paradigm will play a role in future urban food security and indeed many features are recognizable already. However, those voicing concerns with this future argue that it shares features of early versions of the Green Revolution, including the premise that science can indefinitely postpone the time when humankind must live within planetary boundaries. The feared result is a positive feedback loop in which more people become more dependent on the mechanisms that allow the limits to growth to be ignored. As for the industrialized productionist systems, the mechanisms that are holding the food system up are energy- and material-intensive external inputs to the system, so that the system, although based on cutting edge technologies, is not self-supporting or sustainable in the long term. Furthermore, the owners of the supporting mechanism are an ever fewer number of multi-national, vertically integrated agri-businesses who have no particular allegiance to a nation or its population, other than that they are markets for its products. Rather than laud GMOs as the potential savior of the world's food production system, these critics would question the wisdom of copyrighting and privatizing ownership of the genetic information of food.

The life sciences integrated approach could indeed take considerable pressure off the natural environment and possibly enhance other ecosystem services if it was implemented in a way that placed value also on ecosystem services other than food provision. However, it would not change the relationship between urban dwellers and food production, that is, the strong and growing disconnect between the urban population and the ecosystem services of the hinterland. It could even be argued that the life sciences integrated paradigm would exacerbate this disconnect, given its strong emphasis on a high-tech, highly commoditized system that diminishes even further the role of nature and biodiversity in supporting sustainable food systems.

26.6.3 Scenario 3: Ecologically Integrated System

This approach is vastly different from both the industrialized productionist and the life sciences integrated paradigms. The emphasis here is on maintaining the whole suite of ecosystem services rather than maximizing food production at the expense of other services. This implies a focus on production diversity such as polycultures, as well as urban agriculture or urban gardening as an important component of the scenario. The ecologically integrated systems paradigm is the scenario most likely to accept and work within the planetary boundaries.

Characteristics of this approach are the focus on key processes that drive balancing feedback loops in the system. A priority concern is to maintain the fundamental ecosystem functions and characteristics, such as biodiversity, which ensure that ecosystem provisioning and other services can continue to be delivered. Reduction in the use of energy and other inputs as well as waste reduction are key features of the approach, and overall risk management, for example, for insect pests, would be achieved through production diversity, such as polycultures. In effect, natural ecosystem services would be used rather than industrially produced synthetic inputs such as pesticides. The ecologically integrated system scenario is fundamentally based on an integration of the entire food system, with a central focus on whole-farm systems approach that manages land primarily for soil health and water efficiency, so that biodiversity is increased and long term yields are supported.

As an industry, the approach is most closely associated with today's organic farming sector, but other low input and 'nature-focused' farming systems fit the mold. In many areas at the margins of the currently dominant industrial systems, these alternatives are being practiced and refined. The scientific knowledge informing the development of these systems cuts across disciplines and would include lay, farmer and other knowledge sources. The role of formal policy institutions is often regarded with suspicion, although legal mechanisms to regulate, for example, environmental claims (such as organic labeling) are recognized. Typically the emphasis is on developing policy partnerships of collaborative institutional structures, both formal and informal, which include local civil society and social groups.

Within the ecologically integrated system paradigm the consumer is reimagined as an active agent within the food system, whose knowledge and concern recouples them and their consumption to the landscapes and farmers who feed them. As a consequence of this, regional products and local markets are favored, which goes some way to reflecting local and seasonal availability of produce. The consumption of a wider diversity of minimally refined and processed basic produce, with less meat, fats and sugars, is rightly assumed to be healthier than the commoditized food products of the other two paradigms. The overarching environmental assumption is that resources are finite and environmental pathways to replenish them are not finite but rather rate-limited. Hence there is a focus on limiting rates of abstraction of resources to balance rates of replenishment through a land-capacity-first focus. The political support for the ecologically integrated paradigm is weak but growing, most notably among affluent and ethically concerned consumers in first world urban situations.

A tension exists between the desire to have this approach move from the margins, where it is currently developing, to the mainstream. At the margins, the approach is not unimportant but reasonably ineffectual, although it is developing the basis for change at much larger scales. In the mainstream, the ecologically integrated system approach could actually contribute significantly to supply, but there is a risk of falling under the productivist emphasis of the other paradigms (and consequently suffering the loss of its benefit to the food system as a whole).

26.6.4 Conclusions

Arguments to limiting regional consumption to the bioregional capacity need to be considered in light of the fact that about half of the world currently does limit its food intake to bioregional output and starves regularly as a result. First-world advocates of such practices also need to consider how much of their total consumption they are prepared to have constrained by local production, or whether they actually expect the productivist regime to continue as a back up for whatever or whenever supplies become locally unavailable. It is possible that aspects of the ecologically integrated paradigm will form a part of future food system security. However, it remains to be seen how much of a balance can be achieved between local resource limits and the benefits of consuming food from remote ecosystems, especially how those benefits are more equitably transmitted back to support the farmers and the landscapes that produced the food.

In stark contrast to the first two scenarios, the ecologically integrated system approach has the potential to be a game-changer in terms of the relationship of urban dwellers to food production. The emphasis in this scenario on urban gardens would go a long way towards addressing the current disconnect between urban centers and their food. In addition, if the amount of food grown in urban areas and their peri-urban surrounds could be increased from present estimates of about 15 % of food consumption to perhaps a maximum of 30 %, it would also make a significant contribution to taking pressure off landscapes to increase productivity.

Cities in poor regions that cannot afford to displace their point of impact to another landscape (once they have exceeded their local landscape's capacity to provide) suffer chronic food insecurity and shortages. Their predicament is made worse if affluent consumers out-bid them in what little food markets to which they have access. Consequently, these urban consumers of food need to ensure that the landscapes that are provisioning them are being managed sustainably.

However, in affluent cities the primary food security issue is not one of inadequate supplies leading to general malnutrition and starvation. Consequently, in these cities the value of urban food production is more likely to be found in its educative, active lifestyle and community-building roles than its ability to contribute significant percentages of total volumes consumed. Here the poor health outcomes are likely those arising from overconsumption or the consumption of a nutritionally poor diet, with issues such as obesity, type II diabetes, blood pressure and other cardiovascular conditions dominating. Fundamentally, the ecologically integrated system approach, especially the urban garden component, would go a long way towards reconnecting urban dwellers with the biosphere (Folke et al. 2011) generating positive effects on biodiversity.

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Chapter 27
Urban Governance of Biodiversity and Ecosystem Services

Cathy Wilkinson, Marte Sendstad, Susan Parnell, and Maria Schewenius

Abstract In an increasingly urban world the battle for biodiversity hinges on how effectively cities are governed, and how responsive those who run cities are to transforming the urban system to embrace ecosystem integrity and restoration. This chapter sets out the nascent field of urban biodiversity governance, and is the first scientific publication to provide a synthesis review of the urban biodiversity and ecosystem services governance literature. It notes the recent expansion of an interdisciplinary global urban biodiversity and ecosystem services governance agenda, and that a significant body of academic material already has emerged. The chapter focuses on the challenges and opportunities of governing urban biodiversity and ecosystem services at the local, national, regional and global scales. It reveals that although overarching patterns of lack of political will, institutional capacity and knowledge are challenges to making an impact on ecological integrity, there are numerous sites of innovation, and solutions that have been put in practice. While the chapter finds patterns of challenges and opportunities experienced across cities covered in the literature, it is cautious about generalizations, as studies from Africa, South America and parts of Asia are largely lacking. Finally, the chapter considers what is required to improve governance of urban biodiversity and ecosystem services, and sets out a more inclusive research agenda to inform future global assessments of urban biodiversity and ecosystem services, with respect to local to global governance.

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27.1 Introduction

It has been said that "if the 19th century was the age of empires, and the 20th century the age of nations, the 21st century will come to be known as the age of cities" (Choa 2012, p. 79). Furthermore, as the earlier chapters in this book make clear, important parts of the battle for sustainability will be won or lost in cities. To a significant degree, sustainability outcomes therefore depend on the effectiveness of the governance regimes of cities across the world. Not only does the majority of the world population live in cities, urban populations are large consumers of ecosystem services (Folke et al. 1997; McGranahan et al. 2005; Grimm et al. 2008), and urban areas are the primary source of global environmental impacts (Ehrlich et al. 1970; Hardoy et al. 2001; Bai 2007). Few cities, even rich cities, are currently managing their biodiversity effectively (but see the case study chapters in this volume for inspiration on what can be achieved). Finding ways to better govern human-nature relations in individual cities and across the global urban system is thus paramount.

The purpose of this chapter is twofold. First, it provides a history of the belated emergence of an interdisciplinary global urban biodiversity and ecosystem services governance agenda, something that only happened in the late twentieth century for reasons described in Chap. 2. Second, because there has been no published global synthesis of the urban biodiversity and ecosystem services governance literature to date, it presents an overview of scientific material published on the challenges and opportunities associated with governing urban biodiversity and ecosystem services at the local, national, regional, and global scales. The chapter concludes by discussing what is required to improve governance of urban biodiversity and ecosystem services and sets out a critical research agenda to inform future global assessments of urban biodiversity and ecosystem services, with respect to governance.

It has been noted earlier in this volume that the genesis of urbanism was associated with the breakdown of individual and collective responsibility for the shifting ecological base of human consumption, production and the associated evolution of the form of urban settlements. Failure to embed an ecosystem perspective into the fiscal, regulatory and enforcement regimes can be seen at the global, national and local scale. Before establishing where the current interest in governance of urban biodiversity and ecosystem services emanates from, it is worth clarifying the term governance, in distinction to government. Governance can be viewed as "all 'collective action' promoted as for public purposes, wider than the purposes of individual agents" (Healey 2007, p. 17). This can include semi-autonomous relationships between the authorities on various levels, the civil society and private sector and its dynamics over time, with partly conflicting and overlapping agendas. The fragmentation of the capacity of the state to influence the urban system in and of itself has been characterized as the shift from government to governance (Rhodes 1997). With this fragmentation comes the need for governments to operate in a world with a range of other actors and factors influencing outcomes (Stoker 1998). This includes recognizing the capacity of civil society (Lee 2003), how some actors have more influence than others (Healey 2007), how governments are influenced by actors and dominant agendas at other scales (Marcotullio and McGranahan 2007), how governance outcomes are shaped outside the arenas of public control, and the limits of the capacity of the present public institutions (Healey et al. 2002). Which factors influence governance and shape outcomes thus depends on the local context.

In this chapter, we focus on both biodiversity and ecosystem services, and we are particularly interested in the ecology of cities and ecology in cities (cf. Chap. 3). With such a broad scope it is worth highlighting specifically what it is that needs governing and why the city scale is so important. Generalization is not simple – as reading across our rich but diverse city case studies of Bangalore, Cape Town and Stockholm reveals (see Box 27.1).

Box 27.1 Ecological and Governance-Related Challenges in a Selection of Cities Around the World

For more information and references, see Chap. 6, the local assessment of Bangalore; Chap. 17, the local assessment of Stockholm; and Chap. 24, the local assessment of Cape Town.

City	Ecological challenges	Governance challenges
Stockholm	Strategies to densify the city challenges conservation of green areas and wetland habitats. The number of Red-Listed and keystone species, such as oak trees, in urban areas is decreasing. Expected climate changes such as warmer air and water will further affect the future floral and faunal species composition and behavior.	A high exploitation pressure and a system of self-governing local municipalities challenges regional conservation planning. Long-standing lack of realization within planning of the importance of mitigation and adaptation to climate changes. Protected areas do not match critical ecosystem interactions. Underfunding and attempts to find alternative uses of the green areas often leads to degradation. The role of informal land management such as allotment gardens is poorly recognized but has gained support by changes in current planning frameworks under way of recognizing the importance of ecosystem services for sustainable regional growth.

(continued)

Box 27.1 (continued)

City	Ecological challenges	Governance challenges
Bangalore	The fast-growing city periphery experiences a relatively higher fragmentation and loss of vegetation than the city core. Related changes include deterioration of biodiversity and soil quality, aggravation of urban heat island effects, increased pollution of air, land and water, flooding, water scarcity and disease epidemics. Citywide challenges include encroachment on urban water bodies, severe water and air pollution, extensive tree felling, development of green spaces into built-up land and an increase of water-hungry, exotic species in parks. It is expected that the main future challenges will be of rising temperatures due to climate changes, and scarcity of clean water. Loss of lakes, wetlands and urban green spaces are expected to contribute to increasing the challenges.	A multiplicity of governance institutions with overlapping and often uncoordinated jurisdictional responsibilities prevents effective ecosystem management and urban planning. There is little formal recognition of existing and potential role of the civic society, which is directly involved in ecological management as garden owners and park visitors. Furthermore, civic society networks monitor lake encroachment and work towards urban ecosystem protection and restoration at large.
Cape Town	Population pressure creates challenges for the biodiversity-rich lowland areas compared to the highly elevated and formally protected Table Mountain. While local vegetation is fire-dependent and does require natural burning regimes to maintain ecosystem health, accidental fires started inadvertently by people can lead to too frequent and uncontrolled burning that poses danger to nature, people and property. Animals, such as baboons, frequently visit neighborhoods, causing human-wildlife conflicts. Formal housing and commercial development sees the ongoing conversion of remnant land. Informal settlement encroaches on remnant patches of biodiverse vegetation and formal conservation areas. Rivers and wetlands around the City have been impacted by urbanization, for example by pollution, canalization and being cut-off from their connected reaches.	While good national, provincial, and local environmental legislation and policies exist, implementation and enforcement is often weak, due to conflicting demands, lack of implementation mechanisms, or fiscal restraints. Environmental conservation has lower priority than other areas of city development, and is not yet effectively integrated across complementary departments and initiatives. Conservation targets for national vegetation types show that all vegetation types confined to the lowland areas are poorly conserved and currently fall below their conservation targets. Insufficient remnants remain to conserve representative diversity. Several lowland areas have a number of smaller reserves but the scale, number and connectivity of these smaller reserves do not meet identified conservation goals.

Although there are many shared biodiversity and ecosystems problems faced by and emanating from cities, the way in which these manifest in different cities is unique, not least because of the biome or region in which they are situated. Furthermore, each city has a distinctive cultural heritage, development history, planning tradition and social structure. Moreover, the knowledge base about the ecology of and in cities is uneven. This is the first global assessment with a focus on biodiversity and ecosystem governance and so the following sections examine the emergence of the field and provide a scientific review of the published knowledge on the subject. The focus in this chapter is general, and does not deal comprehensively with the sector based issues of water, air, food or land. As the previous chapter set out, even within a specific sector like food, there are complex challenges of protecting and promoting biodiversity and ecosystem integrity that must be met not just by the state but by specialists, civil society and governments. These governance responses will moreover take place across a range of scales, from within a particular city to a nation and across the world. Paradoxically, the acknowledgement of the imperative for a global response to the diversity of urban challenges draws attention to the minimal and fragmented city scale traditions of biodiversity curatorship. With this in mind we turn to explore the missing ecology in city governance.

27.2 Understanding the History of Urban Biodiversity and Ecosystem Services Governance

Ideas change – and this is no truer than in the work on urban biodiversity and ecosystem services. The fluid terrain that we are reporting on is made more complex because understandings of cities and ecological systems are both new and changing fairly rapidly. The values underpinning how contemporary cities should be managed have developed dramatically over the last 200 years as cities themselves have grown and, as a result, the nature of the urban ecological interface is not a static field of enquiry. In Chap. 2 we noted the failure to address the post-industrial-revolution splintering of urban development from its ecological hinterland and base; a situation only recently challenged by the *Cities and Biodiversity Outlook – Action and Policy* (Secretariat of the Convention on Biological Diversity 2012) call to reintegrate biodiversity into the regimen of urban management and planning. This global endorsement of the imperative of addressing the urban scale represents a milestone, in which urbanization has finally been recognized as a necessary component of the international and local biodiversity governance agenda. However, this is a relatively recent development and one that still lacks adequate international uptake. Of special concern is the significant portion of the urban world population that lacks any locally applicable and robust scholarship on ecosystem and biodiversity challenges and opportunities, and for whom the value of new scientific research in shaping urban governance is minimal. The overlap between cities that lie in the scientific shadow and cities that are rapidly expanding and are often poorly managed is high, making the geographical expansion of the urban biodiversity and ecosystem agenda a prerequisite for global impact.

The move to greater recognition of urban biodiversity and ecosystem services within science and policy has been accompanied by increasing cross-disciplinary academic efforts and, to some extent, cross-sectoral professional initiatives. We begin our overview of how urban areas have been identified as key sites of biodiversity action by tracing the emergence of an interdisciplinary global urban biodiversity and ecosystem services governance agenda, and remain mindful that acceptance by many may also imply ownership by none.

27.2.1 The Emergence of a Global Urban Biodiversity and Ecosystem Service Governance Agenda

The relationship between cities and environmental degradation has long been of concern to urban dwellers, although historically the state of the environments of cities was only considered important given the threat of disease. The emergence of penicillin muted the focus of municipalities on the public health threats posed by poor quality air, water and waste for many decades. Recently though, the understanding of the link between effective governance and the urban environment has once more come under scrutiny, though now in relation to global environmental change and the global environmental agenda (Rees and Wackernagel 1996) rather than the threat of disease, although that too is shifting with a resurgent interest in the complex systems that underpin urban health and well-being.

Cities started to grow quite rapidly in Europe and North America following the industrial revolution. Pollution became a serious issue affecting human health, but urban expansion also impacted the integrity of ecosystems (e.g., through the disruption of the biochemical cycles) (Haughton and Hunter 1994). After the Second World War and a following liberalization of global trade, cities developed from having mainly local and regional impact, to becoming global drivers of environmental change (e.g., through land use change) (Marcotullio and McGranahan 2007; Lieberherr-Gardiol 2008) (see also Fig. 2.2).

The contemporary environmental agenda focusing on global environmental change emerged in the early 1970s. Awareness of environmental degradation and the planet as a system with limits to growth emerged in both civil society and among decision makers (cf. Meadows et al. 1972). The environmental agenda of cities is thus necessarily woven into the history of the wider global environmental agenda (Sánchez-Rodríguez et al. 2005; Seto et al. 2012) and the development agenda (Parnell et al. 2007). Recognizing cities as engines of economic growth and centers of production and consumption also implies acknowledging that cities draw on resources from all over the globe (Redman and Jones 2005). Significantly, echoing a point made elsewhere in this volume, the new awareness of the importance of urban ecological governance reform bridged the global and local scales, and conceptualized cities as embedded in a larger natural hinterland – (a hinterland which, given new transportation and distribution capacities, may or may not be physically contiguous). This locational splitting of cities and their resource base compounds the complexity of the governance challenges, thus creating imperatives for internationally orchestrated improvements to urban ecosystem management.

The massive growth of cities in Africa, Asia, and Latin America in the late twentieth century, often without any bulk infrastructure for sewerage or systems of urban regulation to protect the environment, resulted in considerable urban environmental degradation (McGranahan and Satterthwaite 2003; Marcotullio and McGranahan 2007; Pieterse 2008). Indeed, it is in these cities of the Global South, where the majority of future global population growth is expected, that some of the most severe public health and urban ecosystem and biodiversity challenges lie (Parnell et al. 2007), not least because of their weak systems of formal government and planning.

Cities have rarely been a central issue in the international environmental politics arena (Puppim de Oliveira et al. 2011). An early exception is the report *Our Common Future* (WCED 1987) that included a chapter on urbanization and which led to mainstreaming of the term "sustainable development." It recognized a rapid urbanization at a global scale and the central role of cities in the global economy as "the backbone for national development," suggesting that the prospect of any city "depend(ed) critically on its place within the urban system, national and international. So does the fate of the hinterland, with its agriculture, forestry, and mining, on which the urban system depends" (WCED 1987, p. 196). The report had a particular focus on 'less developed' countries and highlighted the lack of capacity of local authorities to deal with uncontrolled population growth. Many African and Asian states were described to have institutional structures highly influenced by their time as colonies, with governance systems intended to govern a rural economy and society, and leave cities as metropolitan spaces of the colonial elite. The political, institutional and legal frameworks in most Latin American (and by implication, African and Asian) cities were held to be inappropriate and unable to match the challenges of rapid urbanization (WCED 1987). The report also highlighted that national authorities were not enabling local authorities to deal with environmental challenges; this unleashed the then-fashionable decentralization impetus to drive a new urban ecological agenda.

The role of local authorities in environmental governance gained further focus during the Earth Summit in Rio de Janeiro in 1992. The event, being a direct response and follow on from *Our Common Future*, resulted in the initiation of Agenda 21, a program for action addressing actors at all levels of society and focusing on the promotion of sustainable development. Local authorities were asked to prepare Local Agenda 21 (LA21) plans based on motivations that included statements such as: "*In industrialized countries, the consumption patterns of cities are severely stressing the global ecosystem, while settlements in the developing world need more raw material, energy, and economic development simply to overcome basic economic and social problems*." (UNCED 1992, p. 45). Countries were encouraged to assess the environmental impacts of current urban policies and growth, and cities were advised to establish networks for cooperation and sharing of best practices. Significantly, what the LA21 program signaled was the importance of cities and other local authorities as important sites of ecosystem government and governance. Since then, the issue of the most appropriate scale of biodiversity and ecosystem governance has been an enduring concern.

Concern over defining the most appropriate scale of action is key, as cities typically follow a trajectory from very local environmental problems to improvement of living conditions by dispersing these challenges both spatially and temporarily, consequently having an effect on long-term global environmental status (Marcotullio and McGranahan 2007). Reflecting how hard it was to insert the global urban agenda into the international environmental governance arena, McGranahan and Satterthwaite (2003) recall that both the urban parts of *Our Common Future* and Agenda 21 were almost dropped due to political disagreements. The progress on LA21 in cities was, unsurprisingly then, slow (Allen and You 2002). In 2005, the landmark United Nations (UN) report on ecosystem services, the Millennium Ecosystem Assessment (MA 2005) was launched, which, whilst including a subsection in the 'Current State and Trends' Section on 'urban systems,' was critiqued as not substantially addressing urban areas throughout the Assessment (Alfsen et al. 2011). Later, in the context of a predominantly urban world, there has been an increasing recognition of cities as actors and important areas of work under the Convention on Biological Diversity, e.g., through the Curitiba declaration in 2007 and later initiatives leading to the Cities and Biodiversity Outlook (CBO) project and publications (Secretariat of the Convention on Biological Diversity 2012). During COP9 of CBD, the decision IX/28 was adopted, encouraging parties to recognize cities in National Biodiversity and Action Plans including the preparation of local strategies and action plans, in addition to initiating an evaluation tool for cities – The City Biodiversity Index (CBI) (see Chap. 32). At COP10, decision IX/28 was complemented by a Plan of Action on Sub-national Governments, Cities and Other Local Authorities for Biodiversity, giving further advice to parties and a request for an "assessment of the links and opportunities between urbanization and biodiversity for the eleventh meeting of the Conference of the Parties" (UNCBD 2010: X/22). In June 2012, twenty years after the first Rio meeting, world leaders met in Rio and highlighted in the outcome document that if "well planned and developed, including through integrated planning and management approaches, cities can promote economically, socially and environmentally sustainable societies" and emphasized "promotion, protection and restoration of safe and green urban spaces; safe and clean drinking water and sanitation; healthy air quality" (UN 2012, p. 26).

This, alongside the introduction of an urban chapter into the fifth assessment report of the Intergovernmental Panel on Climate Change, gave hope that the urban question was now firmly on the international environmental policy agenda. The argument made across this volume – that city scale action is a necessary but not sufficient requirement to meet future ecosystem and biodiversity challenges – only underscores the importance of international (and national) action to make cities more resilient.

Outside of UN processes, many cities across high, medium and even low income contexts have continued to try to deal with problems related to environmental risk, ecosystem health and sanitation. Livability and smart growth policies have received an increasing focus in cities located in rich countries; they aim to reduce urban sprawl into surrounding land by cleaning up the core areas of the city like old industrial sites (typically waterfronts) and making city-center life more attractive for middle- and high-income citizens that often live in suburbs (Allen and You 2002). Cities in developing countries have also struggled with rapid spatial growth. One third of the children growing up in cities live in slums where they are exposed to polluted rivers and air and hazard pollutants (UNICEF 2012). The environmental dimensions of wider urban problems have thus become much more central, such that it is now almost impossible to uncouple a discussion of urban development from that of the urban environment and its ecological base (Allen 2003; Satterthwaite 1997; Swyngedouw 2005).

One aspect of the urban environment that has received relatively poor attention, not just with respect to governance issues, is that of biodiversity. As in the case of climate change and the C40, where there is a global movement to address biodiversity concerns, it has once again been cities, not nation states, which have been at the forefront of the global mobilization. Recently, initiatives by cities to share best practices and support the aims of the Convention on Biological Diversity (CBD) include support for the Curitiba Declarations (2007, 2010), the Durban commitment (2008) and the Bonn call (2008). Gradually, a global movement for biodiversity and ecosystem services that incorporates an overt urban emphasis is emerging in the international community.

Figure 27.1 captures the rich tapestry of organizations involved in and driving an urban biodiversity and ecosystem governance agenda at the global scale. These are both formal institutional bodies of the UN system, but also powerful global NGOs such as ICLEI. A number of high profile international meetings have generated consensus on the key issues and parties have made commitments to implement actions to achieve targets. Learning from past difficulties around implementation, global programs of action now provide the support structures for implementation. Of note in this regard are the diverse major initiatives highlighted in Figure 27.1. The implementing actors for urban biodiversity thus draw not only on pure ecologists, but also statisticians, planners, medics, economists, and social scientists.

27.2.2 Interdisciplinary Perspectives

There are well-known and established bodies of research exploring human–nature relations in and of cities, from disciplines including geography, history, archaeology and, of course, planning. Indeed, there is a long history of attention to human–nature relations through design and planning practice (Wilkinson 2012). Since the emergence of town planning as a discipline, human–nature relations have been high-lighted through the Chicago School of planning, the early British town planners such as Ebenezer Howard (1850–1928), Patrick Geddes (1854–1932) and his influence on Lewis Mumford, and later on through more

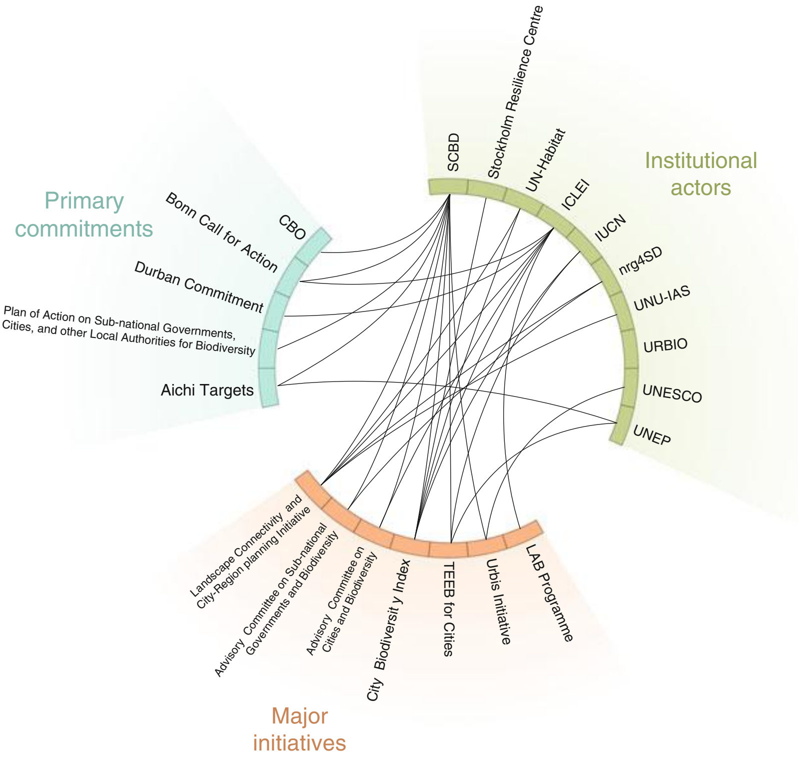

Fig. 27.1 Overview of global governance arrangements for urban biodiversity and ecosystem services (Prepared by and published with kind permission of © UN Habitat 2012. All Rights Reserved)

detailed practice-based attention of how to design with nature. American sociologists at the Chicago school, for example, began investigating human behavior and the environment in cities already in the 1920s. From the 1970s, environmental planning emerged as a sub-discipline (Slocombe 1993) and from the 1990s onwards this relationship is explored through the sustainability discourse (e.g., Owens and Cowell 2002; Rydin 2010). Most recently, the emerging field of urban ecology has taken up this interdisciplinary perspective (McDonnell 2011). Urban ecology is defined as "the study of the ways that human and ecological systems evolve together in urbanizing regions" (Alberti 2008, p. xiv), and it "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" (McDonnell 2011, p. 9).

The emergence of the field of urban ecology is significant because urban areas were not a research priority among ecologists until late in the twentieth century (Grimm et al. 2008). Born of a narrow focus on urban biotopes and concern over introduced species (Sukopp 2002), after the 1970s a new approach emerged that focused on the city as a whole, with a focus on energy flow and nutrient cycling in this system (Wolman 1965; Boyden et al. 1981; Sukopp 2002). The more recent development within research on urban ecology views "cities as heterogeneous, dynamic landscapes and as complex, adaptive, socioecological systems, in which the delivery of ecosystem services links society and ecosystems at multiple scales" (Grimm et al. 2008, p. 756); this change incorporates the field of landscape ecology (McDonnell 2011). As human-dominated systems, a shift from a traditional biophysical focus to a more social and interdisciplinary one is perhaps most logical in cities, and such studies are now increasing in numbers following landmark articles that identify humans as an important driver of environmental change from the local to the global level (such as Berkes and Folke 1998 referred to in Young and Wolf 2006). Key projects aiming to address the urban-ecological knowledge gap include the recent work on urban long-term ecological research programs (LTER) studying Baltimore, Phoenix and Maryland in USA (Grimm et al. 2000). The city case study chapters in this volume illustrate the huge range of work that is being undertaken at a local scale in the area of urban biodiversity – not all efforts are centrally concerned with governance, but for those stewards of the cities' ecosystems, the locally credible science provides the evidence base for policy reform and implementation.

Sociologists and geographers are among the social scientists whose studies, influenced by Marx and his concepts of labor power, metabolism, and uneven development, generated a massive body of work known as political ecology. Political ecologists investigate the production and transformation of social nature and its role in the differentiation of space at a variety of scales with recent emphasis on how society relates to nature under dominating neo-liberal policy frameworks (Pincetl et al. 2011). Urban political ecology research has been especially fruitful in the study of power relations and material flows and fluxes operating across regions and cities (cf. the influential work of Swyngedouw 2006).

Over time and through the work of sociologists, economists and psychologists, studies of social and ecological, as well as economic and technical aspects of the city have become more integrated in urban ecology (Young and Wolf 2006). Research, stemming from geography and political science as well as ecology, has broadened its scope from within cities – viewing cities as something separate from the world – to a research integrating cities into a wider landscape – where they are recognized as global actors of change (in line with Berkes and Folke 1998). A more recent perspective in urban ecology views cities as microcosms – systems where the change predicted in estimates of global environmental change are happening more rapidly. Pioneering social and environmental research is now focused on how to respond to the catalytic role of cities (Grimm et al. 2008; McDonnell 2011). From a governance perspective, recognizing that these 'city microcosms' are far from closed (because the contact between the urban and rural is blurred and the administrative boundaries do not neatly correspond to those of ecosystems) is more relevant than ever.

Moving to the global perspective, cities have also been studied as a global network rendering the planet not only increasingly human dominated, but also urban dominated as "cities need to be viewed as loci in multiple networks of relationships at different scales, rather than as entities" (Ernstson et al. 2010a, p. 537). This interpretation comes from geographers like Beaverstock et al. (2000) in their notion of a world city network or metageography. Swyngedouw and Heynens (2003, p. 899) develop this notion of urban political ecology by suggesting that "the socioecological footprint of the city has become global. There is no longer an outside or limit to the city, and the urban process harbors social and ecological processes that are embedded in dense and multilayered networks of local, regional, national and global connections." This perspective echoes urban ecological studies of cities that view cities as human-dominated ecosystems, with authors like Bolund and Hunhammar (1999, p. 294) arguing that "when humanity is considered a part of nature, cities themselves can be regarded as a global network of ecosystems."

Notwithstanding the well-established and disciplinarily diverse roots of research on urban ecology, it is true that over the last decades there has been a dramatic increase in awareness of biodiversity and ecosystem services issues in and of cities. Moreover there has been a massively expanded response from residents, civil society, local government as well as national and international stakeholders concerned to respond to the critical biodiversity challenges presented in and by cities. In an effort to ensure that we maximize the potential of knowledge to inform practice – for scholars to learn from practice and to encourage the documentation and dissemination of pathways to enhance urban biodiversity and ecosystem services – our attention now turns to providing a synthesis of the scientific literature on governing urban ecosystem services.

27.3 Synthesis of the Scientific Literature on Governing Urban Ecosystem Services

27.3.1 Scope of the Synthesis

A synthesis of the governance challenges and opportunities relating to urban biodiversity and ecosystem services is presented here; it draws on a systematic literature review carried out specifically to inform the CBO process (Sendstad 2012). The purpose of the literature review was to take a first step towards generating a much-needed comprehensive global assessment of knowledge of urban biodiversity and ecosystem services governance. The rationale for drawing on a systematic review of the academic literature is to be transparent about the published, peer-reviewed scientific foundation of knowledge on governing urban biodiversity and ecosystem services. We recognize that local knowledge, traditional knowledge and other knowledge contained in reports generated outside of academia (i.e., grey literature) are also important to the governance of urban biodiversity and ecosystem services. Indeed there is much other material on biodiversity and ecosystem services that is used by cities and urban communities to inform regulatory, distributive and restorative practices. However, as there tends to be a scientific integrity and professional weighting associated with peer reviewed published material, for example in the medical profession but also in global assessments such as the IPCC, our focus at this stage falls on this scientific foundation (see the Preface of this volume for a further discussion of literature included in the CBO).

The synthesis of challenges and opportunities relating to the governance of biodiversity and ecosystem services draws on the published findings of 138 scientific articles published in English in 76 journals. The papers were sourced using categories of words to represent the three main focus areas of the study: **governance** – of **ecosystem services** – in **urban** settings.[1]

Relying on the published academic English language literature creates a significant geographical bias. A total of 88 cities or urban regions from 23 countries were represented in the studies reviewed. There was a clear bias towards Europe (32 studies from 27 cities/urban regions from 9 countries), North America (28 studies from 26 cities/urban regions in USA and Canada) and China (22 studies of 11 cities/urban regions). In addition to these studies, there were also some studies looking at a large number of cities within a given country, e.g., studying land use change response to policy across cities. Africa, South America and parts of Asia are almost totally invisible in the literature, regions on which published data is known to be scarce. A further reason for the lack of profile of cities in the developing world may be limitations due to the selected databases and keyword combinations. Furthermore, in large parts of the world, scientific studies are often published in languages other than English (e.g., French for Africa, Spanish for Latin America, and Russian or Chinese); this results in potentially valuable studies going undetected by the database searches. However, the search results reflect a more general gap in scientific knowledge about the experiences in these under-researched regions. It is thus imperative that future reviews undertake a geographical and thematic corrective, if necessary embracing grey literature and undertaking primary research to ensure better global coverage and to extend the range of issues profiled.

The absence of published scientific work on many important issues and places must be noted as a major distorter of our collective understanding of the scale and scope of the challenges and opportunities for biodiversity that are presented by urbanization. The fact that many of the global biodiversity governance challenges emanate from specific cities or regions suggests that the currently geographically-incomplete knowledge pool may critically undermine universal or networked responses to urban biodiversity problems. Furthermore, the value of the existing scholarship on urban biodiversity governance is undermined by the fact that ideas about biodiversity governance are neither universal, nor do ecological management practices necessarily transplant well from city to city. Given that the bulk of the world's population lives in those cities that have the least biodiversity research, gaps in the sources that inform governance responses must be highlighted as a very

[1] A full methodological note is set out in Sendstad (2012).

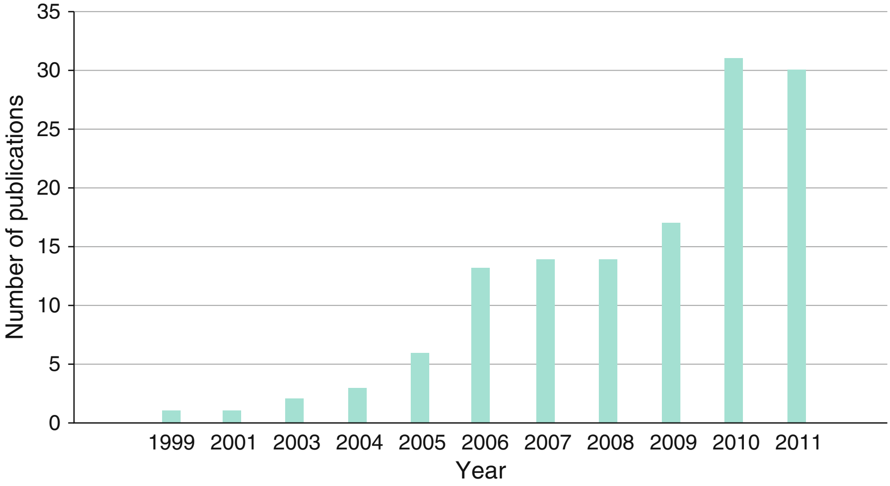

Fig. 27.2 Number of publications sorted by year for studies included in the literature review. The review was finished in spring 2012 so some studies have been included from this year (Modified from Sendstad (2012). Published with kind permission of © Marte Sendstad 2013. All rights reserved)

serious concern. That said, there is, despite a somewhat tardy beginning, now a growing interest in the governance of cities for biodiversity enhancement and protection. Our review suggests that there is sufficient scientific evidence to confirm that how cities are managed impacts both positively and negatively on biodiversity: governance regimes do matter.

Despite its relative youth, the field of biodiversity and ecosystem services has generated a good deal of peer-reviewed material on issues that are explicitly linked to questions of urban governance (see Fig. 27.2). A number of the earlier chapters in this book review the state of knowledge in specific sectors and highlight the uneven uptake of the science as well as the geographically distorted limits to knowledge on critical ecosystems on which cities depend and which city growth impacts (see, for example, Chaps. 10, 12, 21, 22, 26, and 33). Even where there is sufficient science it does not follow that this knowledge will inform action. Several of the published studies highlight the lack of awareness and narrow understanding of ecosystem functioning among decision makers, suggesting that it is not just residents who struggle to absorb the arguments of science at the local (Moll 2005; Li et al. 2005b), regional (Merson et al. 2010) and global scale (Puppim de Oliveira et al. 2011). It is clear then that advancing the urban biodiversity and ecosystem services agenda is only in part a question of proving the biological science; a dominant challenge seems to lie in the institutional capacity to govern biodiversity and ecosystem services as well as in shifting the way science is viewed and used in an urban setting characterized by conflicting views and interests among stakeholders. Before looking at how the science has been used by cities, we pause to reflect on the scope and utility of the available science.

27.3.2 Urban Biodiversity and Ecosystem Services: Governance Challenges

27.3.2.1 Do We Have Enough Science to Reliably Inform Implementation?

At the highest level there is a lack of scientific knowledge, especially about urban ecosystem structure and function (Boyer and Polasky 2004; Niemelä et al. 2010). There is also a lack of scientific literature on urban environmental governance (Wilkinson 2012). As urban governance capacity to implement the findings of scientific research improves, the extent of the pure science gap will become more obvious, especially in cities that currently lack a tradition of using scientific support for ecosystem management. In some cities, there is available science to better inform the governance decisions of city managers, and the published literature is skewed to these well-resourced and well-researched places (see Chap. 17, the local assessment of Stockholm, which sets out the role of science in one of the leading evidence-based biodiversity transformations of urban ecosystem management, but also Chaps. 16 and 24 on the middle-income cities of Cape Town and Istanbul, where rich traditions of local ecological research now inform transformative municipal practice). These cities are perhaps exceptions for the depth of knowledge they are able to garner, but the greater problem is not simply the absence of science – there is a spatial mismatch between where the scientific studies occur and where the world's urban ecosystem and biodiversity problems manifest.

Planners and decision makers, even those committed to a more evidence-based practice (Alonso and Heinen 2011) are not always able to use the publications of scientists for a number of reasons. First, practitioners struggle to accommodate the uncertainly that scholars outline (Fang et al. 2006; Niemelä et al. 2010; Su and Fath 2012). Second, at the local level in particular, there is a dearth of specialist ecological data and analysis needed to support legitimate regional planning and policy development (Peterson et al. 2007; Mendiondo 2008; Boyer and Polasky 2004). Third, while there may be specialist studies available, there is a lack of scale- and context-appropriate scientific tools and methods to capture the complexity of interacting systems, the limits of ecosystems and the drivers of change (Merson et al. 2010; Puppim de Oliveira et al. 2011). Finally, even in contexts where decision makers have access to relevant knowledge, it may take time before this has an effect on policy, public awareness and political action (Lieberherr-Gardiol 2008; Niemelä et al. 2010, p. 3238). One study from New York Metropolitan Area suggested that the connection between science and policy was weak because the scientific view was considered just one of many stakeholders involved in decisions (Alfsen-Norodom et al. 2004). Furthermore, while some see linking science and the views of stakeholders as offering potential for knowledge co-production (Bayá Laffite 2009), there are significant paradigm differences to be dealt with in mediating approaches to urban biodiversity and ecosystem service issues (e.g., Antrop 2001).

27.3.2.2 Political as Well as Intellectual Legitimacy Are Key

Cities themselves are complex systems, and introducing a new emphasis on the science of ecology into how urban areas are managed presents real challenges – not least because of the lack of political legitimacy traditionally associated with 'green issues.' Achieving the necessary political support and changing the habits of residents is also made difficult by the lack of awareness about the diversity of nature, its complexity, as well as human dependence on ecosystem functions across scale (Borgström et al. 2006; Wolch 2007). Some studies suggest that a personal experience may be important for caring about the protection of nature (Dearborn and Kark 2009). In a study by Jim and Chen (2006, p. 342) in Guangzhou (China), residents placed high values on services like air quality and aesthetic enhancement in contrast to facilitation of biodiversity, water treatment, and flood abatement, suggesting that they were unable to value what they could not see or had not experienced directly.

27.3.2.3 Integrating Environmental Equity and Justice

Governance or management of urban biodiversity and ecosystem services inherently raises questions of environmental equity and justice across spatial and temporal scales. Biodiversity and ecosystem services are often unequally distributed within the city (Li et al. 2005a, b); low income and minority groups tend to have lower access and be disproportionately burdened by environmental hazards (Bullard 1997; Adamson et al. 2002; Wolch 2007; Boone 2010; Perkins 2010). Poor people may be perceived as responsible for environmental degradation in spite of having a relative low per capita impact (Zérah 2006; D'Souza and Nagendra 2011) or having been allocated environmentally risky sites (Ernstson et al. 2010a). Ecosystem degradation may, however, be an important cause of urban poverty (MA 2005). Moreover, people who have a higher per capita responsibility for degradation of ecosystem services are often not the ones experiencing the cost. Costs related to environmental degradation and leading to quantitative or qualitative loss of biodiversity and ecosystem services may be displaced across temporal and spatial scales. Environmental inequity may also occur between urban and rural regions (see e.g., Gutman 2007; Sarker et al. 2008), but following globalization, equity is not merely a local or regional issue. The social and ecological costs of improved urban living conditions can be transferred through global trade flows (Hagerman 2007; Meng 2009). The role of institutions and institutional mechanisms in facilitating and influencing people's access to ecosystem services is critical for addressing distributional issues, ensuring that ecosystems are managed in a fair and equitable manner to all involved stakeholders. Payments for Ecosystem Services (PES) schemes are by some considered to be a more efficient approach to biodiversity and ecosystem services conservation. PES schemes do not, however, necessarily integrate concerns of equity, and may have the effect of "possibly even accentuating poverty and equity gaps by putting a cost-effective price to previously low priced or free services" (Pascual et al. 2009).

27.3.2.4 Gaps in Institutional Capacity Undermine Governance Effectiveness

The most frequently documented barrier to more effective ecosystem service management in cities in the academic literature is that of the institutional capacity of formal authority and structures, including the ability of such structures (most often local government) to plan and regulate ecosystem services. Further dimensions of the institutional gap relate to the ability of the responsible parties to acquire and handle relevant urban scale information and cooperate across levels of environmental and urban decision-making. This is not just a local problem, as national and international levels of governance have rarely focused on expanding cities' powers and resources in negotiating policies on the governance of ecosystems (Puppim de Oliveira et al. 2011).

Introducing new governance systems for urban biodiversity and ecosystem protection in cities is not simple. Examples from China are illustrative. In China, the central planning system was developed before decision makers had any significant awareness of the value of integrating environmental concerns into urban planning (Fang et al. 2006; Xu et al. 2011). Embracing the value of ecosystem services often means setting the economic imperatives of city development against the ecological. Findings from a study of Beijing showed that practically this means that compensation mechanisms may fail to protect green areas from real estate development if the fee developers must pay to build on green areas is significantly lower than the income prospects (Li et al. 2005a). Li et al. (2005a, p. 330) further found that the design of the green areas in Beijing focused more on "beautification" than on conserving the ecological value as habitat (see Chap. 3 for more information on trends and challenges in design for biodiversity and ecosystem services). The Chinese experience is not unusual; cities everywhere are faced with having to devise new norms and standards and embed the regulatory and enforcement practices into the planning systems to ensure ecosystem integrity. For most cities, this is an incremental and even ad hoc process that has not delivered a perfect ecosystem management system and the complex thing is that fragmented governance may erode ecological integrity by lack of holistic planning and responsibility (Alfsen-Norodom et al. 2004). This was the case in Toronto, Canada, where an ecologically valuable moraine area was developed piece by piece, due to approvals from different authorities (Wekerle and Abbruzzese 2010).

27.3.2.5 Navigating Competing Urban Priorities

One of the greatest difficulties for municipalities is to introduce a new policy priority into an already resource-stretched institutional environment, especially popular social policies like housing delivery (Barthel et al. 2005; Asikainen and Jokinen 2009; Wekerle and Abbruzzese 2010). Box 27.2 draws from the experiences of a number of cities to show how difficult it is to change the direction and mode of governing in ways that embrace biodiversity. Biodiversity does not simply compete with other spending or development opportunities. Delivery on

Box 27.2 Competing Priorities in Urban Policies; The Examples of Rio de Janeiro City and Tokyo

For more information and references, see Chap. 29, the local assessment of Rio de Janeiro City; and Chap. 8, the local assessment of *satoyama* and *satoumi* landscapes, Tokyo.

Rio de Janeiro city	The city of Rio de Janeiro is expanding at its fringes due to growing informal settlement areas as well as private and privatized public areas. Some of the world's most biodiversity-rich wetlands, and vegetated and forested areas are being covered, regardless of their formal protection status. Local inhabitants have initiated conservation and re-introduction of native local species. In the city at large, official legislation can be efficient but is continuously altered to favor development projects. One example is the golf course for the 2016 Olympic Games, which is being developed inside a high-priority conservation zone. Differing perspectives between people of different income groups can challenge whether or not urban greens be given priority in plans and management. Inhabitants in a low-income area were found to have a large interest in active work to conserve local ecosystems, whereas inhabitants in a high-income area were found to appreciate urban green areas but had a limited knowledge on the ecological benefits and did not actively engage in management of the urban greens.
***Satoyama* and *satoumi* landscapes, Tokyo**	Following a rapid and extensive urbanization in Japan and thus a decrease in human management of rural land, *satoyama* and *satoumi*, i.e., biodiversity-rich landscapes with long-standing management traditions, have decreased and degraded, leading to an overall decrease of biodiversity. At the same time, the support to transfer *satoyama* and *satoumi* to urban areas is undermined as the landscapes are treated separately from other types of urban nature in official conservation policies, such as the *Japanese National Biodiversity Strategies and Action Plan* (NBSAP). Japanese national policies typically provide a weak support for urban nature, as plans to enhance green infrastructure generally are proposed only *after* development plans are accepted, and the inter-relation between such plans and the urban *satoyama* and *satoumi* systems are not clearly identified. In addition, the governance structure also creates challenges: although the official, national aim is to promote conservation, regeneration and utilization of *satoyama*, this is often undermined when local ordinances instead tend to favor economic growth and development, which is shown in Tokyo's increasingly dense city core, where the competition for land is extremely high.

economic growth, jobs and housing constructs its own new pressure on ecologically valuable areas and several studies suggest that ecosystem services are given a lower priority compared to housing (Barthel et al. 2005; Asikainen and Jokinen 2009; Wekerle and Abbruzzese 2010), infrastructure, or jobs (Li et al. 2005a; Peterson et al. 2007; Wekerle et al. 2007; Wang et al. 2009), even if there are strategies in place to protect areas of particular value (Jonas and Gibbs 2003; Li et al. 2005a; Ozawa and Yeakley 2007).

The multiscalar dynamics of the ecosystem create major urban governance challenges because decisions across scales of government and have long-term implications that extend beyond the period for which elected officials are responsible. Ironically, the system of elected democracies and rotating political leadership may mitigate against the more resilient governance of cities, this is especially true in cites that lack strong regulatory or administrative instruments to 'depoliticize' everyday practices of urban management that foster or uphold urban biodiversity and ecosystem integrity. It is important that ways are found for the long-term sustainability of cities and effective ecosystem service management to be taken into account through political decision-making processes. The case of Bangalore (Box 27.3) is an interesting example where traditional values, rather than state regulation, provide the basis for collectively acknowledged values and practices that preserve biodiversity in the city.

Box 27.3 Traditional Knowledge and Civic Society Initiatives Protect Urban Greens in Bangalore, India

For references and more information, see Chap. 7, the local assessment of Bangalore.

Bangalore: Protection of urban greens and blues by a complex web of multiple actors, traditions and norms

Bangalore is India's fifth largest city and with a population approaching nine million, it is one of the world's most rapidly developing cities. Economic growth, paired with a multiplicity of governance institutions with overlapping and often uncoordinated jurisdictional responsibilities, has had a major impact on ecosystems and biodiversity. However, the civic society is involved indirectly in management of urban forests and lakes in a variety of ways, ranging from monitoring encroachment to engaging with city municipalities and political entities for restoration. They are also directly involved, as residential garden owners, park and lake visitors, and initiators of public activities such as lake restoration or environmental public interest litigations.

Social networks, such as the environmental activist group *Hasiru Usiru,* have contributed substantially to keep issues of urban conservation in the forefront of public awareness in recent years. Their efforts have, for example, resulted in influential court rulings on issues of tree felling (Sudhira 2007; Enqvist 2012). The city's bird-watching community has facilitated environmental monitoring and awareness by online discussion forums, meetings and events. In the annual Bird Race, participants have cumulatively logged over 230 species of birds in and around Bangalore in a single day.

(continued)

Box 27.3 (continued)

Local norms and traditions commonly contribute to biodiversity protection. Home gardens in Bangalore are rich in plants selected for their cultural and medicinal properties (Jaganmohan et al. 2012). Even in impoverished parts of the city, greenery and plants play an extremely significant role due to the critical social, cultural, religious, medicinal and food-related ecosystem services they provide (Gopal 2011). Historic cemeteries and sacred sites around mosques, temples and churches provide protection for heritage trees, ecological habitats such as anthills, and keystone species such as the sacred figure. New conservation strategies are needed to carry the strong potential for nature conservation of norms and traditions, into the modernization process of the city.

27.3.2.6 Governance Challenges Related to Scale Mismatch

Challenges related to temporality and scale can be seen as core governance dilemmas. The literature indicates that temporal, spatial, and functional mismatches between ecosystems and the institutions managing them may be an overarching challenge in ecosystem governance (cf. Lee 1993; Cumming et al. 2006) Although scale-mismatch in urban areas as a concept is mentioned overtly in relatively few studies (Borgström et al. 2006; Ernstson et al. 2010b), it is a dilemma that permeates the literature either because of fragmented governance (where several jurisdictions exist within the city or the urban–rural region) or because ecosystem functioning does not align with administrative boundaries (Borgström et al. 2006; Wekerle and Abbruzzese 2010). Box 27.4 provides local examples from Melbourne and Istanbul that detail how scale mismatches in governance can frustrate biodiversity governance. A particular challenge related to spatial mismatch concerns how urban areas link to their regional to global sources of ES (Alfsen-Norodom et al. 2004; Blaine et al. 2006; Gutman 2007; Sarker et al. 2008; Puppim de Oliveira et al. 2011). Studies of aquatic ecosystems and water quality find that land managers upstream can influence ecosystems in cities without taking the needs of urban people downstream into account (Blaine et al. 2006; Sarker et al. 2008). Urban residents however draw on resources from all over the world (Alfsen-Norodom et al. 2004), without necessarily paying the full cost related to ensuring the integrity of the relevant ecosystems from which these resources are derived (Puppim de Oliveira et al. 2011).

27.3.2.7 Trade-Offs

There are many synergies in governance of urban ecosystem services (ES) and biodiversity, like regulating services supporting a number of other services (Raudsepp-Hearne et al. 2010). It should however be recognized that governing

Box 27.4 Scale Mismatches Are an Ongoing Challenge for Biodiversity Governance

For more information and references, see Chap. 16, the local assessment of Istanbul; and Chap. 20, the local assessment of Melbourne.

Istanbul	Spatial planning power was transferred from the central government in Ankara to local authorities in the 1980s. However, a remaining lack of engagement by the civil society in urban development politics allows for the misuse of political power. Although environmental concerns have been taken into account in spatial planning since the 1960s, the management of significant biodiversity locales and ecosystem services is poorly coordinated and fragmented. This stems from a division of responsibilities over several departments within the metropolitan municipality and the central government; poorly coordinated responsibilities; and a complicated juridical framework. As a result, Istanbul faces serious problems for example for the fresh water management, and chronic fresh water shortage is already a long-standing problem. The lack of effective regulations aimed to protect ecosystems, and the weak enforcement of existing regulations, has allowed illegal settlements and developments to expand through valuable areas such as the Ömerli Watershed, wherefrom Istanbul gets the majority of its fresh water. As a result of human activities and the lack of effective watershed management tools, there is an increasing risk of water pollution from different sources such as sewage, industrial wastewater and urban runoff.
Melbourne	As the city grows and expands at its fringes, there is an increasing need to address urban growth and conservation objectives, and management of 'native' and 'exotic' vegetation. Four factors will largely determine the degree to which Melbourne will be able to support a healthy human population and flourishing biodiversity in the future: city growth on the fringe; habitat management in established areas; management of green assets; and directions in local biodiversity governance. Melbourne's principal local planning instrument, planning schemes, are developed by local governments within a framework established by the Victorian State Government. However, many strategies to support biodiversity at both local and regional scales are executed poorly due to political and economic pressures. In addition, certain trends of suburban development can lead to a gradual homogenization of biodiversity. For example, a small number of plant species are commonly used in street and landscape plantings in master-planned estates. Greater appreciation by local governments of the interrelationships between biodiversity and human well-being will allow new ‘green’ solutions to be found in everyday planning and infrastructure decisions. Greater integration of environmental policies with other regulatory instruments will also help to promote biodiversity in the city into the future.

urban ES is not merely about finding synergies, but can often entail navigating trade-offs. This could entail prioritizing some ES at the cost of reducing the provision of others (Rodríguez et al. 2006). One example of this is establishing a homogenous lawn that has recreational benefits, e.g., for sport activities, but has a limited value in terms of people experiencing biodiversity, as that requires a more varied landscape with a higher habitat value. Trade-offs are also common between ES and other goals in policy, both regarding monetary and non-monetary costs and benefits. For example, vegetation does contribute to local climate regulation (Hung et al. 2006), but also requires water, which may be a scarce resource, and vegetation such as trees sometimes must be managed in order to prevent interference with urban infrastructure. Navigating trade-offs raises scale issues but also consideration of the extent to which the decision is reversible (Rodríguez et al. 2006). Matters of environmental equity and justice highlight the challenging trade-offs between various beneficiaries (cf. Rounsevell et al. 2010). Different stakeholders may (unsurprisingly) have very diverging views on these trade-offs and conceptions of their relationship to different ES across the urban landscape – this is politics (cf. Karvonen 2010). These conflicting views need to be taken into account and addressed to be efficient in governance of urban ES (see also Sect. 27.3.3.4, below).

27.3.2.8 Effective Ecosystem and Biodiversity Governance Requires Collaboration

Governing ecosystem processes requires coordination across levels of policy and legislation, as typically all spheres or tiers of government are involved in urban ecosystem services in some way (see Box 27.5) (Peterson et al. 2007). A common issue is that policies focus narrowly on endangered species or habitats, without incorporating ecosystem change over time (Asikainen and Jokinen 2009; Ernstson et al. 2010b). In Sweden, Elander et al. (2005) found that it was challenging for urban planners at the local level to implement national biodiversity strategies, since they were too general and abstract. Bomans et al. (2010) also point out a weakness in spatial policy based on coarse, mono-functional categories, unable to take into account transformations in multiple land uses and related values tied to the rapidly changing urban landscape. Numerous studies indicate a lack of regulation connecting urban consumers of ecosystem services and the people managing the resources they depend on outside the city boundaries (Blaine et al. 2006; Gutman 2007; Sarker et al. 2008; Puppim de Oliveira et al. 2011; Meng 2009). Most cities lack formal regulation, but ironically, comprehensive public regulation (standards) and the associated bureaucracy can also hinder green innovation (Karvonen 2010). For all cities, especially those with weak local government (Bayá Laffite 2009), the challenge is how to work with other stakeholders and communities with strong local knowledge of ecosystems and their uses (D'Souza and Nagendra 2011).

Box 27.5 Challenges to Effective Urban Ecosystem Management That Emphasize the Importance of Cooperation (Adapted from Sendstad 2012)

Coordinating all the actors and tasks necessary to respond to fragmented, heterogeneous and dynamic ecosystems in cities involves significant cooperation. Partnership is a cornerstone of urban ecosystem integrity as:

1. Responsibility for ecosystems is typically shared between government, traditional authorities, major public utilities and other agencies. In other words, cities do not themselves have all the powers needed for the task.
2. Cities do not always have the political commitment or fiscal and institutional capacity to govern ecosystems, even if they have the mandate.
3. Different municipal departments may have conflicting priorities even on the same ecosystems and invariably there are tensions about priorities.
4. Lack of communication between relevant public and private actors involved in management across the urban landscape may hinder a coordinated approach, both within and between adjacent green areas.
5. Lack of regional coordination between adjacent municipalities with planning authority may be a barrier.
6. Cities may depend on ecosystem services, which for a large part, are provided by ecosystems beyond their jurisdiction and control.
7. Lack of regional coordination may hinder management due to conflicts between administrative units or conflicts may hinder regional coordination.
8. If individual cities or city regions implement efficient policies this may have a limited global effect if others do not.
9. Insufficient public budgets for protection, maintenance and enhancement of ecosystem services has led to governments transferring management responsibility to private actors, including volunteers or the private sector in public private partnerships
10. Voluntary/non-governmental organizations have mixed attitudes to working with government but civil society is often involved (directly or indirectly) in urban ecosystem management; this makes civil society a critical partner.

Sources: Puppim de Oliveira et al. 2011; Wekerle et al. 2007; Wekerle and Abbruzzese 2010; Hutton 2011; Meng 2009; Blaine et al. 2006; Mendiondo 2008; Ernstson et al. 2010b; Barthel et al. 2005; Borgström et al. 2006; Karvonen 2010; Li et al. 2005b; D'Souza and Nagendra 2011; While et al. 2004; Hutton 2011; Schmidt and Morrison 2012; Hagerman 2007; Alonso and Heinen 2011; Antrobus 2011; Wilson and Hughes 2011; Rosol 2010; Pincetl 2010; D'Souza and Nagendra 2011.

27.3.2.9 Governance Failures in Urban Ecosystem and Biodiversity Management

Even where the various parties are able to work together to design policy and regulations there are typically major problems of government associated with enforcement (Li et al. 2005a; Bayá Laffite 2009; Xu et al. 2011). If regulations do have an effect, they may not stop fragmentation of habitats over time (Wekerle et al. 2007). In a study of loss of riparian habitat in Portland, Hillsboro and Oregon City, it was found that even though most development projects were hindered, a few larger projects permitted led to loss of ecological function (Ozawa and Yeakley 2007).

Even more common than governance failures through granting permission for dubious projects is the failure to monitor ecosystem integrity over time. One reason for this is the absence of robust scientific monitoring data, which makes it hard to implement regulations or develop a comprehensive knowledge base for management. This has, for example, been found to be a problem in China (Meng 2009). It is not just the absence of monitoring but also the failure to include all relevant variables of the complex systems and variables across all important scales that erodes the legitimacy of the administrative governance of ecosystems (Blaine et al. 2006; Ernstson et al. 2010b; Meng 2009; Wilson and Hughes 2011; Yli-Pelkonen et al. 2006). These weaknesses in governance capacity are not unique to ecosystem service management (Romero-Lankao and Dodman 2011) but they are especially serious in this domain for, as Baird argues, “unless we significantly reduce the lag time between occurrence of stress and management response we run the very real risk of irreplaceable loss of critical ecosystem functions” (2009, p. 9).

27.3.3 Urban Biodiversity and Ecosystem Services: Opportunities

The published scientific literature generates a rich set of insights into the opportunities for governing ecosystem services in an urban world, though careful interpretation of results is needed as opportunities include recommendations from case studies or more theoretical studies that have not necessary involved assessment of success in practice. Although cities have not traditionally been central to ecological management, it is clear that this is a rewarding scale of action and that targeting better ecosystem service governance in cities presents a grand opportunity to promote resilience. Drawing only from the published work, we have grouped lessons from innovative experiences in urban practice into four sub-sections: ecological management at the city scale; opportunities to expand conventional planning; innovations in urban economics and fiscal management; and the role of civil society. Table 27.1 (see the end of Sect. 27.3.3) summarizes some of the broad range of tools and approaches identified in the literature for governing urban biodiversity and

Table 27.1 Summary of some of the broad range of tools and approaches identified in the literature for governing urban biodiversity and ecosystem services

Tools and approaches	Description	Example	References	Scale of action
Urban design				
Larger green areas	Ensure protection of larger less fragmented green areas connected to other green areas in the city as a source of biodiversity and habitat for less disturbance tolerant species, e.g., green belts or larger urban parks	Bogor, London, Mumbai, Seoul, Stockholm	Bolund and Hunhammar (1999), Barthel et al. (2005), Bengston and Youn (2006), Zérah (2006), Colding (2007), Borgström (2009), Arifin and Nakagoshi (2011)	Local-Regional
Bioswales	Street design that hinders surface runoff		Grim et al. (2008); Karvonen (2010)	Local
Pervious paving	Street design that hinders surface runoff		Karvonen (2010)	Local
Green roofs and other forms of vertical greening	Provide habitat, insulate, and reduce urban heat island effect and energy use related to heating/cooling of houses due to its insulation effect		Li et al. (2005a), Oberndorfer et al. (2007), Carter and Fowler (2008), Dvorak and Volder (2010), Karvonen (2010), Xu et al. (2011)	Local
Urban agriculture	Community gardens for poverty reduction; enhancing food security, nutrition and economic status. Urban gardens can also be a food security mechanism, be important learning arenas and reduce the ecological footprint of cities	Bangkok, Badulla, Matale, Moratuwa, Matara, Rosario, Seattle	Alfsen-Norodom et al. (2004), Lieberherr-Gardiol (2008), Seymoar et al. (2010)	Local-Regional-Global
Regulation of land use				
Zoning	Zoning may allow a city to prioritize areas for different purposes with varying building densities and regulations of human activity, ensure protection of areas valuable to ES provision, and plan their linkages	Vancouver, Kyoto	Borgström et al. (2006), Hutton (2011), Morimoto (2011)	Local-Regional

(continued)

Table 27.1 (continued)

Tools and approaches	Description	Example	References	Scale of action
Higher level policy	National/Regional law limiting urban expansion into valuable green areas	Tampere	Asikainen and Jokinen (2009)	Local-Regional
Baseline requirements of private management	Set baseline requirements for management of privately owned land, like incorporating tree planting and maintenance in building regulations. This may include incentive measures for added efforts		Harman and Low Choi (2011)	Local
Ensuring access/property rights	For private persons to invest in structures supporting ES, like trees, it is important to ensure long-term access/property rights in, e.g., urban gardening		Barthel et al. (2010)	Local
Evidence-based management/planning	Legal tools can be used to require that planning should be based on available ecological knowledge, and include the knowledge and opinions of participants		Yli-Pelkonen et al. (2006)	Local-Regional
Planning tools				
Satellite images	Satellite images covering larger areas were combined with high-resolution images to inform management	Charlotte, Salem	Moll (2005)	Local-Regional
Indicators monitoring ecosystem change	Indicators on different levels to set targets and measure and monitor ecosystem state over time	Gainesville	Li et al. (2009), Dobs et al. (2010)	Local-Regional
Biotope Area Ratio	Biotope Area Ratio (BAR) to map surface types	Berlin and Seoul	Lakes and Kim (2012)	Local-Regional
Scenario development	Development of scenarios to inform planning and stakeholder processes, e.g., using multi-criteria evaluation		Mendiondo (2008), Mitsova et al. (2011)	Local-Regional–Global

Ecological footprint analysis	Ecological footprints can track progress, provide early warning, support setting targets and drive positive policy change		Wackernagel et al. (2006)	Global
Knowledge management systems	Regional ecological knowledge database that are regularly updated to inform planning	South East Queensland	Peterson et al. (2007)	Regional
Economic instruments and valuation tools				
Payment for ecosystem services/environmental management	Monetary tools can be applied to enhance ES through city dwellers paying for ecosystem management they benefit from, and adding to regulatory frameworks and incentive mechanisms connecting users and managers. It can also be useful for raising awareness of the value of conserving ecosystems		Boyer and Polasky (2004), Gutman (2007), Sarker et al. (2008), Xu et al. (2011)	Local-Regional-Global
Non-monetary valuation tools	Non-monetary evaluation may use indicators to set targets and monitor change in ecosystem function over time and how it relates to human welfare		Dobbs et al. (2010)	Local-Regional-Global
Tax measures	Property tax reduction in exchange for commitment to protect important habitat on people's properties, incentives for investing in green innovation or compensating land owners for restricted development rights		Bengston and Youn (2006), Carter and Fowler (2008), Alonso and Heninen (2011)	Local
Public funding/awards	Seed funding to support establishment of civil society initiatives, e.g., efforts targeting communities with lower access to ES. Public funding or awards can also support dispersal of best practices and continued effort	Bankok, Badulla, Matale, Moratuwa, Moratuwa and Matara, Seattle	Seymoar et al. (2010), Karvonen (2010), Warren et al. (2011), Wilson and Hughes (2011)	Local-Regional

(continued)

Table 27.1 (continued)

Tools and approaches	Description	Example	References	Scale of action
Green budgets	Having a separate budget line for urban greening		Seymoar et al. 2010	Local-regional
Green procurement	Public institutions having green procurement policy, e.g., hospital buying organic food. Such projects can contribute to general awareness raising	Vienna	Lieberherr-Gardiol 2008	Local-regional-Global
Management principles				
Connectivity	Managing/planning a city-wide network of connected green areas of different sizes connecting the city to the wider landscape (green infrastructure). This entails including green areas with varying characteristics, management and ownership		Bolund and Hunhammar (1999), Li et al. (2005a), Colding (2007), Yue et al. (2009), Xu et al. (2011)	Local-Regional
Diversity	Management of a spectrum of connected habitats to achieve high levels of biodiversity		Barthel et al. (2005), Jim and Chen (2008b), Colding (2007)	Local-Regional
Native species	Support native species adapted to the local environment	Bangkok, and in Badulla, Bogor, Matale, Moratuwa, Moratuwa, Matara.	Seymoar et al. (2010), Arifin and Nakagoshi (2011), Puppim de Oliveira et al. (2011)	Local-Regional
Systems thinking	Holistic management, including control of all pollutants and general dynamics of the ecosystem		Meng (2009) (focus on watersheds)	Local-Regional
Change	Focus on nature management as an ongoing process where one recognizes nature as ever changing and dynamic across scales and adapt institutions and plans as a result		Peterson et al. (2007), Asikainen and Jokinen (2009)	Local-Regional

Experimentation	Experimentation to foster learning and innovation: Pilot projects can over time be scaled up experiments to inform policy and create awareness, also across cities. Model cities can support others	E.g. Dongtan (China)	Borgström et al. (2006 and references therein), Economy (2006), Wackernagel et al. (2006); Lieberherr-Gardiol (2008), Mendiondo (2008), Baptista 2010	Local-Regional-Global
Different kinds of knowledge	Be open for the value of different kinds of knowledge, both scientific and non-scientific		Borgström et al. (2006 and references therein); Asikainen and Jokinen (2009), D'Souza and Nagendra (2011), Evans (2011 and references therein)	Local-Regional-Global
Context	Adapting biodiversity policies to the local context, considering place bound issues of equity and multiple social and ecological factors of relevance	Malmo	Elander et al. (2005), Quastel (2009)	Local
Leadership	Foster leadership, e.g., through awards or trainings targeted to individuals, communities or cities	National Conference of Mayors (USA), Bangkok, Badulla, Matale, Moratuwa and Matara. Review of urban policy England	Pincetl (2010), Seymoar et al. (2010), Wilson and Hughes (2011)	Local-Regional-Global
Synergies	Link ecosystem services to other benefits enhancing human welfare and view ES as an integrated part of urban functions	Bangkok and in Badulla, Matale and Moratuwa, Matara, Manchester, Seattle	Elander et al. (2005), Seymoar et al. (2010), Karvonen (2010), Antrobus (2011)	Local-Regional-Global
Mainstream	Mainstream plans for ES governance across other relevant policies	South East Queensland	Peterson et al. (2007)	Local-Regional-Global

(continued)

Table 27.1 (continued)

Tools and approaches	Description	Example	References	Scale of action
Cooperation and learning	Facilitate networks of people involved in urban sustainability across actors on different levels – departments, private sector and civil society to enhance learning and formal support for ES governance.	Bangkok, Badulla, Chicago, Matale, Moratuwa, Matara, Seattle, New York, and South East Queensland	Bayá Laffite (2009); Young (2010), Seymoar et al. (2010), Karvonen (2010), Schmidt and Morrison (2012)	Local-Regional-Global
Reconcile urban habitats with their natural analogues			Lundholm and Richardson (2010)	Local

Adapted from Sendstad (2012)

ecosystem services. Other useful tools and instruments for better governing biodiversity through sector interventions can be found, for example, in Chap. 11 on monetary evaluation and payment for ecosystem services.

27.3.3.1 Bringing Ecological Management to the City – Principles and Approaches

Creating citywide networks of connected green areas, including water bodies and coastal zones to support species movement, brings conventional ecological management to the urban scale and expands the traditional scope of urban government (Bolund and Hunhammar 1999; Yue et al. 2009). These networks, sometimes referred to as green infrastructure (cf. Gill et al. 2007; Antrobus 2011; Yu et al. 2011), connect the city to the wider landscape, with gradients or distinct zones with different degrees of human use (e.g., Li et al. 2005a; Borgström 2009). Some urban ecological studies encourage management of a spectrum of habitats and a patchy landscape to achieve high levels of biodiversity (Barthel et al. 2005; Jim and Chen 2008b) and to support native species adapted to the local environment, within this structure (Arifin and Nakagoshi 2011; Puppim de Oliveira et al. 2011). Establishing extended protected areas or green belts within the urban limits (as in Mumbai, e.g., Zérah 2006) ensures ecological connectivity and also creates opportunities for recreation and food security (Bolund and Hunhammar 1999; Borgström 2009; Barthel et al. 2005). Larger green areas can – if well maintained, appropriately protected, and connected to a green area network – provide habitat for species sensitive to disturbance and form the backbone of a bigger green infrastructure (Colding 2007; Borgström 2009; Jim and Chen 2008b). For example, the Bogor botanical garden (97 ha) in Indonesia has a rich variety of species and habitats and is important for local biodiversity (Arifin and Nakagoshi 2011). Some species depend on larger unfragmented areas, and typical urban parks may be too small to maintain viable plant and animal populations (Bolund and Hunhammar 1999; Borgström 2009).

In planning and designing urban areas Colding (2007) recommends striving for clustering of different types of urban green patches, both public and privately owned, to increase habitat connectivity across the landscape, complement habitat functions, and nurture key ecosystem processes essential for the support of biodiversity. The inclusion of private or common areas can also make the effects on ecosystem services from cuts in public spending on green areas less severe (Colding 2007), while areas under informal or traditional management can contribute to ecological integrity (cf. Andersson et al. 2007) or even be incorporated into the design of new eco-cities (Arifin and Nakagoshi 2011).

Open space management is not the only ecological practice now undertaken in cities. Restoration or protection policies targeting keystone species can support a number of additional species (Barthel et al. 2005 and references therein). It is often challenging to enhance green areas in cities that are already densely covered by buildings and infrastructure. Access to ecosystems tends to decline with building

Box 27.6 Examples of Urban Ecological Restoration (Adapted from Sendstad 2012)

Several studies address restoration and related opportunities, in particular related to networks of green areas (Li et al. 2005a; La Greca et al. 2011), parks and forests (Li et al. 2005a; Perkins 2009; Xu et al. 2011), grasslands (Xu et al. 2011), wetlands (Jansson and Colding 2007; Tong et al. 2007; Xu et al. 2011), brown fields (Franz et al. 2008), estuaries (Weinstein and Reed 2005), rivers (Li et al. 2005a; Tong et al. 2007), creeks (Karvonen 2010) and watersheds (Mendiondo 2008; Karvonen 2010). Such restoration projects can include innovative experimental approaches to restore ecosystems services, like stormwater management in streets and using ecorevelatory design (Karvonen 2010). It is highlighted as crucial in restoration efforts to identify the problem causing degradation, desired and feasible outcomes to be monitored, and the tolerance of the system to deal with disturbance (Mendiondo 2008). It can be useful to have a good understanding of pre-urban landscape characteristics, like vernal pools and grasslands, to inform restoration efforts and consider if such features could be obtained under urban conditions (Wolch 2007). When reconstructing connectivity it is also important to consider the habitat requirements of relevant species and how each of them can move in the wider landscape (Wolch 2007), and thus how different green areas can complement each other in terms of habitat function (Colding 2007). Also, non-traditional features of green areas, like golf courses, can be valuable in this effort, as they representing an opportunity for management to align conservation, restoration and recreation and support critical ecosystem service functions like pollination (Colding and Folke 2009).

density, but in a study of five UK cities, Tratalos et al. (2007) found variation in effects of density, offering hope for existing built up areas.

In cities having degraded ecosystems, restoration may be the most appropriate solution to ensure access to ecosystem services (Seabrook et al. 2011, p. 409) (see Box 27.6). There is a much more detailed discussion of the technical challenges of urban ecological restoration in Chap. 31.

Where it is not possible to restore and sustain urban ecosystems in line with that of a pre-existing state (due, for example, to irreversible changes and disturbance), some studies argue that one should rather aim for a stable supply of critical ecosystem services and conservation of species that are adapted to human presence (Weinstein and Reed 2005; Weinstein 2008), or reinvent urban landscapes recognizing novel ecosystem features (Seabrook et al. 2011). A more recent approach to enhancement of urban ecosystems is reconciliation ecology, based on an assumption that urban landscapes are unique and thus require a different approach compared to more traditional endeavors (Dearborn and Kark 2009). The approach aims to reconcile urban habitats with their natural analogues, e.g., modifying walls to support climbing vegetation, preparing nesting places for predatory birds on high rise buildings, or building green walls and roofs with substrates supporting different species of plants and arthropods (Lundholm and Richardson 2010). This kind of green innovation can also supplement more traditional restoration efforts by, e.g., enhancing connectivity and habitat diversity in the urban landscape.

27.3.3.2 The Ecological Redeployment of Traditional Planning and Management Tools

Well-established cities have at their disposal a huge array of conventional urban planning tools and instruments, including regulation and zoning. Numerous studies highlight the importance of strong legal protection to avoid ecosystem degradation and maintain or enhance various ecosystem services (Borgström 2009; Wang et al. 2009; Huang et al. 2011; Morimoto 2011; Xu et al. 2011). There are several approaches to regulating areas of importance for ES, and managing the city as a part of the surrounding landscape (Li et al. 2005a; Xu et al. 2011), like smart growth policies and zoning (Hutton 2011). A number of case studies, in particular from Chinese cities, present detailed suggestions for urban planning with a focus on enhancing green infrastructure and limiting encroachment (e.g., Xu et al. 2011; Liu et al. 2012). Zoning may allow a city to prioritize areas for different purposes with varying building densities and regulations of human activity, ensure the protection of areas valuable to ecosystem services provision, and plan their linkages (Lieberherr-Gardiol 2008; Weinstein 2008; Asikainen and Jokinen 2009; Hutton 2011; Yong et al. 2010). Rather than aiming to separate social and ecological aims in distinct zones, Borgström (2009) suggest integrating them in the urban landscape matrix with the aim of having connected green areas to conserve local biodiversity values, planning to maintain ecosystem services both at temporal and spatial scales, and also prioritizing neighborhoods with a lack of access to ecosystem services. The importance of applying such a multifunctional landscape perspective has been emphasized in several studies (Bolund and Hunhammar 1999; Lundy and Wade 2011), and Hagerman (2007) presents a common strategy aimed at increasing access to green space and general quality of life (liveability) in the urban center in order to reduce sprawl.

Another regulatory approach to enhance ecosystem services is to set targets for minimum green coverage across the city (Arifin and Nakagoshi 2011) and riparian area next to rivers for habitat protection; this enhances connectivity and flood protection (Ozawa and Yeakley 2007). The potential value of traditionally, privately or commonly owned land in cities could be enhanced by incorporating these parcels into an ecological zoning or amending their regulation. Authorities may set baseline requirements for management of privately owned land (Harman and Low

Choy 2011), like incorporating tree planting and maintenance in building regulations (Davies et al. 2011) or include incentives for additional actions (Harman and Low Choy 2011). In addition to regulation of non-state land, public authorities can sometimes choose to use established planning codes to acquire private land for safeguarding ecosystem services for the public good (Blaine et al. 2006; Vejre et al. 2010; Morimoto 2011). Where local planning codes are not strong enough, national and global treaties may also influence land use within and outside the cities' jurisdiction (Lucero and Tarlock 2003; Asikainen and Jokinen 2009).

Outside of regulation and zoning, planning tools being used by ecologists are mainly related to mapping and visualizing information on land characteristics and land use; numerous approaches exist as to how this can be done. There are disagreements as to which approach/tool is more appropriate, e.g., how detailed a level of qualitative/quantitative data is required. Commonly applied tools include, e.g., remote sensing via satellite images for detailed management of green areas (Moll 2005), linking land use to ecosystem features through a categorization system (Liu et al. 2012), and developing sets of indicators on different levels to facilitate long-term monitoring of ES (Li et al. 2009). The traditional planning rubric of mapping and monitoring is now being extended with ecological footprint analysis. This comprehensive tool is being applied to support cities in assessing their global impact, potential ecological deficit, and thus vulnerability; setting targets; and tracking progress. Some cities and urban communities have started to test this approach (e.g., Cardiff, London) (Luck et al. 2001; Wackernagel et al. 2006).

27.3.3.3 Economic Instruments and Valuation Tools

There is an increased focus on financial tools in urban management generally and ecosystem service interventions in particular. The economic instruments include monetary and non-monetary valuation tools for assessing and prioritizing urban interventions. Monetary tools are being applied to enhance ecosystem integrity through city dwellers paying for land management protection, maintenance or enhancement of ecosystem service quality outside city boundaries (Gutman 2007; Xu et al. 2011); this adds to regulatory frameworks and incentive mechanisms connecting users and managers (Boyer and Polasky 2004; Sarker et al. 2008). In a survey among urban Australians, Zander et al. (2010) found that residents were often willing to pay for conservation of rivers upstream. Non-monetary evaluations utilize indicators to set targets and monitor change in ecosystem function over time and assess how the ecological health of a city relates to human welfare (Dobbs et al. 2010).

There are some warnings regarding the limits to monetary or non-monetary valuations' ability to adjudicate decisions on all services across spatial and temporal scales, and authors warn that economic valuations that raise awareness among decision makers and others about the importance of such services may not always enhance protection (Boyer and Polasky 2004; Hougner et al. 2006). Ecological accounting can potentially help avoid undervaluation of ecosystems in planning, and support more appropriate compensation mechanisms (Li et al. 2005a; Bengston and Youn 2006; Wang et al. 2009; Gaodi et al. 2010).

Several articles argue that taxes should be used to ensure public interest in multiple ecosystem services (Li et al. 2005b), including property tax reduction in exchange for commitment to protect and manage important habitat on people's properties (Alonso and Heinen 2011), or compensating land owners for restricted development rights (Bengston and Youn 2006). One may also use tax and other fiscal incentives for investing in green innovations, such as incentivizing green roofs for limiting stormwater runoff (Carter and Fowler 2008). Public budgets can also be used to provide seed funding to support establishment of civil society initiatives, e.g., efforts targeting communities with lower access to ecosystem services (Warren et al. 2011; Wilson and Hughes 2011).

27.3.3.4 Civil Society – A Source of Legitimacy, Knowledge and Management Capacity

Civil society associations have an important role in ecosystem governance, as groups voice concern for threatened ecosystem services, or trigger political action to avoid environmental degradation in general (e.g., While et al. 2004; Barthel et al. 2005, 2010; Bengston and Youn 2006; Peterson et al. 2007; Grimm et al. 2008; Asikainen and Jokinen 2009; Wekerle and Abbruzzese 2010; Ernstson et al. 2008, 2010b; Arifin and Nakagoshi 2011; Morimoto 2011). Civil society initiatives reportedly built networks and mobilized action to influence decision makers, which compensated for fragmented governance in Toronto, Canada (Wekerle and Abbruzzese 2010). Other studies suggest that the development of NGOs could contribute to increasing awareness among citizens, enhancing green space management effort, and generating a more structured contact between citizens and public administration (Jim and Chen 2006). For further discussion on urban landscapes as learning arenas and sources of civil society stewardship for biodiversity and ecosystem services, see Chap. 30.

Participatory governance creates a foundation for collective action through creating shared visions/scenarios (Peterson et al. 2007; Seymoar et al. 2010). Government agencies/local authorities have increased their capacity by cooperating with professional civil society organizations in activities like the Los Angeles, California mass tree plantings (Pincetl 2010). Involvement and education of citizens can also contribute to environmental monitoring (Dearborn and Kark 2009). Adaptive co-management strategies in Stockholm focus on urban gardens and parks; these strategies highlight how user groups can be recognized as sources of local ecological knowledge and management capacity to support ecological processes and respond to change (e.g., Barthel et al. 2005; Colding et al. 2006; Andersson et al. 2007). Participatory management endeavors can also enhance other social benefits. Perkins (2009) showed how urban greening programs in poor neighborhoods using volunteers contributed to both enhanced ecosystems and increased ecological awareness, and gave people commonly excluded from the job market valuable work experience. Participatory processes in governance of ecosystem services are characterized by a range of different interests and Elander et al. (2005) recommend identifying different views and potential conflicts early in planning processes as a first step to deal with this. Transparent utilization of land-use scenarios is one policy tool that can be applied to involve stakeholders, enhance trust and public debate, and potentially contribute to dealing with land use conflicts (Mitsova et al. 2011).

A central opportunity of greater civil society engagement in the ecosystem service agenda is the fostering of ecological citizenship – a new set of values reframing the relationship between people and nature, reframing rights and obligations, and supporting changed behavior (Moll 2005; Li et al. 2005b; Jim and Chen 2006; Hagerman 2007; Wolch 2007; Karvonen 2010). Healthy ecosystems are seen as shaping local identity, providing a sense of place and fostering deeper insight into nature (Yli-Pelkonen et al. 2006). Ecological citizenship may also have a wider scope, as experienced in Seattle, Washington, where some have been inspired by bioregionalism and the abundant nature in the Pacific Northwest, thus leading to an increased desire to live in balance with the natural surroundings (Karvonen 2010). In Portland, Oregon, restoration of a river was related to a regional identity – 'people of the Salmon' (Karvonen 2010, p. 173). It has also been suggested that ecological citizenship may have a broader application, as captured in the following quote: "With respect to the environment, the urban ecological citizen is one whose rights include environmental justice but whose duties and obligations are defined by their ecological footprint: our production and consumption habits" (Wolch 2007, p. 379).

27.4 Concluding Discussion

The *Cities and Biodiversity Outlook – Action and Policy* together with this volume of chapters that reflect the scientific foundation of the CBO project underline the significant shift in attention to urban biodiversity and ecosystem services in global policy forums and urban governance structures that operate at the national and local scales. Key purposes of this chapter have been to situate the emerging field of urban biodiversity and ecosystem governance and to provide the first comprehensive global synthesis of researched scientific material on the governance of biodiversity and ecosystem services. The absence of such a synthesis review represented a significant gap in knowledge that this chapter has begun to address. As was shown in Fig. 27.2, it really is only over the past 10 years that significant attention has begun to be paid to the governance of biodiversity and ecosystem services in urban settings in the scientific literature, no doubt a more comprehensive mining of the grey literature would draw attention to other governance trends and it would certainly put the spotlight on other less-affluent regions of the world.

With respect to the policy agenda, the synthesis review of the literature presented here confirms that cities have a critical role to play in the governance of biodiversity and ecosystem services. Whilst the actors that typically lead governance of urban biodiversity and ecosystem services are typically drawn from across the state, in particular government-based planning and environmental management actors, this is not always the case, and it is not the case at all in places with very weak states. Very well-capacitated governments are able to engage with and work extensively with civil society, but in the absence of strong local/regional/national management, other global stakeholders/institutions and local organizations are left to drive much of the biodiversity and ecosystem service agenda. In places where there is no or limited urban governance capacity, residents carry the brunt, through mostly informal micro-solutions. The governance of urban biodiversity and ecosystem services will only be successful with collaborative, cross-scale efforts that better prioritize the value of biodiversity and ecosystem services through urban governance. Good management of the urban landscape for biodiversity can only be achieved with the collaboration of multiple jurisdictions and a large number of public and private actors. These actors need to come from all levels of decision-making, from national, sub-national, and local governments to UN and other international organizations, citizen groups, scientists, NGOs, and businesses both large and small.

The synthesis review shows there is already significant scientific knowledge to inform action (see Table 27.1 for a summary of tools and approaches identified in the literature for governing urban biodiversity and ecosystem services). However, it also reveals the limitations of the current knowledge base given the unevenness of the geographical coverage of research published in English in scientific journals. Notably, the current scientific literature pays least attention to those areas in the Global South with the highest rates of urbanization and that are the most vulnerable areas in terms of their exposure to risk and their capacity to respond to future challenges. This unevenness in the knowledge base presents a significant challenge to the global research community. Subsequent efforts must not only engage with the non-English scientific literature and monographs, but also transparently and robustly engage with the grey literature. A key opportunity in tapping into the grey literature is to access more examples of initiatives to govern urban biodiversity and ecosystem services that have been assessed to some degree. This is a useful complement to the scientific literature on the governance of biodiversity and ecosystem services, which is mainly dominated by theoretical and general case studies overviews rather than robust evaluations of the success or otherwise of governance initiatives in practice.

For the battle for sustainability to be won, biodiversity and ecosystem services in and of cities must be better governed. There are significant challenges, but already many solutions are being successfully put in practice in cities. Addressing inequities in the impacts on biodiversity and ecosystem services generated by cities, the impacts endured by cities and the uneven capacities of cities to govern must be a priority.

Appendix A

Geographic coverage (region, country, city/city-region) of scientific literature review drawn on in the synthesis that informed this chapter (Adapted from Sendstad 2012).

Region	Country	City/city-region	Reference
Asia	China	Beijing	Li et al. (2005a, 2008), Yue et al. (2009), Gaodi et al. (2010), Xu et al. (2011), Yang et al. (2011), Yu et al. (2011)
	China	Changshu	Li et al. (2010a)
	China	Foshan	Yong et al. (2010)
	China	Guangzhou	Jim and Chen (2006, 2008a), Guo et al. (2007), Su and Fath (2012)
	China	Jining City	Li et al. (2009)
	China	Rizhao City,	Wang et al. (2009)
	China	Shenzhen	Li et al. (2010b)
	China	Shiyan City	Dong et al. (2011)
	China	Taiyuan City	Liu et al. (2012)
	China	Urban forest in China	Li et al. (2005b)
	China	Water control in China	Meng (2009)
	China	Wenzhou	Tong et al. (2007)
	China	Xiamen	Fang et al. (2006)
	India	Auroville	Kapoor (2006)
	India	Bangalore	D'Souza and Nagendra (2011)
	India	Mumbai	Zérah (2006)
	Indonesia	Bogor/Jakarta and Sentul	Arifin and Nakagoshi (2011)
	Japan	Kyoto	Morimoto (2011)
	Japan	Tokyo	Gadda and Gasparatos (2009)
	Republic of Korea	Seoul	Bengston and Youn (2006), Lakes and Kim (2012)
	Sri-Lanka/ Thailand		Seymoar et al. (2010)
	Taiwan	Taipei	Jim and Chen (2008b), Huang et al. (2011)
Europa	Austria	Vienna	Lieberherr-Gardiol (2008)
	Belgium	Flanders (region with urban centres)	Bomans et al. (2010)
	Denmark	Copenhagen	Vejre et al. (2010)
	Finland	Lahti	Niemelä et al. (2010)
	Finland	Tampere	Asikainen and Jokinen (2009)
	Finland	Vantaa (in the Helsinki metropolitan area)	Yli-Pelkonen et al. (2006)

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Region	Country	City/city-region	Reference
	Germany	Berlin	Rosol (2010), Lakes and Kim (2012)
	Germany	Leipzig–Halle	Kroll et al. (2012)
	Germany	Ruhr	Franz et al. (2008)
	Italy	Catania	La Greca et al. (2011)
	Sweden	Stockholm	Bolund and Hunhammar (1999), Jansson and Nohrstedt (2001), Barthel et al. (2005, 2010), Borgström et al. (2006), Colding et al. (2006), Hougner et al. (2006), Andersson et al. (2007), Jansson and Colding (2007), Ahrné et al. (2009), Ernstson et al. (2010a, b)
	Sweden	Stockholm/Göteborg/ Malmö/Uppsala/ Linköping/Örebro	Elander et al. (2005), Sandström et al. (2006)
	Sweden	Studied 1869 nature reserves in Southern Sweden, considering urbanization	Borgström (2009)
	Switzerland	Zürich	Schulz and Schläpfer (2009)
	United Kingdom	Cambridge and Waveney	Jonas and Gibbs (2003)
	United Kingdom	Edinburgh, Glasgow, Leicester, Oxford and Sheffield	Tratalos et al. (2007)
	United Kingdom	National measures towards urban green space in England	Wilson and Hughes (2011)
	United Kingdom	Leicester	Davies et al. (2011)
	United Kingdom	Manchester	Gill et al. (2008), Antrobus (2011)
	United Kingdom	Manchester and Leeds	While et al. (2004)
North-America	Canada	Vancouver	Lieberherr-Gardiol (2008), Quastel (2009), Hutton (2011)
	Canada	Toronto	Wekerle et al. (2007), Wekerle and Abbruzzese (2009)
	USA	Akron and Cleveland	Yadav et al. (2012)
	USA	Boston and Springfield	Warren et al. (2011)
	USA	Charlotte (North Carolina), Roanoke (Virginia) and Salem (Oregon)	Moll (2005)

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Region	Country	City/city-region	Reference
	USA	Chicago	Young (2010)
	USA	Columbus	Styers et al. (2010)
	USA	Detroit	Nassauer et al. (2009)
	USA	Gainesville	Dobbs et al. (2010)
	USA	Illinois	Jaffe (2010)
	USA	Los-Angeles	Wolch (2007), Pincetl (2010)
	USA	Miami-Dade and Gainesville	Escobedo et al. (2010)
	USA	Milwaukee, Wisconsin	Perkins (2009), Perkins (2010)
	USA	New Mexico	Lucero and Tarlock (2003)
	USA	New Orleans, Phoenix	Ernstson et al. (2010a)
	USA	New York	Alfsen-Norodom et al. (2004), Blaine et al. (2006)
	USA	North Carolina	Bendor and Doyle (2010)
	USA	Portland, Hillsboro and Oregon City	Ozawa and Yeakley (2007)
	USA	Portland, Oregon	Hagerman (2006)
	USA	Seattle	Robinson (2008), Karvonen (2010)
	USA	274 metropolitan areas	McDonald et al. (2010)
South-America	Argentina	Rosario	Lieberherr-Gardiol (2008)
	Brazil	Buenos Aires and Sao Paulo	Bayá Laffite (2009)
	Brazil	Curitiba	Lieberherr-Gardiol (2008)
	Brazil	Sao Paulo	Mendiondo (2008)
Australia, New Zealand	Australia	Auckland	Grimm et al. (2008)
	Australia	Melbourne	Grimm et al. (2008)
	Australia	South East Queensland	Peterson et al. (2007), Sarker et al. (2008), Harman and Low Choy (2011), Schmidt and Morrison (2012)
	Australia	Sydney	Merson et al. (2010)
Africa	South Africa	Cape Town	Ernstson et al. (2010b)
	South Africa	Port Alfred, Grahamstown and Somerset East	Kuruneri-Chitepo and Shackleton (2011)

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Chapter 28
Regional Assessment of Latin America: Rapid Urban Development and Social Economic Inequity Threaten Biodiversity Hotspots

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Abstract The relationship between cities and biodiversity is extremely complex in Latin America. The region is simultaneously the world's most urbanized, has some of the world's largest social and economic inequities, and hosts some of the world's most biodiversity-rich ecosystems, including several biodiversity hotspots. As cities in Latin America are expected to continue to expand, partly on areas harboring valuable biodiversity hotspots, there is an urgent need to understand how biodiversity and ecosystem services interplay in and around cities. This assessment aims to describe urbanization trends in Latin America and the related impacts on urban biodiversity and ecosystem services, complementing the general framework with shorter case studies of four cities around the region. It also explores the potential for city planning to provide support for biodiversity and ecosystem services. The study found that cities in Latin America exhibit extreme social and economic differences, which generate a complex mosaic of urban settlement structures and ecosystem management systems. Low-income neighborhoods are typically either interspersed with the local ecosystems in peri-urban areas or completely lacking green spaces. High-income neighborhoods have a higher concentration of green areas, but are usually dominated by non-native species. It also found that conservation of ecosystems and biodiversity, and ecosystem services provisioning, are low priorities in urban planning; they are not acknowledged as key elements for the quality of life of the city inhabitants and human well-being. The knowledge base is also limited, as research on the consequences of rapid urbanization on biodiversity and ecosystem services in Latin America is poorly developed. However, initiatives to increase focus in urban planning on support for ecosystems are being taken and examples have been found of urban inhabitants actively promoting stewardship of urban greens.

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Keywords Biodiversity hotspots • Cities • Urbanization impacts • Peri-urban ecosystems • South America

Five Main Findings

- Latin America is increasingly urbanized but the urban population is marked by extreme social inequity, which impacts biodiversity and ecosystem services provisioning
- Urban sprawl encroaches on several global biodiversity hotspots
- Impacts on biodiversity occur in mega cities as well as in mid-size and small cities, the latter two are expected to grow the most in population and in size in the future.
- Research on urban ecology is severely limited, because of lack of funding and prioritization amongst supporting agencies
- Existing policies are insufficient to provide protection for ecosystems in and around cities, and new models of city sustainability need to be implemented.

28.1 Trends in Land-Use Change and Demography

The relationship between cities and biodiversity is extremely complex in Latin America (the 15 countries of South America and the Caribbean). The region is simultaneously the world's most urbanized, has some of the world's largest social and economic inequities, and hosts some of the world's most biodiversity-rich ecosystems including several biodiversity hotspots (Myers et al. 2000). In addition, many of the national economies in the region are based on unsustainable practices of natural resource exploitation. The practices reflect a lack of integration of environmental issues in land-use planning policies and development strategies, as well as low levels of governance and limited information on the affected ecosystems (Naylor 2009).

The main driver of land-use change in Latin America has traditionally been agriculture, but industrialization of agriculture has caused abandonment of poor soils (Grau and Aide 2008). Meanwhile, city expansion is now a significant contributor to land-use changes, and the number of cities in Latin America has grown sixfold in the past 50 years. This has resulted in large rural areas with low occupation, alternated with densely populated cities (Fig. 28.1).

More than 80 % of the population in Latin America lives in cities, and by 2050 the number is expected to reach 90 % (UN 2011). This has resulted in Latin America being the region with the highest proportion of urban inhabitants in the world. The majority of the urban areas were established between 1950 and 1990 as a result of

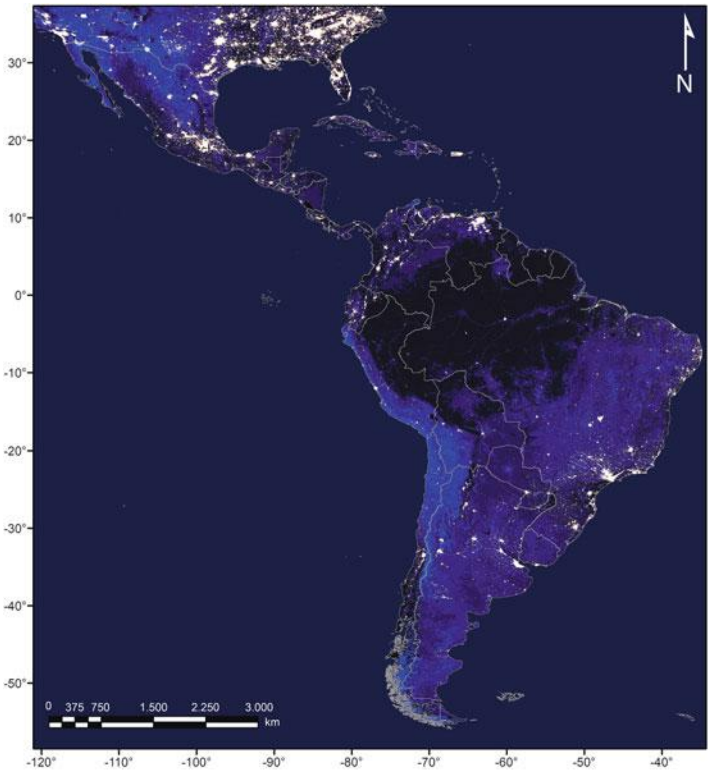

Fig. 28.1 Satellite image of Latin American city lights from April 2012, including national borders. The urban agglomerations contrast to the vast areas with low-population densities. (Image modified from ©NASA Earth Observatory image by Robert Simmon, using Suomi NPP VIIRS data provided courtesy of Chris Elvidge (NOAA National Geophysical Data Center). Suomi NPP is the result of a partnership between NASA, NOAA, and the Department of Defense. All Rights Reserved)

a rapid demographic increase, coupled with an intensive rural-urban migration (ONU-HABITAT 2012). Following a strong decrease in fecundity and an increase in life expectancy, the transition is slowing down. The annual growth rate is currently at around 1.15 % for Latin America, and no significant changes are expected in the near future. This stabilization in urban populations may support economic growth that can offer opportunities to address long-standing regional issues of poor household sanitation and low quality housing; moving forward from an agenda focused mostly in urban infrastructure into a more sustainable, greener development (McDonald and Marcotullio 2011).

As cities in Latin America are expected to continue to expand, partly on areas harboring valuable biodiversity hotspots, there is an urgent need to understand how biodiversity and ecosystem services interplay in and around cities. Increasing attention by planners in countries across Latin America on the importance of including ecological aspects in the planning processes, and their importance for human well-being, provides an opportunity to advance this understanding. However, critical knowledge gaps need to be bridged in order for urban planning and development to be designed so as to support local ecosystems. This assessment aims to describe urbanization trends in Latin America and the related impacts on biodiversity and ecosystem services in cities. It also explores the potential for city planning that focuses on support for biodiversity and ecosystem services, within the urban areas as well as in their surroundings.

28.2 Biodiversity Hotspots, Social Inequity, and Natural Hazards Shape Urban Ecosystems

The effects of urbanization on biodiversity are particularly serious in Latin America because cities are located in or around areas with high levels of species richness and endemism (Liu et al. 2003). The cities are expected to continue to expand, partly on areas harboring biodiversity hotspots (Chap. 1 and Fig. 1.1). The region contains six biodiversity hotspots; for example, the Cerrado Region in Brazil is the most extensive woodland savanna in South America and covers 21 % of the country. The Mesoamerican Forests stretches across most of Central America, is the world's third largest biodiversity hotspot, and fills an important function as a corridor for many Neotropical migrant bird species. The Tropical Andes runs through Venezuela, Chile, Argentina, Colombia, Ecuador, Peru, and Bolivia, and is described as the richest and most diverse region on Earth (Mittermeier et al. 2011). The Chilean Winter Rainfall-Valdivian Forest, covering the central-northern part of Chile and featuring Chile's Mediterranean ecosystem, harbors 50 % of all species of vascular plants in Chile, while also having the country's highest density of human settlements (Armesto et al. 2007; Underwood et al. 2009). In other areas of Latin America, rich coastal ecosystems and river deltas have been the centers for population settlements and urban growth, for example, in Argentina, Uruguay and Brazil.

One of the most conspicuous characteristics of Latin America is that urban populations exhibit extreme social and economic differences. More than 25 % of the urban inhabitants live in very poor settlements, while the richest 20 % earn almost 20 times more than the poorest 20 % (ONU-HABITAT 2012). The structures of inequity go beyond differences in income and housing standards, to also include an uneven distribution of green space availability and quality. Ecosystem differences associated with high-income areas versus low-income areas ultimately affect the ability of depauperate urban ecosystems in poor neighborhoods to provide ecosystem services essentials for human well-being (Barbosa et al. 2007; Reyes and Figueroa 2010).

Conversion of land to built-up urban environments affects ecosystem functions, which contributes increasing environmental vulnerability of new urban areas. For example, many financially poor communities establish informal settlements, often densely built, in vulnerable areas such as riparian corridors, coastal ecosystems, and steep hills. This unplanned development has shown to severely impact ecologically valuable and sensitive areas, for example by sewage discharge into watercourses, infill of wetlands for urbanization, and deforestation. At the same time, 60 % of the natural hazards in urban settings in Latin America are associated with climatic events (Zapata 2010). When services such as flood regulation and storm water retention decrease due to, for example, deforestation (Bradshaw et al. 2007), the effects of natural events can thus be hazardous, manifested, for example, in frequent land and mud slides in Chilean and Colombian cities (Flood Observatory 2012). The effects can be particularly serious certain years due to the natural cycles of changing climate and weather, manifested in El Niño years. Thus, functioning ecosystems play a vital role for resilient urban areas. Lack of data on the functions and values of local ecosystem services represents a main challenge for conservation in these areas. Unplanned urban sprawl may also increase conflicts between nature and humans in peri-urban ecosystems, for example by increasing the risk of wildfires and wildlife-human disease transmission in both directions.

At the other end of the spectrum, the planned areas where the wealthiest segments of the population live, increasingly mimic the low-density urban environments that are common in many places in the USA and other developed countries. These areas are signified by their large, commonly highly energy-demanding houses, large garden areas dominated by relatively few selected species, and a resulting urban sprawl that demands the areas' inhabitants to rely on private transportation. The land conversion to this type of urban areas commonly decreases the availability of natural and often highly valuable ecological habitats (Fig. 28.2). Moreover, the remaining, preferred vegetation provides only limited support for the native communities, as they typically include plants species that are considered aesthetically pleasing but are non-native. These species often become invasive, such as *Acacia* spp. and *Robinia pseudoacacia* (Pauchard et al. 2006). This is one of the main drivers of biotic homogenization across the region, and is a significant threat to biodiversity conservation. Furthermore, invasive animals such as feral dogs and cats, pigeons, rats, and house sparrows, often find such conditions adequate to expand their ranges, in detriment of native species (Sushinsky et al. 2013).

Invasive species are becoming dominant features of Latin American cities, to a large extent because of human interventions, which affects the capacities to produce ecosystem services. An increased understanding of the drivers of the species' establishment, how they impact local biodiversity, and their production of services or disservices to humans, is needed. Invasive trees, by replacing the native vegetation, can increase for example, the risks of fires in peri-urban areas and even cause health concerns because of their allergenic characteristics (Pauchard et al. 2006; Mardones et al. 2011). As cities act as propagule sources, invasive species can extend from the urban centers to natural habitats in the surrounding peri-urban areas (e.g., von der Lippe and Kowarik 2008).

Interestingly, cities can also support a rich biodiversity of native species, capable of withstanding the highly anthropogenic environments that cities represent. Studies

Fig. 28.2 Local people demanding protection of the local wetlands, an important component of the local ecosystems in Valdivia, Chile (Photo by and published with kind permission of ©Javiera Maira 2013. All Rights Reserved)

have shown that while biodiversity tends to decrease along a rural-urban gradient, some generalist native species do flourish in urban and peri-urban ecosystems (Reis et al. 2012). There is no consistent relationship between income and biodiversity. In some cases local communities in low-income areas have managed to reintroduce urban green spaces in their neighborhoods; in others, limited maintenance of vegetation and abandoned allotments have resulted in higher species richness, which include native plant and animal generalists (e.g., Rio de Janeiro and Valdivia, Chile); in some more affluent neighborhoods a growing trend promotes the replacement of non-native vegetation by native ornamental plants.

28.2.1 Identified Research Gaps and Implementation Challenges

Several researchers have highlighted the need for increased focus in research on biodiversity and ecosystem services in urban settings (Pauchard et al. 2006; Gaston 2010; Ortega-Álvarez and MacGregor-Fors 2011). However, despite a rapid increase of articles related to urban ecosystems in recent years, less than 2 % of them focus on urban wildlife and the impacts of cities on biodiversity. Moreover, of the studies of urban wildlife made over the last 40 years, only 3.7 % focused on Latin America.

In order to get a thorough understanding of existing research on urbanization and biodiversity in Latin America, with special attention to South America, a literature search was conducted using Web of Science. Key words were: urban ecology, biodiversity, urban, cities, ecosystem services, South America, and all possible combinations of these terms. The search was then deepened by focusing on authors often cited in the papers, keeping the search focused on urban biodiversity and ecosystem services, rather than for example solely on urban planning.

The findings corresponded to what many authors have previously stated (e.g., Gaston 2010); research on the consequences of rapid urbanization on biodiversity and ecosystem services in Latin America is poorly developed (see Textboxes 28.1, 28.2 and 28.3). Studies focus primarily on land-use change, where urbanization is one of the main drivers, alongside agriculture and forestry (Pauchard et al. 2006; Izquierdo et al. 2008; Rojas et al. 2013). When research has analyzed urban structures and urban morphology, attention has been paid fundamentally to social segregation and inequity (e.g., Ingram and Carroll 1981; Madaleno and Gurovich 2004; Krellenberg et al. 2011), and some on sustainable development (Kopfmuller et al. 2009). However, most of the studies focus only on large cities or megacities, and are restricted to study cases in Chile (Textbox 28.1), Argentina (Textbox 28.2), and the Atlantic Forest area.

Textbox 28.1 Nature and Urban Planning Tools in Chile

The critical issue for urban planning in Chilean cities is the lack of an adequate planning instrument to safeguard biodiversity and ecosystem services, so issues impacting biodiversity directly, such as urban sprawl or green space decrease, cannot be controlled properly through consistent planning tools. The reduction in wetland areas, especially, is a critical issue in growing cities such as Valdivia and Concepción, where these, are constantly in-filled for housing and road infrastructure development, contributing to the decrease in biodiversity and the provisioning of valuable ecosystem services such as flood regulation or recreation.

A clear regulatory framework for nature conservation in cities in Chile is lacking. The General Law on Urban Planning and Construction (LGUC), enacted in 1974 and still valid today, regulates residential and industrial uses, constructions, and the location of public facilities. The responsibility to nominate protected zones lies with the Public National System of Protected Areas (SNASPE; Pauchard and Villarroel 2002). The LGUC only considers those zones which the SNASPE has already awarded protection status as zones valuable enough to protect, leaving many urban or peri-urban areas unprotected.

Some other initiatives have been developed to respond to these planning and conservation problems. For example, a comprehensive landscape design

(continued)

Textbox 28.1 (continued)

strategy was developed in the 1960s for the Santiago Metropolitan Area, incorporating green space and infrastructure using an integrative approach (Pavez 2002). Later, the Environmental Impact Evaluation System (SEIA) was developed as a planning tool to support environmentally sustainable planning. However, in cities as Concepción, the SEIA was used to guide the proposed city expansion plan but the plan focused insufficiently on conserving the natural landscape, and was poorly adapted to the actual growth rate of the city. As a result, impervious surfaces in the city increased by more than 6,200 ha (2000–2010) since the approval of the metropolitan plan (2003), and 16 % of the former natural areas, such as wetlands and native forest, had been lost (Rojas et al. 2013).

Recently, triggered by the earthquake and tsunami on February 27, 2010, Sustainable Planning Programs (PREs) have been implemented by the national government. These plans recognize the role of nature for earthquake recovery, mostly as a buffer zone of forests and dunes along the coast. However, they fail to recognize other ecosystem services associated with nature and local culture. For example, urban wetlands in Valdivia and Concepción are useful for recreation and flooding protection but also for people's emotional recovery and resilience after natural hazards such as tsunami and earthquakes.

The future does, however, seems to be positive for the prospects for increased biodiversity support in Chilean cities, due to a new law that makes Strategic Environmental Assessments (SEAs) mandatory in urban municipal development plans. The law (20.417) was passed in 2010 to include environmental procedures which are progressively being recommended for Chilean urban planning. These should be included throughout the process and are a step towards an integrated planning that considers the interplay between the social, economic and ecological spheres to increase resilience of cities in Chile.

Textbox 28.2 Nature in the City: Some Trends in Argentina

In Argentina, 90 % of the population lives in cities (Ministerio de Planificación Federal, Inversión Pública y Servicios 2011). The increase in urban populations and expansion of city boundaries during the last decades of the twentieth century have created new challenges for the conservation of the local biodiversity, especially in the metropolitan area of Buenos Aires and big cities like Cordoba, Rosario, Mendoza, and San Miguel de Tucumán. In those urban settlements, which contain 50 % of Argentina's population, the natural

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Textbox 28.2 (continued)

landscape has become heavily modified, and as a result the local biodiversity is under high stress (Franceschi 1996; Formiga and Garriz 1999; Martínez 1992; Morello et al. 2000; Guerra 2005). As described by Morello et al. (2000), the invasion of exotic plants and animals, habitat changes due to climate, high use pressure, and vandalism can be observed. A participatory planning process may reduce the negative impacts of local populations on the ecosystems, and help to conserve or restore natural environments. This notion has inspired several scientific studies on urban ecology in Argentina (Faggi and Carretero in press). The studies focus on composition, structure and functions of urban ecosystems, and try to answer how individual plant and animal species and communities are affected by the growth of cities, including the underlying biotic and abiotic mechanisms, in order to identify vulnerable species and to develop effective measures for their conservation.

As a result, since the end of the twenty-first century, the awareness of the need for conservation, rehabilitation, and restoration of urban green spaces and biodiversity. Many cities in Argentina have launched programs on conservation of natural areas, often as initiatives taken by communities or NGOs has increased among both Argentinean city planners and concerned citizens. Consequently, several urban reserves (URs) have been implemented in and near city edges. URs are characterized by maintained natural and semi-natural ecosystems, a high degree of biodiversity, landscape heterogeneity, and the possibilities for recreation and environmental education. They are intended to act as counterpoints to the heavily human-dominated urban landscapes, and provide opportunities for functions not well served by current recreational parks (Perelman et al. 2012). URs have a significant value especially in the metropolitan area where, according to the census data of 2010, the ratio of public green park area per inhabitants reaches just 3 m^2 (Indec 2010), and the parks are unevenly distributed in the region. The URs have added many hectares of urban green areas to the cities where they are located, and at more than 15 m^2 of green areas per inhabitant well exceed the values recommended by the World Health Organization.

A recent example, in Buenos Aires, is the implementation of a new 18 ha urban reserve behind the University Campus, initiated in mid-December 2012. The reserve preserves part of the riparian ecosystem of the Rio del Plata estuary and connects to other urban reserves like *Costanera Sur Reserve* (370 ha) *Ribera Norte* (12 ha), and *Vicente Lopez* (3.5 ha) created in the 1980s and 1990s. These coastal reserves are homes to over 200 species of plants and 400 animal species, and connect to the coastal biodiversity corridor linked to the delta of the Paraná River. All of the reserves have free entry, are easily accessible by public transportation, and provide wonderful opportunities for bird watching as they are home to around 300 bird species. At the same time,

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the URs are also good examples of how nature can recover from anthropogenic activities, as some of them were created on landfills adjacent to the La Plata River where spontaneous, mostly native nature developed quickly. However, new challenges have appeared and the URs now face issues like exotic tree invasions. In a counter-action, the managers of UR Costanera Sur, which became famous through its four large lagoons, have developed a participatory framework to rehabilitate those lagoons that went dry in the last years because of the invasive trees.

In the near future, the metropolitan area needs to address and integrate green space management. This should be conceived as an ecosystem-based policy connecting the network of all types of urban greens, such as parks, green spaces, reserves, river corridors, remnant woodland and urban ornamental vegetation, with the ecosystems in the urban hinterlands. However, it is a distant goal today, since administrative authorities do not perceive the metropolitan area as a whole, and the several municipalities often have differing political interests, which affect the environmental agendas. It is imperative that policies for the metropolitan area, which is home to more than 1/3 of the population, are designed to meet users' needs and protect urban biodiversity with a long-term perspective. These policies should provide increased financial and qualified human resources for program implementation, which holds the potential to effectively safeguard and improve the natural capital.

Textbox 28.3 Colombia: Diversity in All Cities

Colombia's location and topography supports an unusually wide variety of landscapes, and floral and faunal species. Located in the northwest of South America, Colombia borders two oceans in the west and in the north, and contains jungles, savannahs, and mountains. It has the world's largest number of bird and orchid species, the second largest number of plant species, amphibians and butterflies, and the third largest number of reptile species (Revista Semana 2008).

Colombia's cities are still below the megacity minimum of ten million people. The country's three principal urban areas are located on the Andes Mountains and have 7.6, 2.4, and 2.3 million inhabitants in Bogotá, Medellín and Cali, respectively. The second largest group of cities is located on the Atlantic coast in the west, and 12 more, each with a population exceeding 400,000 inhabitants, are dispersed across the nation.

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Textbox 28.3 (continued)

Urban population growth and concentration in Colombia has caused a largely irreversible degradation of natural areas in and around the cities. In Medellín (Figs. 28.3 and 28.4), the second largest city of Colombia, the public administrators and local planning authorities have launched a Green Belt programme (Fig. 28.5) aimed to control the legal and illegal urban expansion up the hills that surround the city.

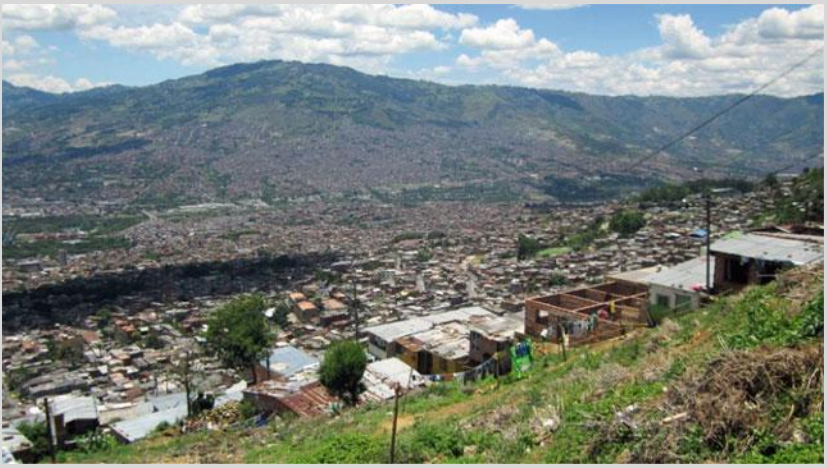

Fig. 28.3 Northwest Medellín, seen from the east slope of the valley. The area is inhabited primarily by low-income settlers, settling higher and higher up the slope, causing informal urban sprawl which intermingles with the surrounding ecosystems (Photo by and published with kind permission of ©Gloria Aponte 2012. All rights reserved)

Fig. 28.4 Medellín southeast, seen from the low west part of town. The urban sprawl in this part of the city is primarily the result of commercial housing development projects aimed to attract upper income families. In these areas, remnant vegetation is replaced by ornamental exotic species and therefore biodiversity is highly modified (Photo by and published with kind permission of ©Gloria Aponte. All rights reserved)

(continued)

Textbox 28.3 (continued)

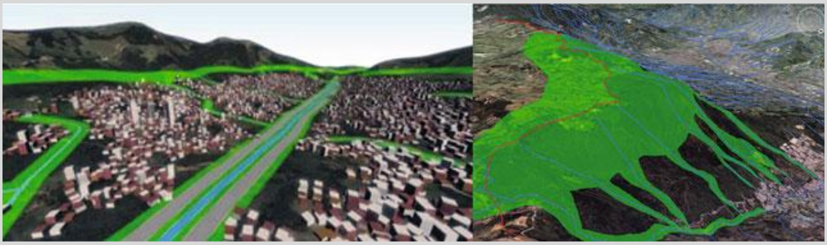

Fig. 28.5 Comparison of two landscape planning alternatives. Rather than basing the ecosystem management approach on a rigid green belt model (*left*), why not develop a model that is inspired by the traditional *poncho* and thus has a shape that is close at heart to the Colombian identity (*right*). This model would allow the water, born at the mountain tops, to flow naturally down the rough terrain, along the built-up environment, through the city's fringes, and be an integral part of the urban landscape, available for people to see and enjoy (Prepared by and published with kind permission of ©Gloria Aponte. All rights reserved)

However, the Green Belt model does not necessarily meet the needs of the Medellín society, nor benefit the local biodiversity, partly because it only addresses the 'green' aspect of the urban ecosystems. The 'blue' aspect is also vital to address, and is crucial for social and ecological well-being alike. Studies have been done on the natural water streams that flow down the hills that surround Medellín, along the valley where the city is located, and out to the rural hinterlands. These natural water bodies are currently used as outlets for household sewage, but the research findings show how instead the numerous streams can be used as means to naturally conserve and reinstate native biodiversity, as an integral part of the densely urbanized lower areas of the mountain hills. As a result of the research, a set of guidelines has been formulated, directed to improve the water quality of the streams, particularly in the border zone between the urban and the rural where the water leaves the urban area. The aim is to create a healthier urban living environment for humans and nature alike. As a secondary result, if the guidelines are implemented properly, the area's natural landscape may be restored and thereby better support local native biodiversity.

It is worth noting that little research has been done on urban ecology in small towns and medium-sized cities, which are the fastest growing areas of the region and together have the largest proportion of the region's population (WUP 2011). Furthermore, they are the areas that are expected to have the highest impact on biodiversity and ecosystem services because they have a greater perimeter/area ratio, and therefore higher interface with non-urban ecosystems (Aguayo et al. 2007).

Textbox 28.4 Mexico: Challenges for a Fast Growing Urban Population

The current urbanization trends in Mexico, such as the establishment of new settlement areas and sprawl of existing urban areas, are transforming natural and rural ecosystems (Garza and Schteingart 2010). The urban biodiversity commonly includes only a limited variety of floral species, which are typically scattered and exotic, and the urban conditions have been shown to accelerate depletion of faunal wildlife communities (Nocedal 1987; MacGregor-Fors et al. 2012). However, encouraging examples of urban areas can be found in Mexico that promote the presence of complex and diverse wildlife communities. High biodiversity-rich areas, where a rich flora of trees, shrubs, and herbaceous plants support ecosystem-specific fauna, often have a positive social impact, can increase the real estate values, and can improve the ecological quality of the areas (MacGregor-Fors et al. 2009; Ortega-Álvarez and MacGregor-Fors 2011).

Urban ecology as a research discipline in Mexico emerged only in the 1980s, with research focused on topics such as air and water pollution, local climate, urban greening, and urban-related fauna (Rapoport and López-Moreno 1987; Gío-Argáez et al. 1989). Fortunately, the interest in studying ecological patterns and processes in Mexican urban areas has increased considerably in the last decade. Many recent studies have focused on bioindicator groups to assess the response of wildlife communities to urbanization, while others have described urban vegetation shifts in relation to socioeconomic variables (e.g., birds) (Ortega-Álvarez and MacGregor-Fors 2011). Results of some of these studies have suggested interesting management and planning activities. However, there is an apparent lack of mechanisms for including the findings in policies, and tools to efficiently implement the policies.

There is a pressing need to fund and support urban ecological studies. A worrisome dearth of knowledge remains, regarding even the most basic knowledge of how urban ecosystems function and interact, especially as current ecological studies are conducted over a limited time span. Two major biases in the research also need to be addressed: (1) the focus is primarily on large cities located in the center of the country, basically ignoring the ecological patterns and processes in medium- to small-sized human settlements of northern and southern Mexico, and (2) research especially targets a few selected wildlife groups, mainly birds. Addressing these biases can contribute to bridging the current knowledge gaps in research. It can also yield suggestions on how to integrate knowledge and evidence-based action to not only increasing ecological quality of urban areas, but also to improve human well-being.

The need to bridge the knowledge gap between the findings in ecological research and the decision-making related to urbanization in Mexico is urgent. Policymakers and planners also need to involve citizens in governance

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Textbox 28.4 (continued)

processes in order to tailor decisions to meet the needs of the people and be effectively implemented. While conventional environmental education could raise awareness about urban-related issues and influence direct actions, several other novel ways can also draw people's attention and get them involved in creating ecologically-friendly cities. Urban areas and biodiversity need not be mutually exclusive, and cities can –and should– promote inclusiveness of nature and wildlife in the urban landscape, rather than maintaining barriers. Finally, it is crucial that decisions and actions in urban development aim to support an ecosystem-based urban development. They need to be carefully documented and analyzed, rest on a solid foundation of transdisciplinary research, and have a systems perspective rather than focusing on individual factors treated as separate from a social and ecological context.

It was also found that extremely few papers explicitly analyze the impacts of urbanization on biodiversity. The existing studies commonly looked at ecological components, but failed to connect these to the social development patterns or the ecosystems that they were part of. Birds are the most studied taxa in urban ecosystems worldwide (Evans 2010), with Brazil, Argentina and Mexico (Textbox 28.4) counting for 79 % of the publications (Ortega-Álvarez and MacGregor-Fors 2011). A general pattern the authors found was that bird species richness declined with an increasing urbanization rate, whilst bird abundances were highest in those areas with high housing density. Other variables such as town size (Ortega-Álvarez and MacGregor-Fors 2011), habitat quality, and availability and heterogeneity (Faggi and Perepelizin 2006), were found to be important factors for shaping bird distribution in cities such as for example Buenos Aires, Argentina.

There is a consistent lack of standard methodology to assess biodiversity and ecosystem services in urban settings, which can undermine comparability and generalizations. This situation may prevent a correct translation of scientific findings into management practices or policies. For example, acknowledging the importance of urban green spaces on ecosystem services provision (Bolund and Hunhammar 1999; Donovan et al. 2005; Nowak and Crane 2002; Barbosa et al. 2007), biodiversity (Clergeau et al. 2006; Cannon et al. 2005; Gaston et al. 2005), and human well-being (Fuller and Irvine 2010) is of extreme importance. Guidelines for access may vary between regions, however the World Health Organization recommends having between 9 and 11 m^2 per habitant; data gathered between 2003 and 2008 in 16 cities in Latin America, show that more than half of them exceed the recommendation (ONU-HABITAT 2012). Common criteria for determining how fundamental measures of "green space" availability, access and quality, needs to be established. Green spaces may be widely available but not necessarily be of good quality, and may not necessarily provide the expected ecosystem services (Barbosa et al. 2007).

Data on urban development and urban ecology is often collected for other purposes than municipal environment management planning, and may thus pass unnoticed by municipalities and researchers. However, properly merged and analyzed, this data could contribute significantly to urban ecology research in Latin America (Sagarin and Pauchard 2012). Today, municipalities rely to a large extent on documents such as Environmental Impact Assessments and urban zoning reports for their environmental planning and management, but the sources are commonly not considered scientifically robust by researchers, and the exchange of information between academia and urban planners is extremely limited.

Funding in Latin America for ecological research and especially for social-ecological research, is not a priority. However, there is an urgent need to understand the interplay between cities, biodiversity and ecosystem services in the region. Issues such as the effects of rapidly increasing urban density on ecosystem functions, how ecosystem services are linked to the availability of different types of urban green spaces, and how socioeconomics, urban morphology, and natural as well as anthropogenic hazards (e.g., landslides, peri-urban wildfires) affect ecosystem provisioning and biodiversity conservation over time should be targeted in research agendas. Such information would be enormously valuable in helping Latin American cities guide their urban planning and conservation policies, especially in more underprivileged countries and cities where little funding is directed to ecological research.

28.3 Conclusions

The rural to urban migration in Latin America is slowing down, as does the population growth within cities. The region is expected to reach its urban population peak within the next few decades. By then, it is expected that 90 % or more of the population will live in urban areas. However, as the examples in this text illustrates, cities keep expanding their boundaries following an influx of low-income settlers from rural areas, and an outflow by financially well-off inhabitants from the city cores to the peri-urban areas and the neighboring rural hinterlands.

The region at large contains some of the richest biodiversity in the world. Much of the urban sprawl in the peri-urban areas encroaches on highly sensitive ecosystems such as rivers, floodplains, wetlands, and coastlines (Chaps. 3 and 22). This in turn increases the risk of damage from natural hazards such as flooding and earthquakes which are common in the region. Furthermore, changes to ecosystems in the urban centers also percolate into peri-urban ecosystems, especially increasing the number of invasive species and pollutants in semi-natural areas.

Knowledge on how local ecosystems function in and around urban areas, how they are interconnected, and how they can adapt to a changing environment, is limited. Partially because of this lack of knowledge and partially because of limited communication between researchers, policy-makers, planners and the public, biodiversity and ecosystem conservation is poorly included in the planning of most

Latin American cities (Chaps. 21 and 22). A major effort needs to be implemented to study and monitor biodiversity and ecosystem services in the region considering both urban and peri-urban ecosystems and their interactions and connections. Although many studies have been done and much is known about these processes in other regions of the world, the unique characteristics of individual cities and ecosystems in Latin America strongly risk limiting the potential to generalize research findings.

Priorities in how to achieve a sustainable urban development remain a challenge. Conservation of ecosystems and biodiversity, and support for the provisioning of ecosystem services, fails to be acknowledged as a primary means to improve quality of life for the city inhabitants. Three key challenges to address in Latin America are thus: (1) to slow down the urban sprawl that is driven partly by extreme income inequity, that generates a complex mosaic of urban settings, that often encroaches biodiversity-rich and sensitive areas; (2) to understand the different dynamics of how humans manage and impact urban ecosystems in different cities, and across different social and income groups; and (3) to increase awareness amongst policy-makers, planners and the public on the importance of functioning ecosystems for human well-being, which is fundamental in order to change how cities foster biodiversity and ecosystem services. Encouraging involvement by the public in management of the local ecosystems, as well as in formal decision-making, can be key to increasing the chances for long-term compromises between ecosystem sustainability and city growth.

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Chapter 29
Local Assessment of Rio de Janeiro City: Two Case Studies of Urbanization Trends and Ecological Impacts

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Abstract Rio de Janeiro city (Rio) with its surrounding areas is undergoing a fast and extensive ecological transformation as a result of the city's social and economical development. The urban landscape expands partly over land with Atlantic Forest ecosystems (22,277.20 hectare (ha) of natural areas which comprise 18.2 % of city area), with some of the richest biodiversity and most endangered species in the world. Urban sprawl and ecosystem degradation are two challenges that the city faces. This chapter presents case studies of two areas in Rio, representing widely differing social-ecological contexts. The first is located in the *Misericórdia (Compassion)* Massif in the midst of the most densely populated and highly impervious area of the city, where residents have gotten together to reintroduce biodiversity in order to restore ecosystem services. The second case study is situated in the *Barra da Tijuca* area, a new real estate development region where the developer was responsible for the restoration and protection of ecological parks to comply with environmental legislation. In both cases, the forests offer crucial and irreplaceable ecosystem services, but the residents' perceptions of the forests differs vastly. The two cases can serve as inspiration for ways to let biodiversity support play a central role in urban planning, design, and retrofitting of urban ecosystems in the future.

Keywords Urban sprawl • Biodiversity hotspots • Ecosystem management • Rio de Janeiro • Case studies

Key Findings

- The city is sprawling
- As an effect the ecosystems are under threat of severe damage and even eradication
- Formal regulations provide protection, but are being changed to support the international sports events and real estate market
- Future challenges for people are to have their voice heard by the decision makers to protect and enhance urban biodiversity and ecosystem services
- Possible solutions/gaps that need addressing are: effective science based long term planning and all stakeholders (scientists, academia, citizens, NGOs) participation, not restricted to the privileged decision makers, entrepreneurs and international sports events organizations

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29.1 Introduction

Rio de Janeiro city has an estimated population of 6.3 million inhabitants (Brasil 2011) spread over 1,255.3 km^2 (Brasil 2008). Partially forested massifs divide the city into zones, which are officially organized since 1977 in Planning Areas (PAs), of which today there are five. Each PA is composed of several districts, in conformity with their historical, natural, social and economic characteristics. The city's average population density is 72.87 (SD ± 40.83) people per hectare (PUB-Rio 1977; Rio de Janeiro 2000).

The landscape where the city rests is shaped by lowlands and rocky hills, partly covered of Atlantic Forest vegetational formations (IBGE 2012). These formations encompass a variety of ecosystems like tropical forests, mangrove (Soares et al. 2003), and *restinga*, a biodiverse ecosystem that covers sandbanks and is composed by herbaceous plants, shrubs, and arboreal species (Rizzini 1979; Araujo 2000; Scarano 2002). This biome is considered one of the world's 25 biodiversity hotspots (Davis et al. 1986; Myers et al. 2000) and is home to many endangered plant and animal species (Box 29.1). Mountain slopes are characteristically covered by evergreen tropical rainforests with a floristic and faunal composition that varies according to related altitudinal conditions, such as temperature and humidity, and to the geographical position of Rio de Janeiro hills, which influences solar radiation incidence (Cerqueira et al. 1990; Brasil 1992; Oliveira et al. 1995). The city's ecosystems are fragile and vulnerable to climate change effects, such as sea level rise, warmer climate, stronger and more frequent storms, droughts, floods, and landslides among others (Gusmão 2011).

Box 29.1 Rio de Janeiro's Threatened Biodiversity

The land area that Rio de Janeiro city covers is one of the areas in Brazil most impacted by colonization and land occupation. The historical development has caused extinction of species, and the contemporary urban expansion continues to pose a threat to many species and native ecosystems. However, a rich biodiversity still remains.

A bibliographic survey was performed for this assessment of the diversity of tree species on the forested hills within the Rio de Janeiro municipality, based on published research on trees with a minimum diameter equal to or higher than 5 cm. Available data was found for *Pedra Branca* State Park (Peixoto et al. 2004), Tijuca National Forest (Matos 2007), and three different urban forest fragments located in the west zone (Santana et al. 2004). The total sample area of the three studies was 7,000 m^2 (0.7 ha), and was found to contain 293 tree species. Among these species, four are exotic: *Actinostemon*

(continued)

Box 29.1 (continued)

klotzschii (Didr.), Pax (Euphorbiaceae), *Mangifera indica* L. (mango tree – Anacardiaceae), *Artocarpus heterophyllus* Lam. (jackfruit – Moraceae), and *Pachira glabra* Pasq. (Malvaceae). The jackfruit is extremely abundant due to its high recruitment rate (Abreu 2008a, b), and the mango tree is very popular, planted in public and private properties all over the city (personal observations). Twenty species are considered endangered or critically endangered (PCRJ 1997), for example including *Caesalpinia echinata* Lam. (Fabaceae), *Ocotea odorifera* (Vell.), Rohwer (Lauraceae), and *Rudgea interrupta* Benth (Rubiaceae). All are classed by the Brazilian Environmental Ministry as priority for conservation funding projects (Instrução Normativa nº6 2008). Freire et al. (2009) conducted a study of trees with a trunk diameter wider than 15 cm, in a 5,000 m^2 (0.5 ha) sample area in the *Pedra Branca* State Park. In total, 264 tree species were found, which shows on a high species richness amongst the larger trees. This richness, and the support that large trees provide for ecosystems, indicate that the trees are key elements of the local ecosystems.

The most reliable data gathered on other ecosystems and vegetation types (herbaceous, epiphytes, and climbing) within the Rio de Janeiro municipality's borders, showed that there were 41 critically endangered and 68 endangered species at the time of the study (PCRJ 1997). The findings also showed that within the municipality's borders, 33 faunal species were rated as critically endangered, and 52 as endangered (PCRJ 1997). Examples of mammal species classified as critically endangered included the tiger cat (*Leopardo tigrinus*), the ocelot (*Leopardo pardalis*), and the capybara (*Hidrochaeris hidrochaeris*). Other critically endangered and endangered species within the municipality's borders include: many species of birds, e.g., *Procnias nudicollis*, and rosed spoon bill (*Platalea ajaja*); fish, e.g., white shark (*Carcarhodon carcharias*), slender seahorse (*Hippocampus reidi*); amphibians, e.g., *Bokermannohyla circumdata*, *Cycloramphus fuliginosus*; and insects, like the butterfly *Parides ascanis*, and the dragon-fly *Idioneura ancilla,* (PCRJ 1997). Some species were already extinct, e.g., medium to large mammals: brown howler monkey (*Allouata fusca*), muriqui (*Bracytheles arachnoides*), golden lion tamarim (*Leontopithecus rosalia*), jaguar (*Panthera onca*), puma (*Panthera onca*), and peccaries, such as the collared (*Pecari tajacu*) and the white-lipped peccari (*Tayassu pecari*).

This chapter will present two case studies that strongly contrast with each other (Fig. 29.1). The first is in the Northern zone (Planning Area – PA 3) where only 4 % of the area is forested, although it is the densest with 117.94 people per hectare.[1]

[1] Calculations were based on data available at the Rio de Janeiro municipality web sites www.sigfloresta.rio.rj.gov.br/ and http://portalgeo.rio.rj.gov.br/bdario/

Fig. 29.1 The two case study areas: The *Misericórdia* massif in PA3 (Case Study 1) and the *Barra da Tijuca* district in PA 4 (Case Study 2) (Prepared by and published with kind permission of © Brasiliano Vito Fico/Smac 2012. All Rights Reserved)

Table 29.1 Comparison of the districts, *Inhaúma* and *Barra da Tijuca*, where the two case study sites are located

	INHAÚMA	BARRA
Area (ha)	348.5	4,799.1
Population (inhabitants)	134,349	300,823
Urbanized %	97.9 %	46.3 %
Green areas %	1.6 %	27.0 %
Average household income (R$)	1,300.00	9,000.000

Source: Instituto Pereira Passos – IPP with IPBGE – Census 2010 data and http://sigfloresta.rio.rj.gov.br/
Data source: Rio de Janeiro (2000, 2002, 2005)

The case study is located in the *Misericórdia* massif in the *Inhaúma* District, in the midst of the largest favela complex of the city where half of the Rio de Janeiro's slums city dwellers live and has a community driven bottom-up approach to ecosystem management (Rio de Janeiro 2005).

The second is of the Southern city area (PA 4) that has more than 40 % of forested land cover. It is located in the coastal *Barra da Tijuca* district, and is composed of three ecological parks in one of the most recently established subdivisions of the city. It is an example of a top-down governed ecological restoration and conservation. The two areas have very different ecological, social, cultural and economic characteristics (Table 29.1) but both are examples of good practices of Atlantic Rainforest ecosystems restoration that resulted in now legally protected forested areas.

29.2 Research Methodology

The assessment builds on a literature review of publications and official documents. Furthermore, semi-structured interviews (Zeisel 2006) were done in order to understand the historical development and legislative framework surrounding the ecosystems and their services. They also served the purpose of creating an understanding for residents' and volunteers' perception of the ecosystem service values. The respondents were selected according to their relevance on the historical process and implication on the restoration practices. Common respondents for both case study areas were city officials involved in different departments of mapping, reforestation, Conservation Units and legislation, and the Municipal Environmental Secretary. In the *Misericórdia* massif case study respondents further included the NGO president, representatives, volunteers, residents and local users of the park. In the *Barra da Tijuca* the case-specific respondents included the owner-president and advisors of the real estate company, City and State personnel involved in the environmental law enforcement during the licensing process in 1990s, landscape architects involved in the parks planning and design, residents, visitors, and employees. The interviewees of each case study were selected according to their participation in the biodiversity restoration and maintenance process.

29.3 Urbanization, Ecosystem Services and Biodiversity; Scenarios and Trends

During the colonization period (1530–1815), the landscape was inhospitable to human occupation due to the dense rainforest and natural humid areas as wetlands and mangroves, what led to an extensive transformation of the original sites. Since early 1900s, hills were torn down to create land over wetlands and the ocean until the contemporary urban sprawl (Oliveira 2007; Correa et al. 2001; Pinheiro 2010; Rabha 2010). The anthropogenic occupation led to extensive and massive deforestation processes that were induced by many economical cycles and population growth (Dean 2002). Contemporary studies estimate that the remaining Atlantic rain forest biome fragments cover 11.4–16 % of the original area (Ribeiro et al. 2009). In Rio de Janeiro city, from 1984 to 2001, the Atlantic ecosystems' canopy cover had an area reduction of approximately 28 % (Rio de Janeiro 2002). The original mangroves in the estuarine areas, lagoons and bay margins covered estimated 257.9 km^2, approximately 80 km^2 remains (Amador 1996).

Severe droughts in the nineteenth century led to a pioneer intensive reforestation of the Tijuca massif mainly with native tree species (some examples of the species used are that of the genus *Cariniana* and *Lecythis* (Lecythidaceae), *Tibouchina* (Melastomataceae), *Handroanthus* (Bignoniaceae), many Fabaceae (legumes) species of the genus *Copaifera*, *Platymiscium, Swartzia* and *Caesalpinia*). Some exotic trees as mango trees (*Mangifera indica* L.), jackfruit (*Artocarpus heterophyllus* Lam.) and species of *Eucalyptus* were also introduced (Drummond 1997).

The objectives of the reforestation were to restore ecosystems services, like water sources, regulate local climate, enable botanic research, and provide recreation. After 150 years, an extensive area of the massif is today forested, protected by a National Park, and is part of the Atlantic Rainforest Biosphere Reserve, a United Nations initiative (Correa et al. 2001; Coelho Netto 2005; Vieira et al. 2010).

The city of Rio de Janeiro was the national capital until April 1960, when it was transferred to Brasilia. The next city's administrations planned and started to expand the urbanized areas focused on the development of new centralities based on private transportation, without proper infrastructure and social housing. The vision of a future mega city led to the opening of a highway system in 1971 to connect 160 km^2 located between the Pedra Branca massif and the ocean in the Jacarepaguá watershed (Rabha 2010). In the decades following that, the *favelas* (slums) rapidly spread over vulnerable steep slopes and soggy lowlands in clusters along the coast or in distant areas with scarce infrastructure, especially sanitation, and green spaces (Abreu 2008a, b). The formal real estate market occupied the most valuable areas, known as the formal city, which created a rapid vertical spread of the city along the coast with a radical landscape transformation, inspired by the American dream of urban Eldorado, having Miami as architectural and urbanistic model (Rabha 2010).

29.4 Case 1: The *Misericórdia* Massif

29.4.1 The Role of Ecosystem and Biodiversity, and Effects of Urbanization

The largest massif at the PA3 is named *Misericórdia*. It is partially covered by one of the last forest remnants of the entire Northern Region of Rio de Janeiro (Fig. 29.2). The region has one of the highest population densities of the city with 37 % of the population distributed over 69 slums (Rio de Janeiro 2002).

The massif was originally a farm that was parceled in the twentieth century, and later it was invaded by homeless migrant workers attracted by the local industrial development. In the last decades it became one of the most violent places of the city, with drug dealers ruling a parallel power over the entire informal settlement (Ventura 1995). In end of 2010 the State Police of Rio de Janeiro took over the region, and now the area is mostly under official control.

Granite quarrying results in severe environmental impacts such as elimination of the vegetation cover and the average particulate material on air (a measure of air pollution) in the vicinity has almost double to the acceptable for human health (90–50 m/m^3). In some places on PA3 it can reach 145 m/m^3, when the limit should be 80 m/m^3 (Prefeitura-RJ 2001). Dense and widespread self-constructed *favela* houses have sealed the soil and substantially altered the landscape structure, together with its processes and flows. Rivers and creeks have been channelized or buried underground, with severe consequences such as floods and landslides.

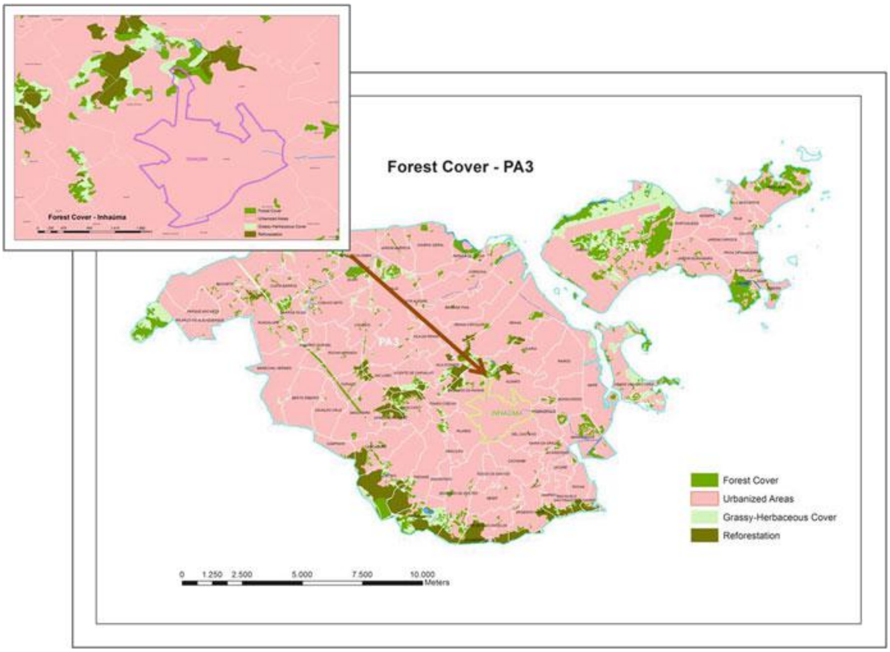

Fig. 29.2 One of the last forest remnants of the entire Northern Region of Rio de Janeiro partially covers the *Misericórdia* massif, located in the Inhaúma district in PA3. The area has one of the highest population densities of Rio de Janeiro City. The reforested hills of *Verdejar* are at the northern border of *Inhaúma* (Modified from SMAC, and published with kind permission of © Brasiliano Vito Fico/SMAC 2012. All Rights Reserved)

To face the challenges of the biodiversity eradication, in late 1980s *Verdejar* NGO started to implement several reforestation and educational practices with the local community located in the *Inhaúma* district. The NGO was established by a personal effort of Luiz Carlos Marins, known as "Poet", who lived nearby the district and used the grass covered slopes to practice sports. He began to plant tree seedlings and remove garbage thrown by area dwellers. Gradually, local people started to join him in his actions and become aware of the environmental issues. The main problems were illegal housing occupation in the grassy steep slopes and rock quarry for the construction industry, of which the latter still remains.

The *Verdejar* members and volunteers have been actively involved in the area conservation and restoration in different fronts. Firstly, they have directly confronted invaders and thus avoided new settlements in the hills. Subsequently, they have worked on ecological restoration, food production and agroforest implementation. In order to motivate local people, they have promoted educational, cultural and artistic actions to clean the trails and planted the new tree seedlings of common native Atlantic rainforest species (e.g., *Schinus terebinthifolius* Raddi (aroeira), *Bixa orellana* L. (urucum), *Handroanthus chrysotrichus* (Mart. ex DC.) Mattos (ipê), *Cybistax antisyphilitica* (Mart.) Mart., *Cordia superba* Cham., *Piptadenia gonoacantha* (Mart.) J. F. Macbr., *Schizolobium parahyba* (Vell.) S.F. Blake, *Trema micrantha* (L.) Blume) and native fruit species (for example *Psidium guajava* L (guava), *Genipa infundibuliformis* Zappi & Semir (genipapo), *Spondias dulcis* Parkinson (mango cajá), *Theobroma cacao L (cocoa) and. Ziziphus joazeiro* Mart. (juazeiro)) and have been managing the forest constantly.

29.4.2 Urbanization Trends and Expected Future Development

Nowadays, the area is largely reforested. *Verdejar* is directly responsible for the planting of 7,731 m^2, and their educational and protective actions allowed 54,263 m^2 to naturally regenerate, with care of local residents. They have also mobilized stakeholders through public demonstrations and local assemblies against the impacts of rock mining on health related problems and environmental degradation. By their actions they also stimulate the implementation of a public program of reforestation in the higher elevations of this region.[2]

Currently, *Verdejar* NGO is involved in official projects, as well as in ecological and social networks that help their improvement with technical and financial support. Although the local infrastructure is simple, *Verdejar* has managed to restore an extensive, environmentally degraded area where their head-office is located, with educational and recreation spaces, an ecological dry bathroom, and lately they are working on a native and edible vegetables nursery and garden. Future objectives include: improve the reforestation of grassy slopes, with more local mobilization and incentive to multiple use and care for the area.

29.5 Case 2: *Barra da Tijuca*: *Peninsula, Gleba F* and *Mello Barreto Parks*

29.5.1 The Role of Ecosystem and Biodiversity, and Effects of Urbanization

The *Barra da Tijuca* district landscape (Fig. 29.3), located in the PA4, was completely transformed after the construction of a network of highways and roads. Two main highways were first built in the early 1970s: East–west *Avenida das Américas* and North–south *Avenida Ayton Senna*. They divide the watershed into four areas, which is intended to facilitate the urban expansion. More roads have been added to the circulatory system over humid areas and are promoting further real estate speculation over rich biodiversity and fragile humid territory. As a result,

[2] The city program is *Mutirão Reflorestamento* (common effort to reforest).

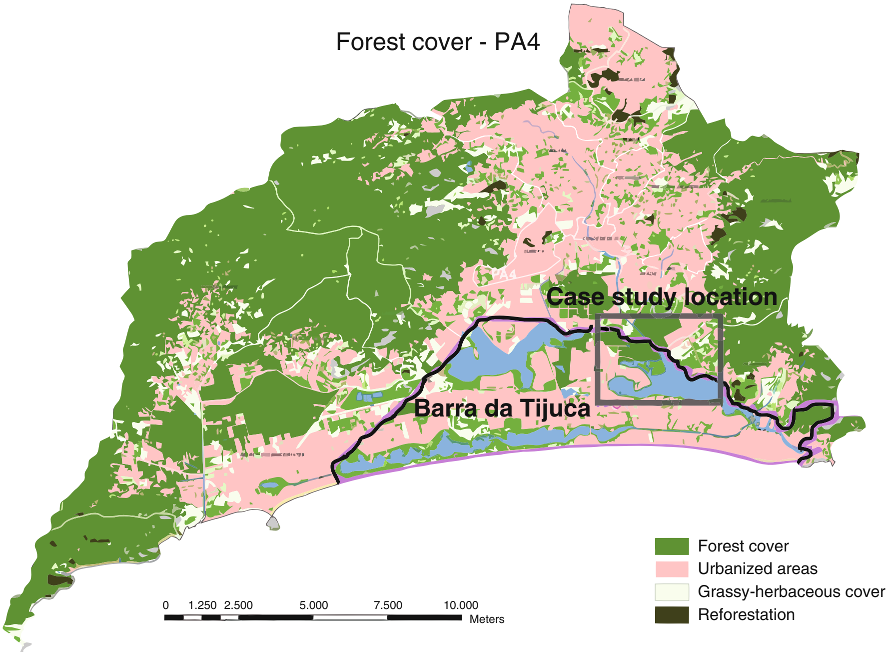

Fig. 29.3 PA4 with the *Barra da Tijuca* borders and the Case Study 2 location (Modified from SMAC. © Brasiliano Vito Fico/SMAC 2012. All Rights Reserved)

in the last three decades, extensive areas of native ecosystems were almost completely transformed or suppressed to give place to high and medium income residential and commercial complexes as well as shopping malls (Pinheiro and Pinheiro 2001). The globalization trend of open public and private spaces led to manicured gardens (Ignatieva 2010) based on water-demanding green grass lawns, a few exotic ornamental species and palm trees, creating *biotic homogenization* or a decrease of biodiversity (Müller and Werner 2010). In approximately 7 years (1984–2001), the *Barra da Tijuca* district lost about 13 % of its natural areas, *restinga* being the most affected, accounting for 41 % of the lost natural areas; followed by forest (19 %) and mangrove (6 %).[3]

The Jacarepaguá watershed, which comprises the PA4, is one of the most vulnerable areas to climate change impacts, especially sea level rise, floods and landslides (Fig. 29.4). The area comprehends most of the largest fragments of native and restored mangrove and *restinga* biodiversity of indigenous Atlantic forest ecosystems in the region, among other local Municipal and State Conservation Units.[4] The study area encompasses ecosystems that are largely fragmented but retains some degree of

[3] www.armazemdedados.rio.rj.gov.br – Tabela nº1783 viewed 08.08.2012 [in Portuguese].

[4] www.sigfloresta.rio.rj.gov.br/ viewed in 08.12.2012.

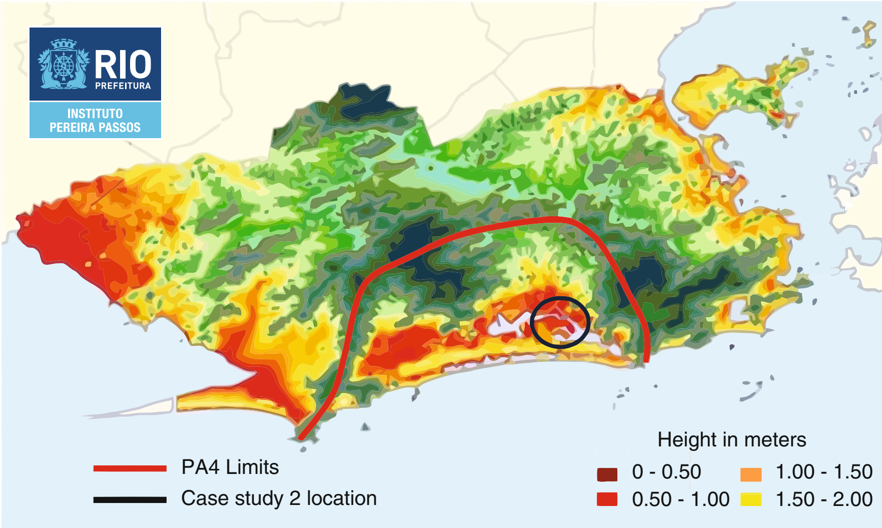

Fig. 29.4 *Barra da Tijuca* vulnerability city map: land up to 2 m above sea level are considered subject to sea level rise. The PA4 is marked in red, and the case study location is marked in dark blue (Modified from Gusmão et al. 2008, p. 93 and published with kind permission of © Rio Prefeitura, Instituto Pereira Passos 2008. All Rights Reserved)

connection to each other, consisting of about 460,000 m^2 of protected forests, along 6.5 km of coastal lagoons, in three public ecological parks in a highly urbanized area. Although the parks are public, two of them, *Peninsula* and *Gleba F. Mello Barreto* Environmental Educational Park, are inside private owned areas.

The first development area, *Peninsula*, was previously a lagoon sediment dredge disposal site with a mangrove fringe that had degraded following human interventions (Figs. 29.5 and 29.6). The entrepreneur envisioned that restoring and conserving the ecosystems in a planned and designed park with native vegetation would give additional sales value to his properties at the *Peninsula* estate.

In 1986 one of the most renowned Brazilian landscape architects, Fernando Chacel, was hired to design a 77,000 m^2, 3 km long lagoon-fronting ecological park. The project was developed by an interdisciplinary team to recover through an "ecogenetic" process (Chacel 2001), an aesthetically designed restoration of the mangrove and *restinga* vegetation, connected by walking trails with diverse facilities to support extensive public use (Jacobs 2007). The park was planned and designed with the support of an interdisciplinary team, headed by Prof. Mello Barreto, a renowned botanist; Sidney Linhares, a landscape architect; and Mario Moscatelli, who led the mangrove restoration. The project aimed to be multifunctional and promote abiotic, biotic and cultural ecosystem services, by providing native biodiversity habitat and connectivity, and enhancing lagoon water quality through mangrove planting, which together also contributed to prevent erosion and sedimentation.

Fig. 29.5 Aerial front view of the *Peninsula* complex in 1997, before it was developed. To the left, the lagoon margin is Mello Barreto Park. To the right is the *Gleba F* (Photo taken by and published with kind permission of © Carvalho Hosken S.A 1997. All Rights Reserved)

Fig. 29.6 Aerial view of the *Peninsula* and *Gleba F* in 2012, after development and establishment of the three parks: two along the lagoon in the left side, and the large green area on the right side. (Photo taken by and published with kind permission of © Carvalho Hosken S.A. 2012. All Rights Reserved)

The project further aimed to provide support for human activities and included the construction of playgrounds, trails, picnic places and rest areas (Chacel 2001).

Construction of the second park, *Mello Barreto,* began in the 1990s. The municipality allied with the residents association (ACIBARRA) and succeeded in recovering an illegally occupied and highly degraded strip of land, along the lagoon in the southern area. The same professionals that were responsible for the *Peninsula* park planning and design were in charge of the ecological restoration and transformed the area into an environmental educational public park. The *Mello Barreto* park was a much more complex project since the area first had to be cleared from houses and their supporting land filling. The parks were designed to combine social and ecological functions: restore mangrove and restinga ecosystems, as well as promote human uses (Chacel 2001).

The *Gleba F* is located on the northern side of *Peninsula*. It includes a 207,061.26 m^2 ecological park area covered by virtually pristine mangrove and *restinga* forests, which hosts a rich native biodiversity but has only to a very limited extent been focus for research (Soares 1998). The ecological park is still closed to the public, but the developer's idea is to start planning and designing it in a way that can preserve the high biodiversity that live in the area, and open the park to the public in a careful manner, not to degrade the environment with respect for the native faunal and floral species. The developer foresees that the park will add value to his properties and that it will also play a role in environmental education by providing people with direct contact with nature, inside a highly urbanized region.[5]

More than 11,000 people[6] live in the *Peninsula* complex. The interviewed visitors and residents[7] stated that the parks provided them with physical, mental and psychological well-being, direct contact with nature (flora and fauna), pleasant temperature and ambience, clean air, lagoon view, recreation and exercise inside a natural environmental and fruit collection straight from the trees, primarily anthropocentric benefits.

Although the PA4 holds great social and ecological values, as shown in this study, challenges are mounting. The PA4 is under heavy pressure from urban expansion, mainly because of upcoming major sports events that have boosted financial investments in the watershed area, including new highways, and sites for the 2016 Olympic Games. The region at large sees a significant number of new real estate developments. However, as shown by the respondents in this study, the parks are vital nodes for the local biodiversity, and hold a strong social-ecological importance for the area. The "ecogenetic" design that mimics native ecosystems is an important instrument to keep raising social awareness of the aesthetic potential of indigenous biodiversity in green areas (Nassauer 1997), and potentially support ecosystem stewardship in the future.

[5] Interview with Carlos Carvalho, Carvalho Hosken S.A. owner-president, in March 23rd 2012.

[6] Data from *ASSAPE*, Peninsula Residents Association in March 28th 2012.

[7] 18 interviewees, 9 men, 7 women, 2 children.

29.6 Governance and Formal Institutions

29.6.1 Case 1: The Misericórdia Massif and the Non-governmental Organization VERDEJAR

The socially oriented movements for the *Misericórdia* Massif preservation started in 1985, and were intensified in the 1990s decade. Local people missed the ecosystem services that were eliminated during the occupation process and decided to protect and restore the reduced remaining open spaces. The locals organized seedling planting actions, ecological hiking, meetings with local communities, and started to draw attention to the region's social and ecological problems. Two NGOs and their representatives were important in this process: *Bicuda Ecológica* and *Verdejar*. In 1995, both of them succeeded in bringing an official reforestation program called *Mutirão Reflorestamento* (meaning Reforestation Common Effort) to initiate the reforestation efforts in a number of hilltop locations. This currently ongoing municipal program temporarily employs local people to plant seedlings with technical assistance by the municipality.

The organizations promoted regular environmental educational lectures in 1998, and in 2000 they organized a seminar about the *Misericórdia* massif's main problems to discuss garbage recycling, water usage, population growth and environmental degradation, among others.[8] The event brought together several different groups of stakeholders, such as academic researchers, local associations' delegates, politicians as well as city technical and judiciary representatives. As a result of the intense mobilization a Municipal Conservation Unit (CU) named *Misericórdia* APARU[9] (meaning Environmental and Urban Recovery Area) was created by the city administration in 2000. However, in spite of its legal framework, the practical development of the CU, such as creating a management council; establishing an administrative venue; hire proper staff to work in the protected area; and develop conservation and educational programs, never happened.

29.6.2 Case 2: Barra da Tijuca: Peninsula, Gleba F and Mello Barreto Parks

The three parks at *Barra da Tijuca* were created in accordance with the municipality's environmental legislation requirements, to restore and conserve legally protected riparian corridors, and the well-preserved native forested fragment located at *Gleba F*.

[8] More information about NGOs' activities and history http://verdejar.wordpress.com and http://www.bicuda.org.br/rede/bem-vindo-a-nossa-rede/bicuda-ecologica/historico viewed 25.072012. [in Portuguese].

[9] Law n° 19144, November 14, 2000. Available at http://www2.rio.rj.gov.br/smu/buscafacil/Arquivos/PDF/D19144M.PDF viewed 29.03.2012 [in Portuguese].

The State and City departments, and the real estate entrepreneur, were involved in an intensive litigation from 1980 until the new development project was approved in 2000.[10] The litigation ensured that the real estate company complied with existing laws, and adapted the planned development to the occupation density and soil permeability, before the final development licenses were received for the *Peninsula* and *Gleba F* residential-commercial subdivisions.[11]

29.7 Concluding Remarks

Almost all of Rio de Janeiro's landscape has been transformed and degraded by anthropogenic activities, which still today threaten the remnant forested areas (Drummond 1997; Soares 1998). The restoration and management of the urban area's ecosystems is thus imperative but needs to consider social, institutional and governance aspects for its long-term sustainability (ITTO 2002).

The 2011 Director Plan[12] favorably regulated the protection and restoration of urban biodiversity. During the last years, additional governmental programs have been implemented to restore, increase and monitor biodiversity in the city, especially *Mutirão Reflorestamento* (Reforestation Common Effort), Conservation Units implementation, *Hortas Cariocas* (Vegetable Gardens in municipal schools) and *Sigfloresta* (GISForests, mapping and monitoring urban native forest fragments). Therefore, there were improvements in the urban-forested coverage (Box 29.2).

Box 29.2 Rio de Janeiro's Land Cover Changes: Past, Present, and Future Trends

From 1960 and throughout the second half of the twentieth century, the establishment of Conservation Units (CUs) was the main tool to promote biodiversity protection. This was followed in 1986 by the implementation of a successful city reforestation program, *Mutirão Reflorestamento* (Reforestation Common Effort). As a result of the implementation of CUs, protected areas grew remarkably, from 97 ha in 1960 to 18,685 ha in 1974. The advance was mainly a result of the establishment of the largest CUs: *Tijuca National Park*, established in 1961 and covering 3,360 ha; *Pedra Branca State Park,*

(continued)

[10] Interview with Carlos Carvalho, president-owner of Carvalho Hosken S.A. in March 23rd, 2012.

[11] Carvalho Hosken S.A. opened all documents related to the history and approval of the real estate incorporation.

[12] Lei Complementar n.º 111/2011 (Legal act) [in Portuguese].

Box 29.2 (continued)

established in 1974 and covering 12,500 ha; and *Guaratiba Archaeological and Biological Reserve*, established in 1974 and covering 2,800 ha.

However, a significant drop in the increase of protected areas occurred from 1990 to 1995, when they increased from 19,951 to 23,387 ha, thus only adding 3,436 ha over a 5 year period (Rio de Janeiro 1998). The pace of CUs protection from 1999 to 2006 was even slower, going from 23,387 to 23,581 ha, only 194 ha in total. On the other hand, *Mutirão Reflorestamento* was initiated in 1986, and had resulted in 1,920 ha of reforested slopes by 2010, mainly in landslide susceptible areas. If this trend continues we can estimate that in 2050, CUs will cover 29,647 ha, which would be an increase of 6,065.85 ha compared to 2006. The increase in other types of reforested areas following the *Mutirão Reflorestamento* afforestation program has followed a linear trend, going up from 1,120 ha in 2000 to 1,920 ha in 2010. If this trend continues, in 2050, 4,777.20 ha will be covered by replanted forests, an increase of 2,857.20 ha compared to 2010.

However, the two policies have not been sufficient to control or compensate for the degradation of native ecosystems. The decrease of natural areas and the increase of urban sprawl can be estimated based on data from 1984, 2001 and 2010. From 1984 to 2001, the area covered by natural landscapes decreased from 43,384.48 to 36,567 ha; a decrease rate of 401.28 ha/year. In the same period the total urban area increased from 33,749.94 to 42,023 ha; an expansion rate of 486.65 ha/year. From 2001 to 2010 the forested land cover shrunk to 2,8536.3 ha; a decrease rate of 892.30 ha/year, and urbanized areas grew to 53,114.60 ha; an increase rate of 1,232.40 ha/year.

Methodology: The projections of past and future land cover change are based on the data published by Rio de Janeiro Municipality (2000). The future projections of CUs and tree canopy covered land in 2050 were made using data tables, and with available GIS data. It is possible that differences in methodology from the data extracted from GIS estimates do not lead to precise figures; however, it is important to note the tendencies that these estimates represent. Regarding CU's and *Mutirão Reflorestamento,* the figures were more dependable for projections for 2050. The projections for 2050 in CUs' total area are based on 1999–2010. The time period data selection was made to correspond to that of *Mutirão Reflorestamento*, which started in 1986 and have reliable data from 2000. Existing land cover change data is quantitative inadequate to make reliable trend illustrations and predictions, and only simple values could be estimated. This was done by calculating the difference between natural and urban land cover, comparing different years and dividing the differences by the number of years, assuming that these had a constant rate.

However, the growing city continues to spread over sensitive areas that suppress natural ecosystems, to give place to roads, rivers channelization and extensive impervious surfaces, which leads to a decrease in natural areas and an increase in urbanized areas.

In spite of the protective legislation, the deforestation continues, mainly in PA4 and PA5, where most of the city's native ecosystem fragments are located. The new infrastructure, oriented to support the international sports events (World Soccer Cup in 2014 and the Olympic Games in 2016), leads to further urban expansion. One of the major impacts on urban biodiversity is the legislation alteration – *Projeto de Lei Complementar* nº 125/2013–from late 2012, which aims to realize the construction of the Olympic golf course in a high priority conservation zone. The legislation alteration fails to demand a proper ecosystem impact assessment and has been accepted without inclusion of the public in the decision-making. On the other hand, a new suggested municipal urban Green Corridors program, if approved by the decision makers, may mitigate some of the negative impacts of the urban development, as the program is aimed to protect vulnerable areas, and connect remnant forest areas located in the massifs and lower humid lands.[13]

The two case studies present two vastly different social-ecological contexts, although both provide ecosystem services. In the *Misericóridia* case, the community played a determinant role to legally and effectively protect and reforest the slopes of the massif. *Verdejar* members are engaged on advance their own formal education (undergraduate and graduate), in spite of financial difficulties in order to continue to develop their projects while they still continuously engage in volunteer jobs. The members have since the organization's beginning attracted, educated and raised awareness of the natural area amongst the local community residents. The *Verdejar* members and their local partners have a deep understanding of the ecosystem services the forest provide locally, regionally and even globally,[14] and continue to work to improve the environmental legislation and governance.

In the second case study, the interviewed residents were attracted to live there because of the green areas, although they only had a vague perception of the ecological benefits of the forests. They valued the biodiversity, and were eager to learn more about the local ecosystems. The region offered an array of indoor recreation and entertainment that competed with outdoor life. The residents' relation with nature was passive, they were not engaged in the planting and management as the *Verdejar* people were.

The case studies show that several projects and strategies were in play in Rio de Janeiro aimed to increase the urban forests, with stakeholders on several social levels involved. The findings point to that education and effective public participation seems to be the key to sustain and support biodiversity in the city in the long term.

Urban forests are important as they provide a number of important ecosystem services, including improvement of chemical and physical environmental processes, energy conservation, carbon dioxide storage (ITTO 2002), and improvement of air quality and urban hydrology (Dwyer et al. 1992; Stromberg 2001). Climate change

[13] http://mosaico-carioca.blogspot.com.br/ visited 03.07.2012 [in Portuguese].

[14] Interviews with NGO volunteers in March 2012.

is increasing the threats of heat island effects, sea level rise, droughts, floods and landslides among others. These events, already common in the city, are becoming increasingly frequent and intense, and causes heavy economic, social and ecological losses (Coelho Netto 2005; Brandão 2004; Gusmão et al. 2008; Gusmão 2011). Changes in land-use and a decrease in urban forest cover contribute to increase the city's vulnerability to the challenges posed by climate changes. Therefore, biodiversity conservation should be prioritized in order to increase urban resilience, by contributing to mitigate GHG emissions, and support the capacity of urban ecosystems to adapt to unexpected changes (Novotny et al. 2010; Blanco et al. 2010).

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Chapter 30
Urban Landscapes as Learning Arenas for Biodiversity and Ecosystem Services Management

Marianne E. Krasny, Cecilia Lundholm, Soul Shava, Eunju Lee, and Hiromi Kobori

Abstract Using examples from Asia, Africa, and North America, we demonstrate how restoration and stewardship projects, including those with significant community engagement, provide opportunities for environmental and biodiversity learning in cities. Although research on such programs is in its initial stages, several studies show positive impacts of urban environmental education and related field science inquiry experiences on participant environmental attitudes, awareness of urban nature, science understanding, and self-efficacy, with greater effects correlated with degree of involvement in hands-on, field-based experiences. In addition, programs that actively

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engage participants in restoration and inquiry reflect social equity, participatory, and environmental principles central to global initiatives in environmental education and sustainability. Such projects also reflect current theories of learning including those focusing on the ways children construct understanding of phenomena they encounter in everyday life (constructivism) and those that describe learning as an outcome of interaction with the socio-cultural and bio-physical environment (social learning). While recognizing the importance of school-based learning, our case examples illustrate the myriad of out-of-school learning arenas connected to projects in which civil society groups, government, and volunteers collaboratively engage in environmental stewardship, such as pond restoration to create dragonfly habitat in Japanese cities, indigenous species restoration at the Edith Stephens Wetland Park in Cape Flats, South Africa, and urban community gardening in vacant lots and other degraded spaces in the USA. More formal restoration projects, such as the daylighting of the Cheonggye-cheon River in Seoul, South Korea, as well as botanic gardens that feature biological and cultural diversity, also integrate nature-based, cultural, historical, and science inquiry learning opportunities. Given that many urban environmental education projects are local in scope, partnerships with global initiatives such as the UN Education for Sustainable Development and the Convention for Biological Diversity Communication, Education and Public Awareness, and with NGOs, governments, and business, are needed to leverage these learning arenas to effect broader regional, national, and even global systemic change.

30.1 Environmental Education, Education for Sustainable Development (ESD), and Communication, Education and Public Awareness (CEPA): A Short Introduction

Growing out of a long tradition of nature study, and reflecting a growing concern about pollution and environmental degradation, environmental education was recognized as a critical factor in addressing environmental problems at the UN Conference on the Human Environment in Stockholm in 1972. Its goals focused on developing individual competencies to work toward solving problems, as articulated in the Belgrade Charter (UNESCO 1975) and ratified as the Tbilisi Declaration (UNESCO 1977), which states:

> The goal of environmental education is to develop a world population that is aware of, and concerned about, the environment and its associated problems, and which has the knowledge, skills, attitudes, motivations and commitment to work individually and collectively toward solutions of current problems and the prevention of new ones.

Twenty years after the Stockholm conference that defined environmental education, the UN Conference on Environment and Development, otherwise known as the Rio Earth Summit, articulated and mandated Education for Sustainable Development (ESD) as critical to all aspects of *Agenda 21*.[1] *Agenda 21* drew heavily from the

[1] http://www.un.org/esa/dsd/agenda21/res_agenda21_00.shtml

Earth Charter,[2] which used a highly consensual process involving civil society organizations to articulate sustainability principles, in preparation for the 1992 Rio Summit. Consistent with the notion of sustainability outlined at the Summit, which was proposed as a more just alternative to a single-minded focus on the environment without regard to other aspects of human well-being, ESD integrates environmental with social and economic concerns. *Agenda 21* Chapter 36 on ESD articulated four major thrusts that distinguish ESD from environmental education*: access to basic education for all; reorienting education to embrace principles, skills, values, and perspectives related to sustainability; public awareness and understanding; and training for the private, university, government, and NGO sectors* (Mckeown and Hopkins 2003). In short, whereas environmental education has focused predominantly on curriculum and activities whose aim is to change individual knowledge, attitudes, and behaviors, the intent of ESD was to effect far-ranging institutional change in educational systems. Such change would come about as government ministries of education proclaimed that sustainable development concepts, including equity, economic development, and environment, would be fused into national curricula (Hopkins 2012).

According to Hopkins (2012), ESD was intended not to be another "adjectival education" like environmental education conceived as an add-on to the school curricula, but rather to infuse all education with sustainability principles through a series of reports, assessments, and guides. Whereas the goals of ESD are clearly more encompassing than those generally associated with environmental education, many think of ESD as a more socially conscious form of environmental education, and some have pointed to a similar tradition of embracing equity and other social concerns in environmental education dating back to the 1970s (Monroe 2012). Lotz-Sisitka (2007) warns that regardless of one's particular viewpoint, one should avoid focusing on the differences between environmental education and ESD, as this may be counter-productive to progress in these related fields over the past 30 years.

ESD implementation suffered in the 10 years following the Rio Summit as a result of limited recognition by governments and lack of an international funding structure. However, it gained in prominence following the launch of the Decade of Education for Sustainable Development at the 2002 Johannesburg World Conference on Sustainable Development. ESD received a further boost at the 2005 mid-decade Bonn meetings, which were attended by ministers and other high level education officials from nearly 100 countries (Hopkins 2012). Further, according to a 2012 report evaluating progress of the Decade, ESD is intended to permeate multiple aspects of learning beyond the classroom, and that such "boundary crossing" to other spheres can be a source of educational innovation.

> The boundaries between schools, universities, communities and the private sector are blurring as a result of a number of trends, including the call for lifelong learning; globalization; information and communication technology (ICT)-mediated (social) networking education; the call for relevance in higher education and education in general; and the private sector's growing interest in human resource development. The resulting 'boundary crossing' is reconfiguring formal, informal and nonformal learning and changing

[2] http://www.earthcharterinaction.org/content/

> stakeholder roles and public-private relationships. This new dynamic provides a source of energy and creativity in education, teaching and learning, which itself provides a powerful entry point for ESD. (Wals 2012, p. 5–6)

A similar trend in environmental education of integrating multiple educational approaches across diverse settings, with an eye toward fostering educational innovation, can be found in environmental education (Krasny and Dillon 2014), including in cities (Kudryavtsev and Krasny in review).

In addition to spawning ESD, the 1992 Rio Earth Summit opened for signature the Convention on Biological Diversity (CBD), which now has over 190 affiliated parties.[3] Article 13 of CBD addresses education, which is carried out through its Communication, Education and Public Awareness (CEPA) initiative. CEPA goals include:

> Communicate the scientific and technical work of the Convention in a language that is accessible to many different groups;
> Integrate biodiversity into Education systems in all Parties to the Convention;
> Raise Public Awareness of the importance of biodiversity to our lives, as well as its intrinsic value.[4]

The CEPA Toolkit outlines a process for implementing a biodiversity communication and education campaign to support the National Biodiversity Strategy and Action Plans (Hesselink et al. 2007). Also, in collaboration with ICLEI Local Governments for Sustainability and the City of Cape Town, the CBD has produced an Evaluation Design Toolkit (Rosenberg et al. 2012), and the CBD website links to numerous biodiversity curricula and educational activities.

Navarro-Perez and Tidball (2012) conducted a literature review of biodiversity education to help inform the CBD agenda. They identified lack of an agreed upon approach for biodiversity education, biodiversity as an ill-defined concept, messaging inappropriate to engaging the public in recognizing biodiversity as a concern, and people's disconnect from nature as challenges to addressing CBD goals. Wals (2002) suggests leveraging this lack of agreement on the definition and importance of biodiversity as a tool to promote critical thinking and help students address normative issues as part of environmental education and ESD programs.

30.2 The "Urban" in Environmental Education, ESD, and CEPA

30.2.1 Urban Environmental Education and Learning Arenas in Cities

Kudryavtsev and Krasny (2012) compiled a history of urban environmental education in the USA dating back to an early 1900s practice of urban nature study, which continues today in city parks and other more natural urban settings. In the late 1960s,

[3] http://www.iejeegreen.com/index.php/iejeegreen/article/view/42/26

[4] http://www.cbd.int/cepa/

urban environmental education expanded from its focus on nature study to encompass the social concerns of urban residents, including those in low income and ethnic minority neighborhoods; thus issues of environmental justice, cultural diversity, poverty, and open space in cities were incorporated into environmental education programs at schools, churches, neighborhood councils, and community centers (Clark 1972; Verrett et al. 1990; Frank et al. 1994). Today, urban activities are incorporated into widely used environmental education curricula such as Project Learning Tree, Project Wet, and Project Wild, as well as in citizen science projects in which students and volunteers collect data on biodiversity and ecosystem services (e.g., the Great Sunflower Project,[5] Project Monarch Watch[6]). In addition, several national environmental education programs focus specifically on urban settings. These include Garden Mosaics (Kennedy and Krasny 2005), an intergenerational, non-formal science education program that takes place in urban community gardens and integrates lessons about biological and cultural diversity alongside stewardship and action (see case studies below), and Celebrate Urban Birds,[7] a network of urban community organizations that engage children in learning about city birds through art, data collection, and stewardship.

Zoos, natural history museums, botanic and city gardens, city parks, plazas adjacent to churches and government buildings, and other less formal green spaces such as school and community gardens and even ditches and canals, provide learning arenas for biodiversity education in cities (Bagarinao 1998; Kassas 2002; Gill 2011; Shwartz et al. 2012). Zoos are of particular interest given that over 600 million people (10 % of the world's population) visit zoos annually, and many zoos are located in cities and have a long history of biodiversity and conservation education; albeit the mission and education programs of zoos generally focus on conservation of rare and charismatic species not generally found in cities (Whitehead 1995; Geser et al. 2009; Anon n.d. a). In the USA, zoos participate in partnerships of universities, museums, NGOs, schools, and youth organizations to offer more locally-based education, often with a strong science-inquiry focus. For example, Prospect Park Zoo and Fordham University engage high school students in comparative studies of insect biodiversity in managed and less managed spaces in New York City,[8] and the American Museum of Natural History has partnered with the Bronx Zoo to involve students in self-guided scientific investigations of urban biodiversity using the web and mobile devices.[9]

Botanic gardens are visited by 200 million people each year, and often include collections of native species, thus offering important learning arenas for biodiversity (Willison 2006). Similar to zoos, they partner with community organizations and schools to tie learning opportunities at the formal gardens to issues facing the surrounding community (Wals 2002). For example, Kirstenbosch Botanic Garden

[5] http://www.greatsunflower.org

[6] www.monarchwatch.org/

[7] http://www.birds.cornell.edu/celebration/

[8] http://fordhamsustainability.wordpress.com/project-true/

[9] http://www.amnh.org/news/tag/urban-biodiversity-network/

in Cape Town, South Africa developed 46 indigenous gardens at schools in the Cape Flats townships (see case studies below).

City parks also can play an important role in urban environmental education. For example, the Sundarvan Nature Discovery Centre, one of multiple programs associated with the Centre for Environmental Education in India, engages youth in nature study in the city of Ahmedabad. Activities include nature hiking, snake ecology awareness, and bird watching. Also associated with the Centre for Environmental Education, the Nandanavanam project in Hyderabad conducts teacher workshops on nature education, and in collaboration with a city park, has developed a brochure describing a pond as a biodiversity hub in the center of the city.[10]

Museum exhibits often focus on biodiversity and more recently, ecosystem services. In Stockholm, Sweden, the non-profit Albaeco organized the exhibition "Manna – Food in a New Light," which explains the provisioning ecosystem service of food production. The exhibit has been on tour since 2004 nationally and internationally attempting to reach urban audiences with a message about where their food comes from.[11]

Restoration practices of citizen activists, non-profits, and municipal governments provide arenas for active learning that contributes directly to sustainable management of urban biodiversity and ecosystem services. Urban restoration projects focusing on degraded or even paved over rivers, derelict transportation corridors, and neglected plots of land are becoming increasingly common, incorporating novel landscape features and learning opportunities. Examples include large-scale urban redesign projects such as the resurfacing of the buried Cheonggye-cheon River in Seoul, South Korea (see case study below, Sect. 30.6.3), and the conversion of elevated railroad beds in Paris and Manhattan to landscaped promenades. Smaller-scale efforts have the potential for more hands-on involvement of local residents, and include restoring ponds for dragonfly and fish in Japanese cities (see case study below, Sect. 30.6.2), reintroduction of oysters and fish into the Bronx River in New York City, and conversion of vacant lots to community gardens in cities across the USA and Canada (Krasny and Tidball 2012). These restoration projects linked to civic engagement are a relatively recent trend in urban planning and environmental activism (Sirianni and Friedland 2009) that create new kinds of informal learning arenas in cities. For more information on restoration ecology in an urbanizing world, see Chap. 31.

30.2.2 ESD and Urban Issues

As ESD has emphasized institutional change at the national level rather than specific programs or curricula, it has had less of a focus on specific settings such as cities.

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[10] http://www.ceeindia.org/cee/nature.html

[11] http://www.mannautstallningen.nu/about_manna.htm

Although Sustainable Urbanization[12] is included as one of 12 ESD themes, social rather than biodiversity or ecosystem processes in cities are emphasized. The Sustainable Urbanization theme states:

> Learning to live together sustainably in cities is one of the most important educational challenges of our time. This requires a focus on:
>
> * Creating a quality learning and educational environment that promotes sustainability;
> * Providing lifelong learning opportunities in cities;
> * Teaching tolerance and mutual understanding in urban societies;
> * Enabling children and youth to learn to live and participate in urban life;
> * Enhancing learning to create inclusive societies in inclusive cities;
> * Developing learning in all its diverse forms.

Despite the relative lack of attention to urban biodiversity and ecosystem processes in ESD, local sustainability education initiatives inspired by ESD and by the *Earth Charter* encompass urban issues. Local initiatives are also consistent with *Agenda 21* Chapter 28 ("Local Agenda 21 – LA21"), which calls for local action to address sustainable development.

In one local effort, the city of São Paulo in Brazil conducted a series of colloquia for teachers aimed at infusing the *Earth Charter* throughout its school system (Inojosa 2010). Topics addressed included the interconnection of the community of life, cultural diversity, the throw-away society, economics, peace and conflict resolution, and ecopedagogy. The teachers also participated in local urban treks, "seeking to observe everything that could be transformed to make urban life and community more sustainable" (Inojosa 2010, p. 240). Following the colloquium series, the teachers worked with a million students in local sustainability activities spanning garden installation, street tree planting, and theatre and music.

In another ESD-inspired urban initiative, geography students at the University of Teacher Training in Zurich, Switzerland identified, assessed, and shared with others (via a field trip) urban examples of positive sustainability practices, as well as practices that have "underutilized sustainability potential" (e.g., busy roads) (Odermatt and Brundiers 2007, p. 44). The latter were referred to as "sustainability fallows," defined as "places where the full potential of sustainable development hasn't been fully realized yet" (ibid, p. 43). This idea is consistent with a movement toward asset-based approaches to urban environmental education, a notion we return to later on in our discussion of civic ecology practices and related learning. In that the Zurich effort did not integrate the ESD focus on systemic level change, it illustrates a trend in ESD that is more closely aligned with environmental education (cf. Monroe 2012).

Even though its origins preceded ESD, UNESCO's *Growing Up In Cities* project reflects *Agenda 21's* participation principles (Chawla 2001). Initiated in the 1970s and since implemented in multiple cities globally, this project engages young people in participatory action research and planning for the future of their city. This and other programs focusing on youth participation in policy and planning represent

[12] http://www.unesco.org/new/en/education/themes/leading-the-international-agenda/education-for-sustainable-development/sustainable-urbanisation/

an important trend in urban environmental education consistent with ESD principles (Chawla 2001; Lane et al. 2005; UNESCO 2007).

An example of urban ESD in higher education comes from Nagoya, Japan's Open University of the Environment, which was created in 2005 with the goal of positioning Nagoya City as a global center of excellence in sustainability (Chikami and Sobue 2008). As one member of a local consortium that received United Nations University accreditation as a Regional Centre of Expertise for ESD, Open University is part of an international network of formal, non-formal, and informal education organizations that are engaged in ESD. The University's unique structure positions it well to address systemic change within the city. While it does not maintain a physical campus, it offers over 100 courses engaging 20,000 residents in using the natural, social, human and historical resources of the city as an arena for sustainability learning. In that it reports directly to an executive committee chaired by the Mayor of Nagoya and maintains strong partnerships with business and civil society institutions, the university has the potential to effect institutional change consistent with the intent of ESD. The Open University also participated in a successful bid to attract the Conference of the Parties (COP 10) to the 2010 Convention on Biological Diversity (CBD). It was at this Convention that an important development transpired intended to foster a social-ecological approach to urban planning — CBD member governments drafted and adopted the Urban Biosphere initiative (URBIS) principles.[13]

30.2.3 CEPA's Commitment to Urban Education

The CBD CEPA initiative has a strong focus on learning in cities. For example, the 2012 CEPA Evaluation Design Toolkit, developed in cooperation with the international association ICLEI Local Governments for Sustainability and the City of Cape Town, focuses exclusively on urban case studies including green school audit programs in Cape Town; Edmonton, Alberta's Master Naturalist Program; Nagoya Open University for the Environment; and a project to reintroduce howler monkeys in São Paulo, Brazil (Rosenberg et al. 2012). CEPA's commitment to urban biodiversity is consistent with the URBIS agreement reached at Nagoya COP10, which creates a recognition system for cities that develop a social-ecological systems approach to urban planning for biodiversity.

30.3 Cities Provide Unique Learning Arenas to Support Stewardship of Biodiversity and Ecosystem Services

Locally-initiated, collective stewardship practices in cities (i.e., civic ecology practices) (Krasny and Tidball 2012), including those designed to convert vacant lots to community gardens, remove invasive plants from city parks, restore degraded streams

[13] http://www.ilgbc.org/download/files/URBIS%20Declaration_1.pdf

and estuaries, and steward urban forests, recognize degraded lands and waters as potential assets, or "sustainability fallows" (Odermatt and Brundiers 2007). They invite local engagement in environmental and community stewardship while providing unique learning opportunities in cities. Several studies provide evidence of the contributions civic ecology practices make to biodiversity and ecosystem services (for more general discussion on urban ecosystem services, see Chap. 11). For example, community and allotment gardens contribute to food production (Lawson 2005), pollinator populations (Andersson et al. 2007; Barthel 2006; Strauss 2009; Ernstson et al. 2010a), and cultural ecosystem services including education (Fusco 2001; Krasny and Tidball 2009b) and social connectivity (Slater and Twyman 2003; Saldivar and Krasny 2004); and urban tree planting contributes to ameliorating the urban heat island effect (Pataki et al. 2011) as well as to cultural ecosystem services and community resilience (Tidball 2013).

Civic ecology practices can be arenas for learning about urban biodiversity and ecosystem services, and civic ecology education programs developed around these practices can contribute to managing for social-ecological systems resilience (Krasny and Tidball 2009a) (see also Chap. 33). These learning arenas demonstrate that people both are part of ecosystems (hence the term "social-ecological systems") and can create something that is of value for both the people and other living organisms in those systems.

Examples of civic ecology education (Krasny and Tidball 2009a) include the Garden Mosaics[14] program, which provides opportunities for youth to learn about science, culture, action, and the environment through working alongside elder community gardeners (see case study below, Sect. 30.6.4); and after-school and summer programs conducted by the Bronx community organization Rocking the Boat,[15] which engage youth in ongoing oyster restoration in New York City's Hudson River estuary. In Japanese cities, young people and adults have become engaged in pond and river restoration to provide habitat for dragonflies and fish (Primack et al. 2000; Kobori 2009; Anon. n.d. b) (see case study below, Sect. 30.6.2).

Civic ecology education has several additional implications for urban environmental and biodiversity education and ESD. Importantly, it integrates social and cultural issues that are foundational to ESD (Krasny and Tidball 2009a, in press). Further civic ecology education addresses concern about the potential counter-productive outcomes of environmental education programs that focus solely on negative messaging about environmental problems (Dickinson 2009), through situating learning in positive expressions of community engagement and environmental stewardship, often in what are perceived as highly degraded urban environments. Moreover, youth may be motivated by the opportunity to contribute as valued members of a community (Olitsky 2007); by seeing how their actions lead to positive changes in their environment (Chawla 2008); as well as by opportunities to link their cultural ways of knowing to science learning, such as might occur when the local knowledge

[14] www.gardenmosaics.org

[15] www.rockingtheboat.org

of immigrant and other adult community gardeners or tree planters is incorporated into the learning activities (Moll et al. 1992; Aikenhead 1996; Shava et al. 2010).

Civic ecology education is emerging as one approach to urban environmental education. However, because civic ecology education emphasizes locally-initiated, small-scale stewardship practices as learning arenas, it lacks a focus on strategic change at the national or global level, as called for by ESD. We return to this issue of strategic impact in Sect. 30.7.2 on policy toward the end of this chapter. But first we describe research and learning theories that support the notion of active engagement in civic ecology and similar hands-on practices as contexts for learning, following which we present four case examples of learning arenas for restoration and stewardship of urban biodiversity and ecosystem services.

30.4 Research on Urban Environmental Education

Much of the research about urban environmental education and ESD programs is descriptive or qualitative (e.g., open-ended interviews of participants or educators) and thus fosters in-depth understanding of pedagogical approaches and the experiences of participants, but provides only initial insights into program impacts (e.g., Bouillion and Gomez 2001; Fusco 2001; Mordock and Krasny 2001; Doyle and Krasny 2003; Krasny and Tidball 2009b; Krasny et al. 2009; Morgan et al. 2010; Wals and van der Waal 2014). More quantitative studies can provide stronger evidence of outcomes for participants, whereas those that combine quantitative and qualitative methods provide both an in-depth understanding of programs and strong evidence of their impacts (e.g., the work of Kudryavtsev 2012; Kudryavtsev et al. 2012; on sense of place in urban environmental education).

Due to various constraints, the quantitative studies often use a quasi-experimental design with control groups drawn from non-participants; given that individuals in the treatment and control groups have chosen whether or not to participate in the educational programs, these studies lack random assignment to treatment and control. Other studies use only pre/post- tests and lack controls, and thus cannot definitively say any effect is due to the program rather than something occurring outside the program. Barnett et al. (2006) used both a pre/post- test and control group research design to test the outcomes of participation in an urban environmental science inquiry program and found positive results related to science interest and understanding among girls and boys, and to sense of stewardship among boys. In a second pre/post- survey study, Barnett et al. (2011) found changes in science self-efficacy (feeling as if one can achieve in science) and ecological mindset related to a 2-week science inquiry program. This study also included qualitative interviews which revealed that the field experience resulted in more positive perceptions of the urban environment and students' ability to positively impact the environment. This result is consistent with a study conducted by Kudryavtsev et al. (2012), which used a pre/post- test, controlled research design to determine the impact of urban environmental education programs on sense of place among youth in the Bronx borough of New York, USA, and found that participants increased the ecological meanings that they attributed to their highly urban neighborhood, including meanings related to local wildlife or biodiversity. Shwartz et al. (2012) integrated quantitative alongside qualitative methods in a study of a gardening programs in Paris, France, and noted positive impacts on short-term knowledge, awareness, and interest related to urban biodiversity in the qualitative interviews; however, the study was limited in that it lacked pre-treatment measures and the control group differed from their treatment group. Using pre/post- tests of participants in a zoo conservation camp, Kruse and Card (2004) found positive outcomes related to environmental attitudes, knowledge, and behaviors, with the degree of change correlated with the amount of hands-on animal husbandry experiences in the various camps. In general, these studies provide evidence of positive impacts of urban environmental education and related field science inquiry experiences on participant environmental attitudes, awareness of urban nature, science understanding, and self-efficacy, with an increased effect correlated with degree of involvement in hands-on, field-based experiences. Given the diverse goals of various urban environmental education programs related to biodiversity (e.g., understanding of science related to biodiversity, changing attitudes toward biodiversity in cities, acting to steward urban biodiversity, or even changes in the social-ecological system per se), defining specific program objectives for research and evaluation is critical (Kudryavtsev and Krasny in review). We address this and other challenges facing researchers assessing learning about urban biodiversity in the final section of this chapter.

30.5 Learning Theories

According to Lundholm and Plummer (2010), learning is a multi-faceted process encompassing cognitive, social, and emotional aspects. Regardless of the context in which learning takes place (e.g., in a classroom, zoo, or civic ecology practice), cognition and understanding are influenced by the way an individual perceives and interacts with the social and institutional setting. In general, environmental learning serves the purposes of fostering content understanding, raising awareness, promoting moral understanding, and developing systems and critical thinking to enable participants to take action as citizens, voters and consumers. Further, scholars whose work integrates learning theory with resource management, organizational behavior, and social-ecological systems describe how learning occurs at the group or organizational in addition to individual level, leading to changes in management practice that directly impact institutions and the environment (Blackmore 2007; Schultz and Lundholm 2010).

In the sciences, we often assume that learning is about transmission of knowledge or skills to students in classrooms and other settings. However, many learning theorists focus less on the more passive process of acquiring knowledge through listening to lectures and reading, and more on the active role of the student in constructing knowledge, interacting with his/her environment, and reflecting on his/her experiences in the process of learning.

Civic ecology practices such as community or allotment gardening, where there is an existing community of practice as well as a rich context for learning that integrates stewardship, social connectivity, advocacy, and sometimes cultural diversity, lend themselves to theories that describe learning as an outcome of interaction with the socio-cultural and bio-physical environment (Sfard 1998; Illeris 2007; Alexander et al. 2009). Such theories variously emphasize learning as constructing knowledge through processes of assimilation and accommodation (Piaget 1952/1936) or constructivism; learning as moving from an inexperienced to skilled member of a community of practice (Lave and Wenger 1991; Wenger et al. 2002; Rogoff et al. 2003); the larger social, cultural and historical contexts of learning (i.e., socio-cultural theory) (Lemke 2001); learning as embedded in the more immediate social and environmental context (i.e., situated learning) (Brown et al. 1989); and the importance of reciprocal interactions among learners' behaviors, capabilities, and environment (i.e., social learning) (Bandura 1977). We group all of these approaches under the broader term social learning, which is considered foundational to ESD (Wals 2007, 2012; Wals et al. 2009). Further, learning may be conceived as reciprocal interactions and changes brought about in the learner and other components of an activity system (Engeström 1999) or more generally a social-ecological system, which we refer to below as ecological theories of learning (Chawla 2008). Despite their different emphases, all these interactive theories have in common their ability to help us think about alternatives to conceptions of learning as an individual activity of knowledge acquisition with little reference to the socio-cultural and environment context.

We provide a short overview of interactive learning theories below with the purpose of broadening thinking among the policy and scientific research communities about how people may learn through participation in civic ecology (Krasny and Tidball 2012), adaptive co-management (Armitage et al. 2007), and related practices that seek to enhance urban biodiversity and ecosystem services. Thus, we focus largely on out-of-school (non-formal), hands-on learning linked to collective stewardship practice. Our discussion of interactive processes in learning is not intended to infer that other kinds of learning, e.g., acquisition of content knowledge, are unimportant, but rather to introduce perspectives on learning that are consistent with social-ecological systems thinking (Fazey et al. 2007; Krasny et al. 2009, 2011; Tidball and Krasny 2010, 2011) and that may be less familiar to our readers.

30.5.1 Constructivism

The constructivist theory of learning originates in the work of developmental psychologist Jean Piaget starting in the 1920s, with a central focus on the ways children construct understanding of phenomena they encounter in everyday life, as well as of concepts and theories they are exposed to in and out of school. Piaget's interest concerned the process of conceptual development – the ways in which intellect and cognition develop – and this constructional process is described in terms of assimilation and accommodation. This means that individuals construct knowledge by drawing on their existing understanding and in so doing both integrate (assimilate) new information with existing thinking and change (accommodate) their understanding. Thus, learning as a process of assimilation and accommodation is a consequence and outcome of the individual's interactions with others and the environment (Piaget 1952/1936).

Constructivist theory researchers today pay attention to and investigate not only the learner (including his/her prior knowledge, interests, emotions, and goals) but also content and context (Lundholm in press). A review of empirical studies conducted from 1990–2011 on students' conceptions and learning about the environment concluded that environmental learning means learning about complex phenomena (Lundholm and Davies 2013). It means linking nature, society and the individual/self, as for example connecting ecosystems services such as food production with economics (i.e., price and willingness to pay), issues of water quality with legislation, or fisheries with co-management. Any such link between nature, society and individual will not be unidirectional, and thereby the complexity of these phenomena is real and becomes a potential challenge to grasp (Lundholm and Davies 2013). Further research is needed to investigate the kinds of learning challenges presented by acquiring systems thinking, and exploring ways that education and communication can enhance such learning.

Constructivist learning theory suggests the following principles relevant to fields of communication, pedagogy, and environmental education: (1) learners (young as well as adults) build on their existing knowledge when encountering new information, (2) learning is dependent on learners' interest and goals, and (3) learning takes time (Vosniadou 2001; Vosniadou et al. 2008; Lundholm 2011). Also, the learning process is complex, encompassing people's emotions and their affections as they engage with environmental content (Rickinson et al. 2009; Lundholm et al. 2013; Wals and Dillon 2013). Together this implies the need for awareness as to how learners interpret environmental information and how they engage or disengage with content and topics.

30.5.2 Social Learning Among Individuals

In applying Lave and Wenger's (1991) notion of learning as participation in communities of practice, i.e., learning that occurs through the interactions of novice and more experienced participants in a common profession such as teaching or common practice such as environmental stewardship, questions arise as to how to structure the learning experience so as to foster increasingly skilled levels of participation over time. Hogan (2002) found that proper mentoring and scaffolding by adults is critical to learning among secondary school students working in a community environmental organization, and Bouillion and Gomez (2001) described a sequence of progressively more complex learning experiences for primary school students in Chicago focused on riverbank restoration, which resulted in student learning and improvements in the local community and environment. This work suggests that rather than simply plopping a young person into an ongoing civic ecology or other community of practice, structured and progressively more challenging opportunities for interacting with experienced adults who actively model the practices, coach novices, and provide scaffolding are critical in enabling a young person to move from being an observer of a practice to a peripheral participant (someone who participates in but has not yet mastered the practice), and then to a full or skilled participant (Brown et al. 1989; Rogoff et al. 2003; Gauvain 2005).

Research also suggests that students learn science through participating in authentic research communities (Brown et al. 1989; Crawford 2012), such as citizen science programs in which volunteers collect data that contribute to larger scientific studies (Bonney et al. 2009). Examples of citizen science programs that contribute to biodiversity monitoring and learning abound (e.g., the extensive array of bird monitoring projects conducted by the Cornell Laboratory of Ornithology[16]); a smaller set of projects collect ecosystem services data and foster related learning (Krasny et al. in review). These include the Great Pollinator Project, which focuses on monitoring bee populations and thus provides an indirect measure of the regulating service pollination (AMNH 2012), and the Lost Ladybug Project, which provides an indirect measure of the regulating service pest control as carried out by predatory insects (Anon. 2011). In an example more akin to ESD, O'Donoghue engaged communities in southern Africa facing a cholera epidemic in conducting simple experiments of water contamination and in workshops to discuss their findings, an approach he refers to as the Open Process Framework (Taylor 2010).

30.5.3 Social Learning Among Organizations and Groups

Natural resources and adaptive co-management scholars have expanded on the notion of individual learning as increasing levels of participation in a community of practice, to suggest that learning also may be an organizational or group process that occurs as an outcome of specific forms of participation in resource management (Armitage et al. 2008). In this context, social learning is defined as the process by which stakeholder interactions move beyond participation to encompass concerted action that brings about policy change, or more generally a collaborative process among multiple stakeholders aimed at addressing management issues in complex systems (Schusler et al. 2003; Keen et al. 2005; Blackmore 2007; Ison et al. 2007; Mostert et al. 2007; Pahl-Wostl et al. 2007; Plummer and Armitage 2007; Plummer and FitzGibbon 2007; Fernandez-Gimenez et al. 2008). The ability to take concerted action depends on gaining adequate knowledge through less structured hands-on experiences and through more intentional experimentation directed at understanding the impact of a management practice, as well as through discussion and reflection on the outcomes of such experiential learning and experimentation (Armitage et al. 2008).

[16] http://www.birds.cornell.edu/citscitoolkit

Critical reflection, along with collaboration and communication, are core concepts and ingredients for enhancing organizational learning; however, they may be hampered within organizations that promote conformity and reinforce power relationships (Marsick et al. 2000). Despite these challenges, Schultz and Lundholm (2010) present ample evidence of organizational learning among local stakeholders and managers in UNESCO's Man and the Biosphere reserves program.

In resource management contexts, social learning can entail engagement in participatory decision-making, such as simulation modeling (Pahl-Wostl and Hare 2004), participatory map mapping (Ison et al. 2007), or search conferences (Schusler et al. 2003), as well as direct participation in hands-on stewardship activities. For example, volunteer efforts to restore degraded prairie and savannah habitats in Chicago demonstrate how, through a series of informal planting and land management experiments (e.g., controlled burns to suppress invasive species), lay people and scientists were able to continually improve upon means of managing their social and biophysical environment for biodiversity and cultural ecosystem services (Stevens 1995; Jordan 2003; Moskovits et al. 2004). Organizational learning may also occur in the private sector. Cramer and Loeber (2007) describe a multi-level social learning process among participants in a Dutch government initiative to help businesses develop strategies that balance "people, planet, and profit," and Hanson et al. (2012) outline a process for businesses to analyze their dependence and impact on ecosystem services.

30.5.4 Ecological Perspectives on Learning

The constructivist and social learning theories described above emphasize how learning occurs through interactions of the learner with the social and bio-physical environment, during which both the learner and environment experience change. These notions of reciprocal change are more explicitly addressed by an ecological perspective on learning, in which the learning environment or context, including tools, practices, and people, "afford" learning opportunities and thus are referred to as affordances or affordance networks (Greeno 1998; Barab and Roth 2006; Chawla 2008). However, in order to actually learn from these affordances, the learner must demonstrate certain behaviors, referred to as his or her effectivity set, which may in turn generate new affordances in an expanding cycle of learning (Barab and Roth 2006). Echoing this notion of learning as reciprocal change, Pahl-Wostl (2006) states that social learning within the context of resource management "assumes an iterative feedback between learners and their environment, i.e., the learner is changing the environment, and these changes are affecting the learner." Delving more deeply into how this change occurs, activity theory posits that learning emerges through interactions among six elements of an activity system: the subjects (participants), objects (e.g., garden or other social-ecological system that is the focus of practice), community (the wider community impacted by the activity), tools (e.g., seeds), rules (e.g., allowing removal of invasive species but not of native species), and division of labor (i.e., roles of participants and other community members) (Engeström 1987). Similar to ecosystems, the activity systems that afford learning opportunities have boundaries, which limit the interactions between the learner and other elements of the system. These boundaries may expand, as when learners are faced with a dilemma, and respond by reflection and creating innovative means of solving the problematic situation, which in turn leads to new ways of interacting with the social and bio-physical environment (Engeström 1987; Engeström et al. 1999). Describing how this might occur, Engeström (2001, p. 137) states:

> Activity systems move through relatively long cycles of qualitative transformations. As the contradictions of an activity system are aggravated, some individual participants begin to question and deviate from its established norms. In some cases, this escalates into collaborative envisioning and a deliberate collective change effort. An expansive transformation is accomplished when the object and motive of the activity are reconceptualized to embrace a radically wider horizon of possibilities than in the previous mode of the activity.

In an example of an expanding cycle of learning relevant to urban environmental education, youth and adults engaged in urban community forestry may at first operate within a bounded "tree planting system" and face a dilemma when soil compaction and tree vandalism cause tree mortality. In response to this dilemma, the youth and adults seek out more effective methods of tree planting and devise means to involve local residents in the planting efforts, while continuing to monitor mortality. Eventually, the dilemma, the changes in how trees are planted, and the ongoing monitoring lead to critical reflection that results in a transformation of the original tree planting activity system into a new activity system focused on influencing policy makers to support urban community forestry (Tidball and Krasny 2011). In this way, similar to components of ecosystems, activity systems interact with and are nested in larger systems (cf. Wimberley 2009). An expanding cycle of action, dilemma, and adaptation has parallels with the adaptive cycle of growth, disturbance, and reorganization that is foundational to social-ecological systems resilience thinking (Holling and Gunderson 2002; Krasny and Roth 2010).

In their focus on questioning fundamental ways of doing business, and on dilemmas or "surprises" coupled with critical self-reflection, social and ecological perspectives on learning reflect Argyris and Schon's (1978) notions of multiple loop learning. Multiple loop learning moves from immediate problem solving to a process of questioning and reflection. For example, stakeholders who monitor water quality learn about pH and other measures of the health of a body of water; such learning is referred to as single loop learning. Stakeholders who not only collect data but also question their data collection and management goals and procedures engage in second loop learning. Finally, stakeholders who not only question the management procedures but also the assumptions behind the management paradigm, e.g., the differential value placed on input from various stakeholders, engage in triple loop learning. While difficult to facilitate, such multiple loop learning is critical to adaptive co-management (Armitage et al. 2008).

In short, both social theories and ecological perspectives on learning define learning as successful participation and increasing possibilities for action in a social-ecological system (Barab and Roth 2006). They refer to learning systems comprised of individuals interacting with each other and with elements of the biological and physical environment. Through these interactions, the individual, the broader community of individuals with whom he or she interacts, and the biological and physical environment are transformed.

30.6 Case Studies of Learning Arenas for Managing Urban Biodiversity and Ecosystem Services

The research and theory described in the previous sections support tenets of ESD's Sustainable Urbanization theme, including lifelong learning, learning as participation in urban life, and learning in diverse contexts, as well as a long-standing tradition of participatory processes in environmental education (Reid et al. 2008; Schusler et al. 2009; Læssøe 2010; Læssøe and Krasny 2014; Læssøe and Pedersen 2014). Participation in urban stewardship and management also may reflect exemplary approaches to biodiversity education, including accurate observation, identification and monitoring of backyard biodiversity, habitat design, and learning about how humans both depend on and shape biodiversity (Van Weelie and Wals 2002).

In this section, we describe four urban environmental education programs chosen because they (1) reflect the learning theories discussed above through presenting significant opportunities for participation and interaction, and (2) are situated in learning arenas that demonstrate the positive role of humans in restoring biodiversity and degraded social-ecological systems in cities. Thus, learning takes place through hands-on participation in practices that restore both environmental and community value, or in sites where such restoration has already occurred. The first case integrates multiple learning arenas in Cape Town, South Africa, including a new and an established botanic garden in the central part of the city, and civic ecology practices in the Cape Flats townships. The second example comes from pond restoration projects to restore insect and fish habitat in Japanese cities. Next we turn to the Cheonggye-cheon River restoration project in Seoul, South Korea. Finally, we highlight the Garden Mosaics project, which originated in North America and has been adapted for use in other parts of the world.

30.6.1 Cape Town, South Africa

Soul Shava

Situated in the Cape Floral Region biodiversity hotspot and UNESCO World Heritage Site, Cape Town is home to a wealth of biodiversity preserves with significant outreach and educational efforts.[17] We feature three Cape Town learning arenas here. For an extended social-ecological analysis of Cape Town, see the Chap. 24 local assessment.

[17] http://whc.unesco.org/en/list/1007

30.6.1.1 Green Point Biodiversity Showcase Garden

A legacy project of the 2010 World Soccer Cup and adjacent to Cape Town Stadium, Green Point Park houses the Biodiversity Showcase Garden. The redevelopment of the Green Point Common into an urban park is one of the City's Local Action for Biodiversity projects.[18] The immediate goal of its Biodiversity Showcase Garden is to "showcase the amazing diversity of plants and animals in the Greater Cape Town area," whereas its ultimate aim is that "the people of Cape Town will learn to value our local biodiversity and feel inspired to make changes in the way they live to ensure that future generations can also benefit from it."[19]

The Biodiversity Showcase Garden features over 300 local Cape plant species, along with animal sculptures, interactive signage, demonstration gardens that offer suggestions on how to grow indigenous plants in your home garden, and displays of locally indigenous Khoikhoi plant use. It is separated into People and Plants, Wetlands, and Discovering Biodiversity thematic sections.[20] To complement the learning that takes place through experiences in the garden, the City of Cape Town produced a nearly 100-page lesson plan and activities guide for primary school children (Hitchcock 2011). Encouraging follow-up activities after a one-time experience is consistent with research that demonstrates the importance of repeated experiences in bringing about learning and changes in behaviors (see Sect. 30.4, Research on Urban Environmental Education, above).

30.6.1.2 Kirstenbosch Botanic Garden

Situated on the opposite side of Table Mountain from Green Point Park, the world-class Kirstenbosch Botanic Garden features an extensive collection of indigenous South African flora, including the unique natural vegetation of the Cape Floristic Region/Kingdom planted in a naturalistic setting. In 2004, the Cape Floristic Region, including Kirstenbosch Botanic Garden, was declared a UNESCO World Heritage Site making it the first botanic garden in the world to be included in such a designation. The garden also features a vast array of onsite education and school and community outreach programs. Onsite offerings for school groups encourage learners to discover the environment through careful observation, and recording and interpretation of data. Biodiversity lessons focus on fynbos, afromontane forest, and succulent species indigenous to the Western Cape; evolution of mosses, ferns, gymnosperms and angiosperms; and global warming and waste impacts on biodiversity as well as personal response to these issues.[21] School teachers

[18] http://www.iclei.org/index.php?id=lab

[19] http://www.sa-venues.com/attractionswc/biodiversity-garden.htm

[20] http://blog.sa-venues.com/provinces/western-cape/biodiversity-garden/

[21] http://www.sanbi.org/programmes/education-hcd/kirstenbosch-nbg-education/biodiversity-education

accompanying learners visiting the garden are exposed to practical activities that can be used in their own school gardens or neighboring natural areas, thus enabling longer-term experiences for the students.

Kirstenbosch's Outreach Greening program aims to: establish indigenous, water-wise, school and community gardens; encourage ecological awareness and environmental responsibility; develop gardening skills to enable economic empowerment and local environmental action; promote the educational value of indigenous plants and gardening; and develop partnerships between communities and organizations. Through its Outreach Greening Schools program, botanic garden staff work with schools for a minimum of 3 years to establish and maintain indigenous and vegetable gardens on school grounds. The teachers attend workshops to build their capacity to create interpretive signs and develop curriculum-linked lessons that can be facilitated in their school gardens. Kirstenbosch also facilitates Community Greening Projects to establish community indigenous gardens.[22]

30.6.1.3 Cape Flats Nature

Moving from the Kirstenbosch Botanic Garden down the slopes of Table Mountain and inland to the Cape Flats, one encounters a network of small nature preserves dotting a 30 km stretch of township settlements. In the early 2000s, the University of Western Cape Environmental Education and Resources Unit developed a series of resources and workshops focused on local urban biodiversity to take place at the Cape Flats Nature Reserve. Secondary students were provided opportunities to engage in field research on the impacts of urbanization and ecology of the Cape Flats, including population and community ecology and ecosystem structure and function. Primary school learners participated in guided walks in the Reserve, which incorporated sensory awareness activities.[23] The reserve is also used by the university as a base for ecological teaching, environmental education, and research.

In 2002, the South African National Biodiversity Institute partnered with a consortium of NGOs and government agencies (City of Cape Town, Table Mountain Fund, World Wildlife Fund–South Africa, and the Botanical Society of South Africa) to launch the Cape Flats Nature initiative. Its goal was to increase the value of a chain of nature reserves in the Cape Flats to the surrounding communities, through helping with community upliftment, building organizational capacity, and creating education and employment programs. This led to the communities engaging more actively and positively with the sites, and thus developing a stronger appreciation for their conservation (B Pitt, personal communication). Subsequent environmental education taking place in nature reserves in the Cape Flats has included programs that use nature immersion experiences to help youth address personal challenges; physically challenging hikes to foster leadership

[22] http://www.sanbi.org/programmes/education/outreach-greening-programme/kirstenbosch-nbg

[23] http://www.bcb.uwc.ac.za/eeru/EEprograms/default.htm

Fig. 30.1 Youth and community members planting an herb spiral at Edith Stephens Wetland Park in the Cape Flats, South Africa (Photographed by and published with kind permission of © Sam Huckle 2013. All Rights Reserved)

skills and learning about history, flora and fauna; biodiversity monitoring; school and community gardening and tree planting; clean-up of polluted areas; and programs that encourage residents to conserve water (Pitt and Boulle 2010). Other projects include the rehabilitation of the Edith Stephens Wetland Park, plant monitoring and fire-awareness in the Harmony Flats Nature Reserve, the consolidation of hiking trails and the monitoring and reintroduction of animal and bird species in the Wolfgat Nature Reserve, and an alien-vegetation clearing project in the Macassar Dunes (Fig. 30.1).[24]

What marks all the Cape Town biodiversity education projects is the pride they demonstrate in preserving the Cape Floral Region's unique biodiversity, while at the same time integrating local cultural and historical perspectives, ranging from traditional uses of plants to the political reality of post-apartheid South Africa struggling to address ongoing issues of poverty and injustice. Such integration of biological and cultural diversity is foundational to ESD.

[24] http://www.impumelelo.org.za/what-we-do/impumelelo-innovations-awards/2005/gold/cape-flats-nature-1

30.6.2 Japanese Cities Restore Urban Aquatic Systems and Biodiversity

Hiromi Kobori

In Japan, urban biodiversity education is integrated into ongoing initiatives to restore aquatic habitats, along with the dragonflies, fish, and other fauna that depend on ponds and streams. The principles of satoyama – a traditional land-use system characterized by a mosaic of agriculture, grasslands, woods, and wetlands that fostered greater diversity of plants and wildlife than nearby less managed forested areas (Kobori and Primack 2003) – provide guidance for restoration efforts. For an additional discussion of satoyama landscapes, see the Chap. 8 local assessment.

In Honmoku Citizens Park in Yokohama, people were not happy with a concrete-lined pond, which was home to ornamental fish but devoid of plants and frequented by only three common dragonfly species. In 1986, citizens' groups, scientists, and city government partnered to construct a winding stream with pools in both shady and sunny spots, and to shovel soil into the pond to create earthen banks for native aquatic plants. As more ponds were restored and created, 27 species of dragonflies migrated to the ponds from the surrounding environment. Traditionally dragonflies have held symbolic importance to the Japanese people, and soon school children and dragonfly aficionados were coming to the ponds to learn about nature. The visitors also helped steward the ponds and their inhabitants – they removed unwanted plants, dredged sediment from the ponds, and captured crayfish and foreign bluegill sunfish that prey on dragonfly larvae (Primack et al. 2000). What started as the restoration of one small pond has sparked a movement – 130 dragonfly ponds have been created, many serving as sites for the public to learn about and help steward nature. Various sectors in the city have been working together to catch, number, and release the dragonflies among restored and created ponds, thereby demonstrating that some ponds are ecological stepping stones for dragonflies, and together form an ecological network of dragonfly habitat in Yokohama, a city of 3.7 million inhabitants (Fig. 30.2).

A second Japanese project engaged university students in restoration of butterfly habitat. Importantly, in this and similar projects in Japan, participants have monitored project outcomes, sometimes adjusting their practices based on results. In addition to monitoring increases in butterfly populations, this project used pre/post- surveys and word association tests to evaluate the project's impact on student learning. The researchers found that students developed a concern for and interest in butterfly conservation and increased their proficiency in articulating concepts related to butterfly habitat (Kobori 2009).

The Japanese restoration projects provide examples of integrating participatory processes of stewardship with science inquiry. They also leverage the fact that particular species, such as dragonflies, hold cultural meanings in Japan, as well as the public's awareness of the need for active conservation if these species are to survive in the Japanese landscape. Finally, through Regional Centres of Excellence in ESD,

Fig. 30.2 Created dragonfly pond in the elementary school yard is used for monitoring dragonflies, for education in various subjects, and for fun (Photographed by and published with kind permission of © Kiichi Matsushita 2013. All Rights Reserved)

networks linking institutional and community stakeholders have enabled these local educational efforts to spread widely and be adapted to other localities (Kobori 2009).

30.6.3 Cheonggye-Cheon River Restoration: Seoul, South Korea

Eunju Lee

The Cheonggye-cheon Restoration Project created a 5.8 km landscaped greenway that runs alongside the revitalized Cheonggye-cheon River in Seoul, South Korea.[25] It involved daylighting a river that had been buried under city streets, and dismantling an elevated freeway above the former river corridor. The restored corridor runs from Seoul to an ecological conservation area outside the city, and is split into three zones marking the transition from an urban to a more natural landscape. The history zone includes the streambed and stones of historic bridges as decorative elements. The middle urban and cultural zone features waterfront decks, fountains, waterfalls, stepping stones for crossing the stream, and opportunities to wade in the water. The stream widens as it reaches the final zone, which is designed to look

[25] http://webarchive.nationalarchives.gov.uk/20110118095356/http:/www.cabe.org.uk/case-studies/cheonggyecheon-restoration-project

Fig. 30.3 People use stepping stones to cross the restored Cheonggye-cheon River in Seoul, South Korea (Photographed by and published with kind permission of © Cheonggye-cheon Museum 2013. All Rights Reserved)

overgrown and untamed, but sections of the pier and overpass remain as industrial memories. Because the stream's flow is intermittent, water levels are supplemented by pumping the Han River and by treated wastewater; the long-term goal is to include more treated wastewater as the city water treatment system improves (Fig. 30.3).

While focused largely on providing cultural ecosystem services to Seoul residents and tourists, the restored stream is also designed to channel flood and treated water, thus providing a regulating ecosystem service. In addition, the restoration project has greatly enhanced biodiversity along the stream corridor. In 2010, 5 years after the project's initiation, what was once a thruway now housed 25 fish species, 37 bird species, and 248 terrestrial insect species. Artificial features, such as rocks placed in the streambed to create riffles that aerate the stream and the roots of streamside vegetation, foster this biodiversity (Reed 2011).

The Cheonggye-cheon Museum located along the stream corridor not only commemorates the restoration of the river, but also presents its history, culture, and restoration process as part of Seoul's future vision of an environment-friendly, human-centered urban space.[26] In addition to exhibits and exhibitions centering on the stream and the urban development of Seoul, the museum sponsors educational programs for adults and children focusing on cultural and natural history, and restoration. Through these activities the museum hopes to deepen awareness and

[26] http://www.museum.seoul.kr/eng/eng_intro/eng_org/1173488_649.html

understanding of the stream and its restoration, and promote the museum as a cultural space central to the Cheonggye-cheon area. In a separate educational program, the Seoul Metropolitan Facilities Management Corporation holds an Eco Classroom focused on the Cheonggye-cheon ecosystem. Students monitor the animals and plants in the river with the help of both experts and lay citizens.

The Cheonggye-cheon Restoration Project has been widely cited as an ambitious and successful example of large-scale urban restoration for cultural ecosystem services. It serves as an inspiration to other cities seeking to transform neighborhoods plagued by traffic and associated environmental, economic, and community decline through ecological restoration. Such urban restoration projects, including the High Line Park in Manhattan and other rail bed to park conversions, offer important learning arenas for ecosystem services and biodiversity. For an additional detailed discussion of urban ecological restoration, see Chap. 31.

30.6.4 Garden Mosaics

Marianne E. Krasny

The Garden Mosaics program seeks to "connect youth and elders to investigate the mosaics of plants, people, and cultures in gardens." Learning activities take place largely in community gardens although the program can be adapted for school gardens (Kennedy and Krasny 2005; Krasny et al. 2005).[27] Consistent with the ESD focus on cultural diversity, Garden Mosaics activities emphasize learning from the traditional or practical knowledge of community gardeners. Community gardeners in the USA come from all walks of life, including immigrants from developing countries and African-Americans with roots in the rural southern states; similarly in South Africa and other countries, community gardeners are often immigrants or internal migrants to cities coming with rural, agricultural backgrounds. Through Garden Mosaics, these diverse gardeners share with youth the ways in which they have adapted agricultural practices from their homeland to highly urbanized settings, which the youth capture in gardener interviews and compile into Gardener Stories (Fig. 30.4).

Learning from the practical knowledge of gardeners is complemented by learning from science resources produced at Cornell University. For example, the program resources include Science Pages, or fact and activity sheets, that describe the biology, history, and uses of plants likely to be found in community gardens, as well as concepts and garden features such as biodiversity, soils, and insects. Educators guiding young people in conducting the Gardener Story interviews can use these pages to help the youth develop an understanding of the science and cultures associated with the practical knowledge shared by the gardeners. The program also encompasses activities to foster observation and data collection, including the Garden Hike and

[27] www.gardenmosaics.org

Fig. 30.4 Youth record a gardener's story in the Bronx, New York, USA (Photographed by and published with kind permission of © Alex Kudryavtsev, Cornell University Civic Ecology Lab 2013. All Rights Reserved)

Neighborhood Investigation activities, through which youth collect data on vegetables, soils, and the role of gardens in their community.[28]

Further, drawing on what they learn in their Gardener Story interviews and other investigations, as well as through the information and learning activities outlined in the Garden Mosaics Science Pages, youth conduct Action Projects to enhance their community. Thus the program is designed to facilitate science learning, intergenerational mentoring, cultural understanding, and community action to enhance biodiversity and to foster food production, cultural, and other ecosystem services. Garden Mosaics curriculum materials and training videos are available for free online, enabling any educator or parent to access and adapt the materials for their own setting.

In that community gardening is one form of civic ecology practice, Garden Mosaics is considered to be a civic ecology education program. Similar to the educational activities situated in urban nature reserves in South Africa, ponds in Japanese cities, and the Cheonggye-cheon River corridor in South Korea, Garden Mosaics provides a model for thinking about opportunities for embedding learning in ongoing community restoration and stewardship practices. Other civic ecology learning arenas include urban tree planting, invasive species removal in city parks, and oyster restoration in urban estuaries (Krasny and Tidball 2009a, in review).

[28] http://www.youtube.com/CivicEcologyLab

30.7 Challenges and Moving Forward: Research and Policy

Thus far, we have described a number of practices that support learning about and enhancement of biodiversity and ecosystem services in cities. Although empirical research on the learning outcomes of such initiatives is still limited, the program activities are consistent with constructivist, social, and ecological learning theories. In this section we address some of the challenges in the approaches described above, related to the need for assessment and evaluative research, and to broadening impact and policy considerations.

30.7.1 Research

Assessing learning outcomes of the programs described above is extremely challenging due to a number of factors as follows: (1) Each program is unique, hampering quantitative assessment, replication, and cross-practice comparison. Thus, large-scale assessments across multiple settings, such as are conducted to compare science learning in schools, are not generally feasible. (2) Participation is idiosyncratic. For example, in attempts to assess learning outcomes of the Garden Mosaics education program, the evaluator would show up to observe a learning activity only to find that it was canceled due to weather or to an emergency involving a troubled youth and her family. Further, in a widely distributed program such as Garden Mosaics, participation among the community organizations and educators is voluntary and designed to address the needs of each learning setting; thus, fidelity to the program goals and activities varies widely (cf. Penuel and Means 2004). (3) Programs often take place in neighborhoods that lack a history of collaborations with university and other research scientists, and where residents may in fact distrust outside researchers. Such lack of trust calls for participatory and engaged research approaches entailing months and sometimes years of residing in and getting to know the community in order to ensure access to study participants and validity of results.

Despite these challenges, we have several examples of successful studies assessing outcomes of urban environmental education programs such as those described in the case examples above. For example, former Cornell PhD student Alex Kudryavtsev evaluated the impacts of youth programs in the South Bronx, which encompassed civic ecology education along with other activities (e.g., citizen science and recreational boating). Kudryavtsev measured changes in youth participants' sense of place, including place attachment and ecological place meaning, using a quasi-experimental controlled, pre/post- survey research design. Although participants did not experience changes in place attachment as a result of participation in 5-week summer programs, they did experience a change in their ecological place meaning (Kudryavtsev et al. 2012). In other words, through engaging in environmental stewardship and related activities along the Bronx River and in community gardens, roof top gardens, and small city parks, youth who live in one of the highest density, lowest income, and most industrialized neighborhoods in the USA were more likely to attribute positive ecological value to their neighborhood (e.g., more likely to say the Bronx is a place where I can view wildlife or enjoy nature). Kudryavtsev lived in the South Bronx and volunteered at his study programs for a year prior to conducting his research in order to build the trust and partnerships that allowed him to carry out his work. In a separate study, Krasny and Tidball (2009b) present preliminary evidence of science, social, and action learning among youth participants in Garden Mosaics; however, the data were largely self-reporting on the part of youth and their educators rather than an in-depth or controlled study. Unlike Kudryavtsev, the authors did not have the opportunity to spend a long period living in the communities that were the subject of this evaluation.

Currently, Cornell's Civic Ecology Lab is conducting preliminary research to identify instances where civic ecology practices include monitoring of their biodiversity and ecosystem services outcomes. With relatively few exceptions, we have found that practitioners are not monitoring their outcomes, although some express interest in partnering with university researchers to conduct such monitoring (Silva and Krasny 2013; Krasny et al. in review). We intend to use the findings of this study to design university-civic ecology practice partnerships that enable participatory monitoring of ecosystem and learning outcomes. However, a long-term commitment to participatory research will be critical for such monitoring partnerships to bear fruit.

Given the potential for cities to move from acting as sinks to becoming sources for ecosystem services (Bolund and Hunhammar 1999; Colding et al. 2006; Dearborn and Kark 2009; Barthel et al. 2010; Ernstson et al. 2010a; Niemelä et al. 2011; Sassen and Dotan 2011) (see also Chap. 11), and the importance of learning in building urban capacity to provide such services, we recommend efforts to expand monitoring and assessment partnerships among researchers, educators, and civic ecology practitioners. Partnerships that entail significant local practitioner participation in the monitoring and assessment activities will foster new capacities and learning related to assessment methods and their outcomes.

30.7.2 Policy

In this chapter, we began with a discussion of global policy initiatives in education, including the Tbilisi Declaration defining environmental education as changes at the level of individuals, ESD calling for systemic change at the national level that leads to a global transformation in classroom and non-formal education, and CEPA, which integrates a call for change in national education systems with communication and public awareness campaigns targeted at individuals. Later in the chapter, we presented case examples of programs consistent with learning theory and with the participatory approaches invoked in ESD and other global sustainability initiatives. The local case studies come from Africa, Asia, and North America, and are consistent with *Agenda 21*, which proposes addressing sustainable development at multiple levels and a role for local government in implementation of the agenda, including capacity building and involving youth in planning, decision-making, and implementation.

Whereas the importance of local, out-of-school and lifelong learning arenas for public understanding of science has been documented (Falk and Dierking 2002; Bell et al. 2009), the pathway forward for implementing policies to support such learning is less clear than for formal school learning. For example, civic ecology practices and youth participation in urban planning (Chawla 2001; Lane et al. 2005; Læssøe and Krasny 2014) are emerging as an important trend in urban environmental education, and are consistent with the integration of social, cultural, equity, as well as environmental concerns outlined by ESD and CEPA and increasingly by environmental education. However, as predominantly local efforts, these approaches have not generally outlined a means for effecting more strategic change. Such strategic change might occur through resource management and community development agencies, and professional associations and other "shadow networks" (Pelling et al. 2008), in addition to ministries of education.

Partnerships between local educational practices and universities may be critical in setting the stage for more strategic and broader outcomes. For example, universities can play a role in documenting social-ecological system and learning outcomes of various local programs, as well as in building the capacity of local practitioners to collect and analyze outcomes data, to reflect on their results, and to adapt their practice accordingly. Such partnerships will not only strengthen local practices; they will also make them more visible to city and national governments, NGOs, and international organizations (Krasny and Tidball in review).

Universities and local government can also serve as bridging organizations, linking and networking local practices within individual sectors (e.g., community forestry); across sectors; across governance institutions (non-profits, government, business, universities); and across scales (local, regional) (Olsson et al. 2007; Ernstson et al. 2010a). Such bridging occurs through creating face-to-face and web-mediated platforms for discussion, sharing resources, and action on the part of diverse stakeholders. Similar to practitioner-researcher partnerships, bridging across multiple levels can build capacity of individual organizations and practices, while also making their impact more visible to government and NGO policy makers. Through facilitating knowledge transfer and social mobilization, bridging organizations can also foster social innovations (Bodin and Crona 2009; Moore and Westley 2011).

Governments that are committed to supporting local educational innovations such as those described in this chapter will operate less through mandates (e.g., mandating new curricula) and more to support the creation and expansion of community-organized initiatives. This can occur through an environment shaping approach (Weinstein and Tidball 2007; Tidball and Weinstein 2012), which calls for recognizing existing virtuous cycles of greening, civic renewal, and learning, and empowering the agents associated with such cycles to enable them to expand.

Similarly, in addition to creating arenas for dialog and collaboration to address issues at a landscape level, policy makers should seek to understand and actively manage the underlying social structures and processes for ecosystem management (Folke et al. 2007; Olsson et al. 2007). Shaping or managing the environment to support local innovative practices can include such actions as providing financial and technical support and passing enabling legislation (e.g., legislation to grant land tenure and management rights to community or rooftop gardens that serve as learning arenas for biodiversity and ecosystem services). Rather than take the lead, government in many cases will support a civil society organization that has a history of innovation and building trust with the local community to play the lead role in such social-ecological innovation networks (cf. Ernstson et al. 2010b; Ernstson and Elmqvist 2011). In sum, policies to support local initiatives should reflect a series of principles distilled from the literature, including identifying and providing mechanisms to support existing social capital, civic renewal, learning, and place-based stewardship and virtuous cycles of greening; and building the capacity of the local agents through providing secure land tenure, learning opportunities including those encompassing participatory research and monitoring, and economic incentives linked to social rather than personal aims (Ruitenbeek and Cartier 2001; Weinstein and Tidball 2007; Krasny and Tidball in review). For an additional discussion on urban governance for biodiversity and ecosystem services, see Chap. 27.

30.8 Conclusion

We have presented four case examples of learning arenas that support the provision and management of biodiversity and ecosystem services in cities. These case examples are consistent with the social equity, participatory, and environmental principles of global initiatives in education and sustainability (Lotz-Sisitka 2007; Lotz-Sisitka and Raven 2007; Wals 2007; Wals and van der Waal 2014), and with theories that describe the learner as an active participant in shaping his/her learning (Roth 2004; Illeris 2007; Chawla 2008). Given that the examples we have provided are local in scope, partnerships with global initiatives such as ESD and CEPA, and with NGOs, governments, and business are needed to leverage these learning arenas to effect broader regional, national, and even global systemic change.

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Chapter 31
Restoration Ecology in an Urbanizing World

Steven N. Handel, Osamu Saito, and Kazuhiko Takeuchi

Abstract As the world becomes more urbanized, the need for ecosystem services in our population centers has become a priority. The restoration of functioning habitats within cities is being successfully attempted throughout the world. Urban sites available for restoration ecology progress are usually small, surrounded by urban infrastructure, and isolated one from another. This fragmentation constrains the quality of natural communities that are pragmatic ecological targets. Defining restoration goals also must deal with urban abiotic stresses, including the heat island effect, disturbed soils, modified local hydrology, and chemical pollutants in the air, water, and substrate. Existing biodiversity in cities also has atypical taxonomic structure, driven by the loss of many plant and animal species from the original site communities compounded by the addition of non-native plants and animals with high reproductive rates that invade the native remnants. These invasives can further

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reduce site biodiversity. Changing community structure includes an overabundance of herbivores such as deer in North America and introduced insects on all continents. Also, the availability of mutualist species needed for community persistence may be low. Progress in urban ecological restoration requires remedies to mediate the physical and biological changes. Political organization for restoration progress will require cooperation among levels of government and building of new teams of ecologists, engineers, and design professionals to manage restoration planning. However, progress on all continents show that urban ecological restoration has developed successful protocols and can contribute to our cities' environmental and public health.

31.1 Adding Restoration to Urban Environmental Improvements

31.1.1 The Big Apple, Still Ripening

It's been called "Delirious New York," this largest of United States metropolitan areas (Koolhaas 1978). This is a place of great tides of human movement (White 1949). Tides of millions, though the decades, flow in from around the world, creating a great immigrant center. Tides of people, every day, commuting, swishing back and forth from the suburbs, going from bedroom to board room, then return. A slow incremental tide of strivers moves in from other parts of the world, dreaming of rising success in finance, the arts and communication, quickening the frantic throb that makes the city alive. It is the same pattern now found throughout the urban world, now the lifestyle of most humanity.

All the souls that swell these urban tides throughout the world must be supported by an ecological foundation that the local governments must somehow supply. The resources to fulfill the realized multidimensional niche of the human population must be present, the ecology profession would say, but who would respond to *that* Commandment? There are millions of people in this urban world now. Urbanization continues, and five billion people will be urban by 2030 (Seto et al. 2012), crowding streets and pressuring the infrastructure that must support this buzzing lifestyle. The tides may overflow the foundation.

What can the ecological professions offer to this horde? How can the principles of ecology be used effectively to serve these people, if not the millions of other animals that pass through the airspace above and the waters around, that shape a city's borders?

The financial and artistic elites of our major cities receive much attention, but somehow the professions that supply sustainable services must push their way into the spotlight. In New York, America's major metropolitan area, can Broadway—the Great White Way—get a little greener?

Ecological restoration of city parcels can add services to urban centers, but re-establishing ecological links to cities must first address many stresses that have been caused by human activities.

31.1.2 Justifying Investment

Ecological restoration is an investment, not an expense. Ecological restoration was originally seen by many scientists as a way of repairing small scale damage to landscapes tattered by human land uses. Farms get abandoned, and soil and biodiversity must be returned to an approximation of their original structures. Development scars the edges of an intact or severed watershed, and that edge must be sealed again with a sustainable biotic community. Invasive species increase in numbers, changing community organization, and those species must be managed to return towards the original species richness and dominance relationships.

In recent history, 'nature' has been popularly envisioned as an entity distinct from the city, a pristine place located far away from human influence, urban infrastructure and asphalt paved streets (Chap. 2). This attitude is slowly changing. Planners and government agencies now see nature in the city as having value in ecosystem services, not just beauty. These services that are not directly voted upon by city councils, but give real value, replace precious tax dollars in a city budget.

31.1.3 Defining the Ecological Target

There are detailed species lists for many of the world's urban floras (Clemants 2003). These urban areas have complex habitats, not just human-defined industrial and residential zones. In most cities, many landscapes are highly maintained, mowed, used for active sports, and crisscrossed with hardened walkways, but pockets of unmanaged landscape remain unkempt, urban thickets holding on within the crowds. It is here that ecological community patterns develop, albeit of a species mix that is idiosyncratic to local conditions, with elements introduced from many continents (see Chap. 5 for additional discussion on patterns and trends in urban biodiversity).

For example, in the New York metropolitan area, there are over 11,300 ha of public park land in the city alone (Fig. 31.1) (see Chap. 19). Plant identity and distribution here has been studied for over 250 years. Many of these plant surveys include relative numbers and distributions, allowing comparisons of the old and the present (e.g., Robinson et al. 1994; Aronson et al. 2007). In just one borough of New York, 443 native plant species were lost and 481 non-natives gained from 1879 to 1991. Decisions to determine what habitats can be restored are extremely difficult in these dynamic urban areas. The long history of species losses is evidence that conditions are no longer favorable to much of the original biodiversity. The list of species that can maintain their population structures into the future is not yet known.

The appropriate ecological target for restoration rests on many sources of information (Swetnam et al. 1999; Egan and Howell 2001; Gargiullo 2007). The extreme degradation of urban areas may make this sourcebook of information a fictional account of future biodiversity. In this sense, the past is not prologue. Even conserving the species that remain will be difficult with increasing human population pressure and the impact of the modern lifestyle (McKinney 2002).

Fig. 31.1 Natural areas in New York City are scattered throughout the city and include forested, salt marsh, aquatic and marine habitats (in *green*). The hatched areas are federally protected as the Gateway National Recreation Area, and include the Jamaica Bay Wildlife Refuge. Grey areas are paved airports (Modified from Open Accessible Space Information System (OASIS) at www.OASISnyc.net. Published with kind permission of © Open Accessible Space Information System (OASIS) at www.OASISnyc.net 2013. All Rights Reserved) (Color figure online)

Physical, chemical, and biotic stresses are rampant in urban areas. However, successful examples of adding habitats to urban centers can be found throughout the world. Restoration ecology actions can supply ecosystem services to counter these stresses. Restored urban habitats may be different from historic local vegetation because of continuing human-associated activities. However, even modified habitats can add value to human population centers. We summarize here the variety of stresses that urban restoration ecology must confront and give case histories of representative solutions.

31.2 From Urban Stresses to Solutions

31.2.1 Abiotic Stresses That Accompany Urbanization

31.2.1.1 Fragmentation

Long-term urban development and a growing population have erased the original physical landscape that supported our biota. The continued addition of homes, workplaces, roads, power and water facilities, athletic fields, dumps, cemeteries, and the whole witches' brew of infrastructure have together diced the original intact habitats into a fine-grained mix of isolated parcels (Gilbert 1989; Forman 1995). The ability of any species to maintain populations in such a landscape depends on the size, shape, adjacent conditions, and the soil quality of each remaining patch created by this long history of building. For example, the well-documented ecological "edge effect" comes from eliminating moist, shady, extensive habitat pieces around isolated fragments. The modified environment favors only those species that bask in well-lit, hot, and dry urban environments (Cadenasso and Pickett 2001). Many animal species are driven to nest and feed on the edges, increasing attacks by many predators that patrol these habitat boundaries (Askins 2000). Small, dissected habitats are sinks for our native biota as often as they are refuges.

Fragmentation also lowers movement of species from place to place, weakening metapopulation connections. Interest in restoring corridors among fragments in urban areas is high (Bennett 2003), but this is difficult when potential connection routes are dominated by other land uses or important infrastructure needs. Adding parcels of habitat within urban centers can lower landscape fragmentation.

Case history: New Zealand – As an isolated island, New Zealand developed an endemic flora and fauna which is remarkable, but fragile. Introductions of new plants and animals by Polynesian and then European settlers have caused enormous changes in biodiversity and habitat structure (Mooney and Hobbs 2000). However, there are recent urban restoration initiatives which are challenging old problems with new attitudes.

In Hamilton City, on the North Island, a series of deep gullies marbles through the fabric of urban residential areas. Many small patches of native vegetation, averaging 1.1 ha each, remain in the city, from upland forest to emergent herbaceous vegetation in the wet lowlands. Restoration of the gully network is now recognized as a way to add environmental, aesthetic, scenic, and cultural values to the city (Clarkson and McQueen 2004). The project leaders term their work "reconstruction," as so much of the original biota is lost and cannot be completely "restored" (Clarkson et al. 2007). Reintroduction of some native plant species additionally will provide materials for traditional medicine of Maori citizens. Even aquatic life in the lowland streams will benefit from the improved water quality and diminution of siltation that riparian plantings provide. Support for these efforts on public and private land comes from City Council initiatives and pressure from international agreements. Similar urban initiatives are being undertaken in Australian cities where many nonprofit organizations participate in producing native plant materials for urban restoration projects and doing the actual planting (Buchanan 1989).

Of particular interest in the New Zealand example is the emphasis on the landscape ecology advantages of local urban initiatives. In many cities, home gardens are the source for invasive plant species, which can sweep through wildlands. In Hamilton, residents are encouraged to use indigenous plant species in hedges, public parks, and home gardens. These effectively increase population size and range of these species in adjacent natural areas and serve as corridors for the movement of native animals that forage on these plants. The indigenous plantings in home gardens are also sources of native seeds that can be dispersed into the small habitat remnants, replacing the invasive seed shadow from introduced horticultural species, which are so common in many other urban settings (Clarkson and McQueen 2004). For example, the privet (*Ligustrum* spp.) hedges of Australia and New Zealand have led to massive invasive problems in urban moist soils (Daehler 2003).

At the wider scale, there are ecological links between city restorations and the surrounding agricultural and wildland biodiversity (Green and Clarkson 2005). Many native animals and plants are scattered through agricultural lands near cities and these do help support ecosystem services for farming. In New Zealand, there is a drive towards "sympathetic management" of these production lands to support regional needs for increased biodiversity (Green and Clarkson 2005). In these ways, there are sustainable practices which jointly support urban habitats and surrounding commercial activities. There is a serious need for better monitoring to show the precise value of this synoptic approach so that it may be emulated.

One of the advantages of urban restoration efforts is the educational value of small urban preserves so that people can understand the importance of native biodiversity to support regional agricultural practice. Additional work in New Zealand forest plantations has shown how they act as linkages among restored urban preserves, which increases the value of forestry lands for wider societal needs (Norton 1998). Small urban restoration efforts are biotically linked to landscape ecology principles and regional sustainability improvements. Reduction of fragmentation may have economic and well as biodiversity value.

<u>Case history: Hebei, China</u> – Similarly, restoration of habitat has been completed in coastal Hebei Province—a new 60 km^2 park was built along the marine zone of Qinghuangdao City (Padua 2013). Although this city has a large tourist presence,

Fig. 31.2 Restored beachfront habitat at Qinghuangdao City, Hebei Province, China (Photographed by and published with kind permission of © Turenscape 2013. All Rights Reserved)

the coastal zone was degraded by rampant development and habitat fragmentation. The new park is 6.4 km long and restores native coastal vegetation to manage erosion and to supply habitat for coastal zone wildlife. The area had been a national bird reserve, and a dozen of small coastal ponds were installed to serve as resting and feeding grounds (Padua 2013). The concern for improving visitor facilities was met by a series of "floating" boardwalks resting on the new dunes and new grades (Fig. 31.2). In this way, visitors could see the complex ecology of the restored beach on the way to a new Wetland Museum that explains the area's wildlife biodiversity. Creating a new tourist destination in conjunction with lessening ecological fragmentation helps justify the project's cost and advances the social need of public understanding of habitats for the city.

Case history: Beijing, China – Habitat fragmentation can be lessened by replacing industrial land-use with functional habitats. The 2008 Olympic Summer Games received special attention from ecologists and landscape architects. The athletic facilities were surrounded by a new lake, wetlands, meadows, and diverse woodlands (Fig. 31.3). The master plan from Sasaki Associates of Massachusetts addressed the extensive urban planning needs of an Olympic event: many new facilities that had to be constructed, new habitat parcels that could be enjoyed after the games ended, and an economic vision to transform the entire site into a public and convention center. A former light industry area was cleared and transformed into a Forest Park,

Fig. 31.3 (**a**) Constructed wetlands and boardwalks offer educational, aesthetic, and wildlife values for Beijing. This element of the Olympic Forest Park lies to the north of the National Stadium of the 2008 Olympic Games (Rendered by and published with kind permission of © Sasaki Associates 2013. All Rights Reserved) (**b**) A view over the Beijing Olympic Forest Park's southward along the Beijing Imperial Central Axis, showing the man-made complex of woodlands, meadows, lake, islands and wetlands, and the landscape connection of the Park to the rest of the Olympic Green and venues (Photographed and published with kind permission of © Beijing Tsinghua Urban Planning & Design Institute 2013. All Rights Reserved)

near the Fifth Ring Road of Beijing (Dong et al. 2006). The many native plant species introduced into the new Park can spread into the surrounding neighborhoods to restore the native diversity, which previously had been long gone. In this way fragments can be continually linked.

The Park was placed on the central geographic axis of Beijing, which lines up with the Emperor's throne in the ancient palace in downtown Beijing. This placement emphasizes the importance of this Park to the cultural life of Beijing and brings it special attention for Chinese citizens. Natural history combined with cultural affection highlights the new ecosystem services which have been added in conjunction to the Olympic Games. Together with the coastal ecological parks, Beijing's Forest Park is an expression of the value of ecological structure to the burgeoning new Chinese urban landscape. The joining of ecological principles with landscape design is a model of interdisciplinarity.

Case history: Budapest, Hungary – Similarly, in Budapest, Hungary, the concern was to add a network of green spaces throughout the two sides of the ancient city (URGE-Team 2004). The topography is diverse, and commercial development had eliminated a variety of habitat types (Beynon 1943). A green belt system both protects remaining green spaces by acting as a buffer against development stresses, and allows for additional connected and restored green hectares. Some existing areas will be enlarged so that a variation in scale of green spaces is achieved. Some park areas, such as Szent Istvan Park, will emphasize public recreation, while others, such as Orczy Garden, will have a passive horticultural goal.

In an old, densely populated city, the restoration goals may not allow a full complement of woodland or meadows species to survive. However, incremental increases of some native populations of plants and animals and their subsequent ecological service potential may have increased political support for ecological structure when combined with cultural advantages.

31.2.1.2 Suppression of Disturbances

Many species require early successional ecological conditions. These are driven by disturbances on the landscape, originally of natural causes. Fire, the pounding of a migrating herd, and seasonal flooding played their role in creating temporarily open land. This is quickly occupied by some fast invading species before a subsequent competitive battle replaces these pioneers with other species.

People create disturbances with almost everything they do, but it's a different kind from natural events, and rarely supports the life history needs of native early successional species. Before we can restore meadows, shrublands, and herbaceous marshlands, we must find a way to reestablish disturbance regimes. Not only is this practically difficult, it is often legally prohibited. For example, in many cities, purposely lit fires are prohibited so that air pollution does not increase. Those meadow species which require a landscape recently burned now have no place to become established. Similarly, those wetland habitat herbs which require seasonal flooding for recruitment of new seedlings are the city's herbaceous homeless; stream and riverine edges are more often concrete and stone riprap than exposed alluvial soil. This infrastructure is to curtail erosion but bioengineered options are possible. On streambanks, use of fascines and life stakes are more appropriate options (Schiechtl and Stern 1996). Plowing or disking of land to promote new microsites can be an acceptable urban alternative to fire. However, targets of restoration must be adjusted to allowing disturbance techniques within a city's guidelines.

This is not to say cities are immune from large scale and natural disturbances that create their own new needs and trajectories. The study by Saito and Takeuchi on the 2011 East Japan earthquake and its massive effects (Box 31.1) offers a sober reminder that cities, for all their technology and modernity, are subservient to geologic forces. The Geopark being proposed for this region in Japan is restoration at a level few urban ecologists have considered in the past. However the lessons there are of humility for dwellers of coastal megacities and their environmental managers.

Box 31.1 Ecological Restoration After Natural Disasters: The Great East Japan Earthquake

Osamu Saito and Kazuhiko Takeuchi

Damage to Biodiversity and Ecosystem Services

The Great East Japan Earthquake and resulting tsunami of 11 March, 2011 have had significant direct and indirect impacts on ecosystem services in the affected area. In particular, most of the fishing villages along the Pacific coast endured catastrophic damage. Agricultural land such as paddy fields was also flooded by the tsunami, and livestock and agricultural industries suffered radioactive contamination from the Fukushima Daiichi nuclear plant disaster.

The inundated municipalities lost about 25 % of farmland for rice production and 5 % for other crops. There have been significant decreases in livestock due to the disaster, and to the decision to euthanize all livestock in the exclusion zone surrounding the Fukushima Daiichi nuclear power plant. Problems related to the safety of animal products are extremely wide-reaching and felt nationwide. It will take several decades for fisheries in the region to recover, with 21,589 ships, many fish farms, and fish processing facilities damaged by the tsunami (Box Fig. 31.1). Sea fisheries and fish farms in the study areas previously comprised 14 and 19 % of the national catch respectively.

Of the 83 km^2 below the previous highest tidal level prior to the earthquake, 3 km^2 were below sea level. However, due to ground sinking, the area below the highest tidal level and the area below sea level have changed to 111 km^2 (a factor of 1.34) and 16 km^2 (a factor of 5.3), respectively (Box Fig. 31.2a).

The destruction of one third of the 33 km^2 of coastal pine forests in the region affected the tide prevention and sand erosion control services they provide (Box Fig. 31.2b).

(continued)

Box 31.1 (continued)

Box Fig. 31.1 (**a**) Marine vessel carried to the roof of the local inn by the tsunami in Otsuchi, Iwate prefecture (24 April, 2011). (**b**) Large fishing vessel remained intact at the middle of the affected area of Kesennuma, Miyagi prefecture (20 August, 2012) (Photographed by and published with kind permission of © Osamu Saito and Kazuhiko Takeuchi 2013. All Rights Reserved)

(continued)

Box 31.1 (continued)

Box Fig. 31.2 (**a**) The former paddy field was completely inundated due to tsunami and ground sinking in Rikuzen Takata, Iwate prefecture (20 August, 2012) (**b**) Destroyed coastal pine forests in Rikuzentakata, Iwate prefecture. One survivor has been called "the miracle pine" that symbolizes resilience of the affected area (23 April, 2011) (Photographed by and published with kind permission of © Osamu Saito and Kazuhiko Takeuchi 2013. All Rights Reserved)

(continued)

Box 31.1 (continued)

Various cultural services have been affected, with damage to 17 % of the 1,357 cultural properties located in Aomori, Iwate, Miyagi, and Fukushima prefectures. The number of travellers visiting the region for multi-day trips over the annual *Obon* holiday (in mid-August) declined by 42 % compared to the previous year.

Rebuilding and Ecological Restoration

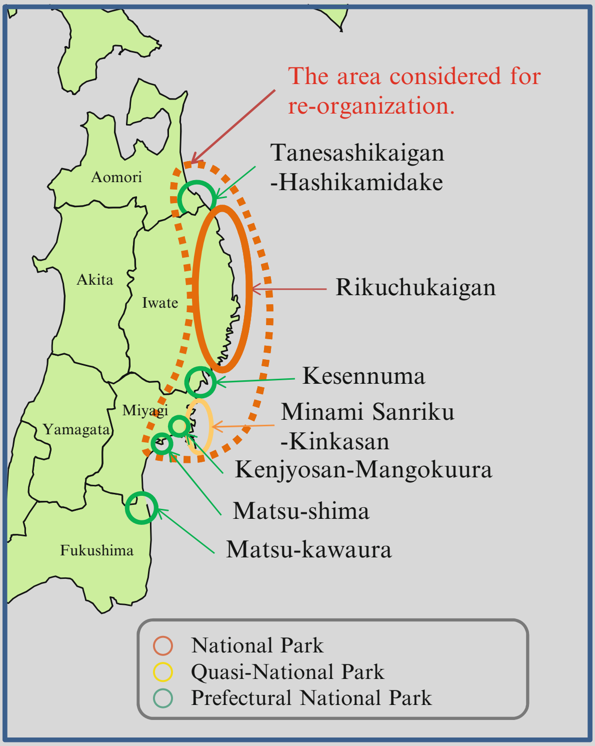
Much of the disaster-stricken area contains a range of typical Japanese *satoyama* (terrestrial) and *satoumi* (coastal) regions, especially the Sanriku coastal areas (the Pacific coast of northeast Japan). These traditional landscapes represent a balanced relationship between human beings and nature. However, due to rapid urbanization, shifting resource needs, and industrialization, both satoyama and satoumi have declined in the last 50 years, thus affecting the coupled ecosystem services (for further discussion on satoyama and satoumi landscapes, see the local assessment in Chap. 8).

With fishers utilizing satoumi, and also cutting and using the wood from satoyama nearby, most local residents were involved in both farming and fishing. In these areas, satoyama and satoumi are connected by small rivers, and the linkage of forest–river–sea provides the community with the bounty of nature through material flow (e.g., nutrition). Given the recent abundance of forests and expansion of unmanaged forests (coniferous plantations) in upstream areas, the nature of satoumi in downstream areas has been distorted, and the linkages between agriculture, forestry, and fishery have disappeared.

In order to improve the resilience of these societies, the link between satoyama and satoumi needs to be strengthened. Planning amenity and recreational spaces for local residents and visitors would also provide safe places of refuge during natural disasters. The Sanriku coast boasted many national and prefectural parks whose facilities were destroyed during the disaster. Facing restoration of the destroyed parks, the Ministry of the Environment, Japan has been proposing a new type of national park which will further the aims of disaster prevention and mitigation, as well as the revival of the fishery industry (Box Fig. 31.3). The initiative aims to contribute to the recovery of these areas by reviving satoyama and satoumi and recreating linkages between forests, rivers, and the ocean. This new park is also expected to function as a "Sanriku Geopark," preserving the memory of the earthquake and tsunami, and providing education on the geology and geography of a natural rias coastline. In addition, the development of various renewable energy sources

(continued)

Box 31.1 (continued)



Box Fig. 31.3 "Sanriku Reconstruction National Park Initiative" and existing national parks (Rendered by and published with kind permission of © Japanese Ministry of Environment 2013. All Rights Reserved)

including solar, wind and geothermal energy in satoyama and satoumi have been seriously discussed, in order to balance the needs of environmental protection and energy production.

Reviving Industries and Regional Communities

Even before the earthquake, the primary industry in the region was threatened due to an aging population and a decline in agricultural workers. Thus the revival of agricultural, forestry, and fishery industries cannot be carried out without measures to address the lack of human resources. There is a need to encourage private companies and other newcomers; to integrate production, process, and circulation; to revive recreation and tourism; and to promote high

(continued)

Box 31.1 (continued)

value-added industries. Fundamental restructuring of land use in the affected areas should be considered with a vision for future industries and society.

In the cities and villages affected by the March 11 disaster, reconstruction began with the transfer of housing to higher ground. This presents a good opportunity for compacting and re-zoning these areas, considering the changed needs of a shrinking and ageing population. In spatial planning for compacting and re-zoning the cities or villages, land which is highly vulnerable to natural disasters should be restored as agricultural land, woodland and wetland. In this context, the importance of the connections between various stakeholders in the local area should be emphasized. Workers engaged in the agriculture, forestry, and fishery industries expected to continue to be the main laborers in future. But all citizens need to be involved on an equal basis in the management of common resources, including those who are working in the government, private companies, non-governmental organizations, as well as urban residents.

The Diversity of the Region

The area affected by the 11 March disaster was much more extensive than had been expected, and each local community experienced varying degrees of damage. It is therefore impossible to construct a universal model for the revival of all the areas. Rather, a detailed examination of the link between the degree of damage and natural and social factors must be conducted, and then a plan developed in accordance with each specific natural environment and social capital. During the past several decades in Japan, public participation in city planning has been effectively introduced, bringing significant benefits due to the experience and enthusiasm of locals to engage in the development of their local community. A bottom-up approach is therefore needed in the construction of a vision for rebuilding the region, rather than a traditional top-down model. However, during construction after World War II, each local region was developed as a more or less similar or uniform landscape everywhere, without unique characteristics. In this sense, local diversification is a key for each region within the process of post-disaster rebuilding.

31.2.1.3 Physical Change

The concentration of hardscape in the city and the exhaust from thousands of cars and chimneys has created a heat island, raising the ambient temperature several degrees above adjacent rural areas (Alberti 2008) (Chaps. 1 and 25). The hotter air consequently heats the soil and waterways, changing relative abundances of organisms that are sensitive to these temperature regimes.

Building and rebuilding over the centuries also has modified the original soil structure (Bradshaw and Chadwick 1980). Fill brought in to elevate new structures is often mineral soil or construction debris and is inappropriate for healthy soil biota (Bullock and Gregory 1991). Piping of rain water and constriction of normal groundwater flows has significantly changed the rate and frequency of water movements in the remaining urban surficial stream corridors (Ehrenfeld 2000). This results in flashy and fast-moving flows after precipitation events and undercutting of stream banks. These actions destroy aquatic microhabitats needed for invertebrate and vertebrate reproduction.

In these ways, the physical world in the built city is completely different from the atmosphere, substrate, and hydrosphere that were the pre-urbanized landscape. A seed or juvenile of any native species that enters an urban world confronts novel conditions. Physiology, behavior, and life history events are challenged by the new physical environment. Again, targets of urban restoration ecology must be compatible with new conditions, not historic environments.

31.2.1.4 Combining Habitat and Infrastructure Needs

A case study of hydrology change and urban restoration can be found at Sweetbrook Park, New York City, where stream restoration rescued a flooded street in Staten Island. In 1964, the huge Verrazano-Narrows Bridge was opened and the human population of Staten Island tripled within 20 years. This was "delirious New York" at its fastest. The rapid urbanization overwhelmed the old infrastructure of water supply and many streets suffered from urban flash flooding. The usefulness of the remaining stream corridors as an element in stormwater management was considered.

Sweetbrook Park was the first element of the City's Staten Island Bluebelt initiative (Eisenman 2005). The 49 km^2 Bluebelt drainage area is ribboned by parcels of historic stream corridors that somehow escaped being placed in culverts. These streams are natural elements that can reduce flooding, improve water quality, and add an overlay of green space. The Bluebelt project restores wetlands, stream restoration, stilling basins, and connections to estuaries. 92 stormwater wetlands are included and the first one at Sweetbrook Road, completed in 1995, exemplifies the ecology-human interactions that can occur (Thompson 1998).

Sweetbrook is a narrow remnant of ancient New York habitat. Most of this stream was in pipes underground, but a 427 m section had escaped domestication. In 1994, the banks included tall, 1 m diameter oak trees, tulip trees, and maples. Scattered on the stream banks were small populations of ostrich ferns (*Matteuccia struthiopteris*), Jack-in-the-pulpits (*Arisaema triphyllum*), and other native wildflowers that had been exterminated over much of the island. Alas, most of the site was the common detritus of human civilization: steel shopping carts; torn upholstered sofas; twisted unloved bicycles; construction debris of wallboard and concrete stacked like a failed modern sculpture; and enough broken bottles to christen a grand fleet of ships. These were removed and work for a new ecological community commenced.

During storms, high volumes of water entering the stream segment were shunted into a new 3.6×3.6 m tunnel that leads to a treatment plant. A splitter in the pipe allows base flow to enter the historic streambed, nurturing the plant and animal communities there. A ponding area was dug to create habitat for turtles and fish. Stone riffles were added to the stream bed for invertebrate habitat and to dissipate storm flow velocities. The city did this ecological restoration because it meant that the new pipe installation could be smaller and less expensive. Less water would enter the treatment facilities downstream. Boulders and logs from downed invasive trees were used to stabilize the banks; these also enhanced wildlife habitat. Fencing along the street curb stopped additional dumping of debris. By making this parcel useful and clean for local residents, it was now valued by the community, not ignored or continually degraded. The home owners near the stream now guard over its health rather than feel gloomy about its dereliction.

This first section of the Bluebelt showed how stormwater management needs, restored ecological health, and public appreciation could all come together in a cost effective manner. Many other sections of the Bluebelt have now been built and it is considered a model of best management practices for urban stormwater (Eisenman 2005).

Similar initiatives are possible in many cities. In Britain, there is a tradition of urban conservation initiatives throughout the country (Goode 1989; Fitter 1945). In suburban Birmingham, former farmland and grazing meadows have been modified to a 67 ha country park consisting of wet meadows and poorly drained woodlands. This is located adjacent to some residential areas and serves for stormwater drainage as well as recreational space. There was particular concern for restoration of habitat for the white-clawed crayfish (*Austropotamobius pallipes*), Britain's only native crayfish species, which had been found in adjacent headwaters. In this way, both ecosystem services for the city as well as critical habitat for an endangered species could be served. Part of the new park also is used for active recreation.

31.2.1.5 Chemical Change

Human action has added noxious compounds to the air and soil which are stressful to plant and animal species (Gilbert 1989). Very many compounds are hydrocarbons which can be degraded over time, but heavy metals and other toxins linger in the soil and atmosphere. This presents long-term barriers to restoration of species populations. Although sites of extreme pollution are sometimes improved by bioremediation techniques (Cummings 2010), the chemical pollutants often are not removed, but are sequestered on-site (EPA 2009). This eliminates many sections of the urban landscape from long-term ecological health. The wide-scale landscape starts to resemble a vast sheet of Swiss cheese where the holes are areas of past land-use which are biotically depauperate, and have been removed from a fully functioning biodiverse future. They are not sources for restored ecosystem services; they are sinks for urban environmental hope.

A series of urban restoration projects and parks have been done in China in order to remediate such pollutant loads. The rapid and continuing industrialization and urbanization of China and India has been accompanied by concerns about wide-scale environmental degradation (Liu and Diamond 2005). However, these countries have a long tradition of sophisticated landscape architecture in their urban centers.

A center for this ecological design vocabulary is in Beijing, yielding a series of new urban parks (Saunders 2013). The credo of the landscape architecture team is, "design with place, design with prudence, design with nature, and make nature visible" (Yu et al. 2001).

Expressions of these ideas on the land are now seen throughout the country[1]. The new riverside Houtan Park in Shanghai, for example, was built along the Huangpu River in conjunction with the Shanghai World Expo in 2010 (Goldhagan 2013). This 14 ha park stretches 1.7 km along the river in a thin band 30–80 m wide. The park's goals are to create a beautiful public experience, but also to biologically treat the contaminated water of the river and celebrate the agricultural and industrial past of this part of Shanghai. The treatment wetlands quickly upgraded the water quality which then was used for non-potable functions. Terraces containing rice and other field crops border public walkways which were built along the river. These allow an experience of urban farming as well as industrial heritage. The paths are elevated (Fig. 31.4) and a porous rip-rap wall was built between the river and park to allow for changes in water elevation. This will advance flood control over the coming years. The "living system" of the new park is meant as a demonstration of an "ecological culture" which can inform future urban initiatives (Gordon 2010).

Restoration of urban parcels also lessens the biotic effects of chemical pollution by supplying refuges for organisms when new pollution events occur. This landscape ecology effect gives habitat destinations for organisms that are threatened at damaged locations. For example, if a heron rookery is destroyed by an oil spill or fire elsewhere in a harbor (Burger 1997), the birds can seek shelter at the newly restored habitat parcel nearby. In this sense, the value of ecological restoration is for the wider biotic community as well as at the project site itself. This has been discussed for rural areas (Fischer and Lindenmayer 2008), but urban zones usually are the most fragmented landscapes, and these "stepping stones" of restored habitat have great value to improve population persistence. The boundaries of the restored habitat may be drawn on a map, but the ecosystem services extend well beyond the real estate line.

31.2.1.6 Hot Times

Increased greenhouse gases, many of them concentrated in our urban centers, have been causing long-term heating of the atmosphere (Chap. 25). Also, hotter urban areas, due to the heat island effect of cities, cause immediate biotic changes. Many species are unable to function properly with an increase in ambient temperature. For other species, their geographic range will change as winters are mild and/or summers are hotter. This change in species ranges has already been seen for many biotas (Walther et al. 2002; Araújo and Luoto 2007). However, not all species migrate at the same rate. There is concern for disconnections between mutualists (e.g., pollinators and their host flowers, soil fungi and host roots). Equally important are dangerous

[1] http://www.turenscape.com/english/

Fig. 31.4 (**a**) Shangai Houtan Park rings the city's river and adds habitat and urban agricultural lands to a former polluted and industrial zone. (**b**) Access to the horticultural and habitat plantings and the river front are now possible through a new network of public walkways (Rendering and photograph by and published with kind permission of © Turenscape 2013. All Rights Reserved)

new connections between pathogens that spread to new, now warm, areas and hosts that have no evolutionary history with the new microbes. New damaging infections may become frequent.

Designs for ecological restorations are dealing with moving targets of climatic and landscape conditions. However, restoring habitats in cities can lower the heat island effect through transpiration and shading, and provide new buffers against expected ocean level rise. Rising seawater around major cities can be reflected in coastal ecological park designs.

For example, in New York City, the 300-year-old commercial waterfront along the East River, a tidal strait, is being transformed into a 32 ha Brooklyn Bridge Park, following the design of Michael Van Valkenburgh Associates (Berrizbeitia 2009). This will combine both public recreation grounds and native coastal habitats[2].

Although this strip of land is narrow, removal of the hardscape and old fill, then adding walkways will increase the functional length of the river edge from 3.9 to 6.4 km, and can allow for a wide variety of restored communities (Fig. 31.5). These will include salt marsh, coastal meadow, native woodland, freshwater wetlands, and salt-tolerant shrublands and dune habitats. Together these habitats support perching and wading birds, fish nurseries, marine invertebrates, and terrestrial insects (Urbanski and Gleeson 2012). When sea levels rise, this complex of coastal habitats can shift in response to the tidal regime. This design is more effective as an ecological amenity than attempting to continually maintain the old commercial infrastructure.

This can increase civic interest and action to protect the urban marine environment.

The new walkways over the East River can also act as wave attenuators. The 2012 Hurricane Sandy in the United States has been a prod for new solutions to ocean rise. Restored coastal marshes and dunes can reduce wave surges and add a very visible regulatory service for urban residents as a result of the ecological design. Reconfiguration of streets that end near the marine zone is also being suggested; these "marine streets" (Fig. 31.6) gently decline into the saltwater and can support a variety of intertidal and adjacent upland habitats as ocean level changes (Wilks 2011). In these ways, computation of the value of coastal ecological restoration must include advantages during storm surges and opportunity value for organisms that may only episodically visit a site (Bennett 2003).

31.2.2 Biotic Changes in the City

31.2.2.1 Loss of Historic Species

> When I consider that the nobler animals have been exterminated here, – the cougar, panther, lynx, wolverine, wolf, bear, moose,… etc., – I cannot but feel as if I lived in a tamed… country. Is it not a maimed and imperfect nature that I am conversant with?
>
> (Henry David Thoreau, 1855, quoted in Cronon 1983, p. 4)

When written, over 150 years ago, Thoreau's musing about suburban Boston, USA, described a food web that had already lost many major vertebrate species. Boston's population has quadrupled since then, and its suburbs have evolved from agricultural to urbanized (Binford 1985). In many metropolitan areas, similar

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[2] http://www.brooklynbridgepark.org/

Fig. 31.5 (**a**) Brooklyn Bridge Park forms a thin band of new habitat along the eastern edge of the East River of New York City. This land was formerly a commercial port zone for over 300 years (Rendered by and published with kind permission of © Brooklyn Bridge Park Conservancy 2013. All Rights Reserved) (**b**) New ecological habitats and structure at the Brooklyn Bridge Park, NY (Photographed by and published with kind permission of © Steven N. Handel 2013. All Rights Reserved)

Fig. 31.5 (continued)

impoverishments have occurred. A reconstruction of the original living landscape in New York, for example, records a huge diversity of habitat types and an assumed wide biodiversity (Sanderson 2009). Given the physical and chemical changes summarized above, the landscape is no longer favorable to many native species. The addition of hundreds of alien species from other continents has created new biotic communities in the remaining unpaved urban substrates. These new communities, rich mixtures of alien and native, are nicknamed "synthetic vegetation," with species joined together by human activities (Bridgewater 1990). These new community types vary from place to place, and the normal temperate ecosystem mosaic of meadow, shrub land, and forest has been replaced by peculiar urban parcels. Individuals leaving one patch for another may often encounter very different vegetation (Gilbert 1989). The aggressive nature of new invaders leads to new vegetation trajectories.

The animal community structure also changes with urbanization. In eastern North America there is an overpopulation of white-tailed deer (*Odocoileus virginianus*), and they are destroying many urban natural habitats (Drake et al. 2002). In addition to causing car collisions, devouring residential plantings, and spreading tick diseases, the vast, peripatetic herd can completely degrade our natural lands and

Fig. 31.6 "Marine Streets" reconfigure the ends of urban streets that are adjacent to saltwater into slowing, sloping infrastructure elements. These include coastal habitats of salt tolerant plants. The street ends are public park amenities that are resilient to the expected increasing changes in ocean levels near our cities (Reproduced from Wilks 2011. Published with kind permission of © Ecological Restoration 2011. All Rights Reserved)

designed public parks. The thousands of deer eat all but the most noxious plants. This behavior favors the continued spread of alien species that have spines and poisons (Waller and Alverson 1997). This is the "perfect storm" of habitat damage (Baiser et al. 2008). Consequently, many of our public parks have canopy trees, but denuded understories. Native shrubs and wildflowers have been eliminated, and tree regeneration is almost impossible with this herbivory load. Many urban residents have never seen a healthy woodland park with multiple vegetation layers. The most common sight is a clearing with scattered trees.

Although many European cities are neither small nor new, and are surrounded by heavily managed lands for industry or agriculture, urban habitats there still are widely used by many animal taxa, but the present urban wild communities are different from the historical suite of species (e.g., Botkin and Beveridge 1997; Fernandez-Juricic and Jokimäki 2001; French et al. 2005; Brenneisen 2006). The overall challenge for restoration is to mesh elements of the past that remain with the new species, which seem here to stay.

31.2.2.2 Managing the Overabundance of Herbivores

Hunting is not possible in most urban areas, so mechanical barriers to destructive herbivores are most useful. For example, Eagle Rock Reservation sits on a high, sloping ridge in the middle of middle-class suburban communities in Essex County, New Jersey, USA. Designed by Frederick Law Olmsted in 1907, the 165 ha park is substantially oak woodland with some headwater streams. However, this historical park has been severely degraded by a plague of deer and by the slow accretion of invasive plant species that are common in the metropolitan area. Visitors here enjoy the services of the canopy trees but the ground layers are missing or atypical, having only some regional invasive herbs. Most visitors here assume this is a "natural condition;" the deer damage is of longer duration than the age of most park users.

The ecological remedy here had two prongs; one is habitat intervention, and the other is social science. The landscape team needed to reestablish the presence of native plant populations but also gently show the public that their beloved park was an ecological skeleton that was continuously weakening. Deer hunting had been established by the county government despite objections from animal lovers who were not convinced about the damage the deer were doing to other wildlife species. The restoration ecology remedy was to sequester some areas from the deer by tall fencing, plant many new native species within the fenced area, and reveal to the public how a healthy woodland parcel should be structured. By comparison, the areas outside the fenced parcels would be seen for the first time as relatively empty of vegetation complexity and lacking many ecological functions.

It is expected that seeds from the thousands of trees, shrubs, and wildflowers planted in 2010 within the exclosures will be dispersed out to cause local spread of new populations (Hoppes 1988; Handel 1997). Managers are confident that this will occur, as fruit-eating birds are still common in the trees here. The plants have grown quickly, and freed from deer herbivory are producing many fruits. They are being spread at no cost by the bird community and the wind, and seedlings are emerging. However, any new recruits to the ground layer vegetation will still be devoured by the deer population. Population success for plants includes growth and maturation phases (Harper 1977); demographically, the exclosures are not yet succeeding. The park managers hope that a near-term value will be a public understanding of the weakened status of vegetative dynamics outside the fencing. In time the whole park can be fenced, access for human visitors established, and the deer devastation eliminated. Without these proactive steps, this Olmsted park will be biotically degraded for many more decades. In this restoration project, the practical target was the critical stage of showing to the public their park's problem and beginning the slow rise in plant birth rates to secure healthy plant populations.

31.2.2.3 Recovery from an Invaded Landscape

Another example of urban restoration, but without deer as a driver of the degradation, was done in a historic public park, where managers are using a suite of native plant species as community resistance to the invaders. Designed in 1868, Prospect Park in Brooklyn, New York, transformed a farming area into a complex 213 ha design of woodlands and large meadows with a sylvan waterway and lakes fed by a reservoir system (Colley and Colley 2013). This park is the major playground for two and half million city residents. Years of visitors tramping freely through the woodlands caused destruction of vegetation, compaction of soil, and vandalism. A 25-year-long ecological restoration plan was designed in 1984 to restore sustainable vegetation. This time line to complete the project is remarkable for the length of institutional commitment, "mostly lacking in local government undertakings" (Toth 1995).

The 101 ha woodland remnant was fenced during the restoration to eliminate human traffic. Invasive plants were removed, and then a large diversity of herbaceous and woody native plant species installed. This effort was funded both by public funds and an extensive fundraising effort that targeted corporate donations and the many users of the Park. The effort is managed by a private organization, the Prospect Park Alliance[3]. This public-private partnership for urban parks is a model appropriate for developing cities as well as established ones. The stakeholders for the park's restoration determine phasing and raise funds, but also play a proactive role in the physical restoration of the grounds and historic buildings. Managing the restoration of Prospect Park will take decades, and the park administration tries to secure at least 5,000 volunteer hours of work each year. Some of these helpers are students, but many others are young professionals who work in the park as a social activity, not as a beginning of an environmental career. Use of volunteers physically improves the land and vegetation, but also builds a new social network of people who become knowledgeable and personally concerned with maintaining the ecological health of Prospect Park. These social attitudes are seen as a necessary partner to the financial resources needed (for additional discussion on urban landscapes as learning arenas and sources of civil society stewardship for biodiversity and ecosystem services, see Chap. 30).

For many decades, the park was managed with benign neglect, under the assumption that the remnant woods would persist, as in a "natural" stand. In reality, most of the

[3] http://www.prospectpark.org/about/alliance

trees were installed by the designers, Olmsted and Vaux, in the nineteenth century (Bluestone 1987). However, small urban remnants suffer from many physical stresses and require active management.

An additional destructive management intervention occurred during the 1960s and 1970s, when many elements of the understory vegetation were purposely removed. This vegetation was perceived as a threat to public safety, as muggers and thieves could hide within the understory (Toth 1991). Vegetation was seen not as an amenity and fountain for ecosystem services, but as a mask for danger. This public attitude had to be overcome by education through guided tours and signage. Then a dense and ecologically functioning new understory could be socially accepted by the neighborhood.

31.2.2.4 Disconnected Mutualists

The disconnect between a plant and its mutualists (pollinators, seed dispersers, mycorrhizae, etc.) may be a significant constraint to restoring sustainable habitats in cities (Handel 1997). All habitats require reproduction and recruitment of new individuals; without mutualists such as pollinators, this may be impossible. Also, habitat patches are required for breeding and feeding territories, or else reproduction for many bird species may be impossible (Askins 2000). Without these dispersers, movement of diaspores among patches will be rare, and resilience to local stressors weakened. The overall pattern in cities may be dysfunctional plant populations within new community types whose species have no common evolutionary history. But solutions to restoring these interactions in cities are possible.

Restoration of invertebrate communities is possible, with attention to microhabitat structure for nesting and overwintering (Kirby 2001), and selection of habitat elements that address the host-specific feeding and ovipositioning requirements of many pollinators (Menz et al. 2011). Design of soil conditions and plant communities that addresses mutualist niche requirements must be part of a restoration design. For example, open ground of different textures of soil will be favorable microhabitats for many invertebrates (Kirby 2001). This can be done in a cost-effective manner if ecological perspectives are added to the design teams.

Locations for doing such comprehensive ecological restoration in our older cities may be difficult to find. However, large urban landfills can be re-purposed into arenas for this work. Close to population centers, inappropriate for many construction uses because they are unstable, and often owned by government agencies, these urban landfills are becoming new targets for restoration progress (Harnik et al. 2006). Old regulations often prohibited the planting of trees on landfills, because of fear that roots would penetrate the cap and subsequently cause pollution of groundwater. In most cases, these worries are unfounded (Robinson and Handel 1995). For example, near London, the large Pitsea Landfill has been covered by a restored and extensive oak forest (Dobson and Moffat 1995). Many public agencies are now encouraging the reuse of landfills into green space and its many advantages (e.g., NJ Meadowlands Commission 2006; EPA 2009).

The rebranding of urban landfills into land which has new public value requires a series of ecological interventions. These include attention to the quality of the final cover of soil, the determination of plant community types that can thrive on thin soils, the ecological constraints of relatively small habitat size surrounded by industrial and residential districts, and the administrative problems of transferring an engineering feature into a natural resource venue (Robinson et al. 2002). However, each completed example builds the momentum for changing these odiferous sites into habitats with diverse advantages.

Initiatives are occurring throughout the world. In 1988, in Mumbai, India, the Metropolitan Region Development Authority created a nature park covering 14 ha of a former city landfill (Monga 2005). Within a few years, 53 species of butterflies and 44 species of reptiles and amphibians were found using the new habitat (Raut and Pendharkar 2010). In Israel, the huge Hiriya landfill adjacent to Tel Aviv is being turning into a nature destination, Ariel Sharon Park (Alon-Mozes 2012). In China, several landfills are now being refitted into habitat parcels (Wong and Bradshaw 2002). This particular opportunity for urban restoration has itself taken root.

31.2.2.5 Genetic Constraints

Not all genotypes of a species can thrive in the extremely modified conditions of cities. Ecotypic variation of plants in response to very local conditions has been demonstrated for many species (Briggs 2009). Although landscape designers will routinely choose plant species based on soil moisture conditions and available insolation, the peculiar and novel conditions in modern cities may require a small subset of genotypes within a species that are able to succeed (Handel, et al. 1994). Sometimes urban conditions change significantly over a few meters. For example, populations of plantain (*Plantago lanceolata*) near roads can be lead tolerant, an evolutionary response to gasoline additives. Populations 4 m away from the road lacked this level of heavy metal tolerance (Wu and Antonovics 1976). Similarly, populations of dandelions (*Taraxacum officinale*) have evolved prostrate leaves in response to mowing and other human disturbance; adjacent populations in unmowed ditches have vertical leaves (Solbrig and Simpson 1977). Dandelions are not included in planting lists despite their charm, but the evolutionary ecology evolution principle has been established. Small scale habitat differences must be remediated by subtle genotypic selections.

The availability of urban-adapted plants is not large. Some plants are known to be horticulturally tolerant of the polluted air and soil in urban conditions, but a wider urban biodiversity is needed to secure sustainable ecosystem services (Hufford and Mazer 2003). With climate change, more heat-stress tolerant genotypes may be needed in the planting palettes (see Chap. 25 for further information on climate change and urban vulnerability). This must be developed by the next generation of urban restorationists and their nursery manager partners.

31.2.3 Social Organizations Are Not Congruent with Ecological Needs

31.2.3.1 Human Ecology Constraints

In addition to the physical and biotic changes in urban settings, there are many social and political decisions which constrain our ability to restore habitats. Political boundaries within metropolitan areas are complex, with decision making shared among many levels of government (see Chap. 27 for additional coverage of urban governance of biodiversity and ecosystem services). For instance, in the New York City area, three states (New York, New Jersey, and Connecticut) have political boundaries drawn centuries ago, without any regard for ecological gradients. Flows of species, energy, and nutrients occur according to forces completely separate from political boundaries. At best, any ecological initiative must be approved by several regulatory bodies and win acceptance from a diverse group of people. At worst, the political entities involved have procedures and rules which make effective ecological improvement impossible in the short-term. For example, in the State of New Jersey Meadowlands, fourteen different towns controlled parts of this 7,900 h watershed (NJ Meadowlands Commission 2006). Each made their own decisions about zoning, local roads, and environmentally sensitive areas. The state eventually set up an agency in 1969, the Meadowlands Commission[4], with zoning authority that superseded the power of the individual towns. Then a holistic approach to land-use planning, water flows and habitat restoration was able to proceed.

In addition to the horizontal problems of political boundaries, there is a slippery temporal axis of institutional memory. In many environmental agencies, case studies and decisions are handled by a young professional staff, typically with inadequate time to fully explore all ecological needs. The best ones are often promoted and move on to other responsibilities. Consequently the institutional memory about any one land parcel can be short. Records of research, background information, and actions on the ground are easily forgotten (even with computerized records, the rapid evolution of software and drive media can cause old, even 10 year old, records to be inaccessible). With each change, the institutional memory of a restoration project that was initiated becomes hazy.

Furthermore, urban dwellers' lack of botanical and habitat knowledge becomes a social problem for land stewardship. To the naïve eye, a city lot of invasive weeds is green and lush and defines Mother Nature. Most city dwellers have never seen historic or healthy vegetation, which is increasingly a pedantic concept in the urbanized world (Del Tredici 2010). Political leaders must explain the need for restoration to citizens who often are unaware that a problem exists (Chap. 30 includes further discussion on urban landscapes as learning arenas and locations of citizen stewardship for biodiversity and ecosystem services).

Public agencies absorb new responsibilities and, like living cell membranes, cannot always control the moieties that enter. This is particularly true in developing countries, where policy needs and economic conditions are rapidly changing (Chap. 27).

[4] www.njmeadowlands.gov

For example, land parcels may be administered by agencies that have neither ecological perspectives nor ecological staff. The potential for seeing and advancing ecosystem services can remain hidden. New professional teams are needed that include ecological perspectives to grab ecological value from nontraditional urban opportunities. Examples of urban projects that have acted on these opportunities are now widespread (Beatley 2000; Baycan-Levent and Nijkamp 2009).

In Dublin, Ireland, the city has approximately 2,000 ha of green spaces. Some are formal neighborhood parks and squares and others are habitat strips along the rivers and the Royal Canal, which dissect the city (Kingston, et al. 2003). The canals and rivers, in particular, have been identified as wildlife corridors from the suburbs to the city center and as an amenity useful for the major tourism economy of the city (Kingston et al. 2003). The 1999 city development plan requires at least 10 % of the area on new development projects to be green space (URGE-Team 2004). Here, ecological and economic advancement of the city are being seen as partners, necessary elements of urban planning.

In Turkey's largest city, there is new interest in restoring Istanbul's natural corridors from the forests in the hills above the city to the Marmara Sea edges by the Golden Horn (see Chap. 16 for a full local assessment of Istanbul). Similarly, progress is being made in urban habitat restoration in Izmir (Hepcan 2012). There are significant variations in natural community types that accompany the soil and elevation gradient above Istanbul. This corridor restoration also advances protection of local freshwater, which is a significant part of the city's drinking water supply. In addition, the required dredging of the waterways of the city can continually provide silt and mud towards the restoration of certain riverside microhabitats[5]. In one example, an old stone quarry was filled in, then planted with young trees to prevent future erosion into the waterway. This action increased the biotic health of the waterway as well as providing new terrestrial habitat. Engineering and restoration needs were both met. The natural resource initiatives were matched by other zoning advances; wastewater treatment plants were installed and industrial facilities removed away from the water edge to outside the city to increase the ecological character of the site. Over 100 ha were landscaped to increase the size of the greenbelt.

Finally, in Slovenia, the capital of Ljubljana has a large, 164 km^2 marshland area to the south, whose hydrology has been damaged by a new ring road network. New zoning is pushing back industrial and development activities away from the marshlands. Better public access to the remaining marshlands will help increase visits, affection and civic concern for the marshes through new recreational opportunities. Coordination among several municipalities that control the marshland area will facilitate informed management and persistence of the remaining wetland areas. Only through coordinated effort was the long-term sustainability of the lands possible (URGE 2004). In addition to the wetlands, the forests to the northeast side of Ljubljana are also being fragmented by a new roadway. There, new forest corridors are proposed to mitigate against the road's damage, and recombine fragments (Pirnat 2000). Transportation and environmental officials are cooperating for urban environmental health.

[5] www.ibb.gov.tr/en-US/

31.2.3.2 Financial Resources

Similarly, financial resources are often not available for the full time frame needed for ecological success. In developing countries, ecological restoration may not have the same urgency for funding as other needs such as public health and safety. Budgeting and ecological cycles spin at different speeds. If a restoration project is included as part of a capital construction project, an add-on to the building of a new road, for example, the monies that are contracted for the project must be spent within a specific time course. This may start at the beginning of a fiscal year, say July 1, and end 1–3 years later, on June 30. This stricture allows for careful government review of spending. However, restoration activities neither follow this particular timetable nor this short project lifespan.

Sometimes restoration sites must be revisited after several years, as local conditions change and additional plant species can be installed. The progress of ecological succession requires one suite of species to mature and facilitate the next wave of introductions after modifications of the habitat. Project money must be available several years into the future to complete this type of ecological restoration. Money managers must understand these ecological requirements. Ecologically savvy partners at other agencies must explain these needs. In these ways, political sophistication must evolve to nurture ecological complexity. In situations where elected officials want projects completed quickly, to display managerial skills, restoration may have to be explained as an investment, with some value now, but much more in the future.

There has been a series of new large-scale initiatives which are supplementing traditional landscape design principles with a modern appreciation for ecosystem services. For example, in India, coordination among cities and environmental institutions is leading to many new urban green spaces and sharing of information to drive municipalities to increase ecosystem services in the rapidly growing cities (CUGS 2012). Internationally, ICLEI, the global cities network, has resources and conferences to push the restoration agenda[6].

31.3 Conclusions: "The Tangled Bank," Joining Skills to Evolve a New Urban Ecological Future

An interest in ecosystem services within modern cities must consider the real possibility that the ecological past itself is extinct. The changed biotic and abiotic conditions have presented us with an unclear view of the future structure of urban habitats. We wish to have the functions of ecosystems but cannot with confidence understand what species mixes can reliably offer them. The various environmental constraints discussed here have each been successfully parried by new ecological links and actions in different parts of the world. The field of urban ecology rests on

[6] www.iclei.org

general ecological principles, but the protocols and project training needed to advance the restoration of ecological services in our cities require new disciplinary training.

So many ecologists enter the profession with love for natural lands and for an exploration of nature untrammeled by human actions. In fact, such areas are few now (or arguably non-existent), and tomorrow's environmental health can never be separated from the actions and areas of the human population. Urban environments are heterogeneous; each parcel a memory bank of past land-use and its accumulative effects. Rather than general procedures for urban ecological restoration, we may need a menu of solutions that must be sorted through to determine which remedies are appropriate for a specific site's history. The future biotic community on that site may be quite different from the historic vegetation because largely, people are not abandoning our cities (though see Chap. 12 for exceptions concerning shrinking cities). In fact, urbanization is growing at increased rates (see Chaps. 21 and 22).

Urban design teams and planning initiatives regularly include professions such as lighting, acoustical, and transportation planning; public safety; graphic arts; public outreach coordination; and structural and geotechnical engineers. Adding restoration of habitat structure and urban ecosystem services to urban planning requires the inclusion of ecological professionals, driven to the urban world. Changing institutional organizations and training is extremely difficult, but new methods can bring new efficiencies and new perspectives. The multiple advantages of urban conservation and restoration must be understood as a valuable partner to action in rural, "wild" settings (Dearborn and Kark 2010).

At the very end of Darwin's *Origin of Species,* he famously writes, *"It is interesting to contemplate a tangled bank, clothed with many plants of many kinds, with birds singing on the bushes, with various insects flitting about, and with worms crawling through the damp earth, …so different from each other, and dependent upon each other in so complex a manner…"* Here Darwin uses the metaphor of the "tangled bank" as a venue not of disorder and confusion, but as the location for the creation of new and useful forms. A city can also be a venue for new elements that grow out of the tight collaboration of professional skills, "so different from each other," that can include urban restoration ecologists. Darwin's biological example may be the proper metaphor for a new moment in urban planning and advancing ecological resources.

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Chapter 32
Indicators for Management of Urban Biodiversity and Ecosystem Services: City Biodiversity Index

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Abstract Capturing the status and trends of biodiversity and ecosystem services in urban landscapes represents an important part of understanding whether a metropolitan area is developing along a sustainable trajectory or not. However, this task also represents unique challenges for policy makers and scientists alike, challenges that lie at both the methodological (scaling, boundaries, definitions) and institutional levels (integrating biodiversity and ecosystems with social and economic goals). In this chapter we report on the experiences from municipalities in several countries where the newly developed City Biodiversity Index (CBI) has been applied and tested. The purpose here is not to compare or rank different municipalities but rather to deepen our understanding of the science underlying the

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indicators and contribute improvements to the CBI in different contexts. Based on experiences in implementing the CBI in 14 cities in Japan, and in Lisbon (Portugal), Helsinki (Finland), Mira Bhainder (India) and Edmonton (Canada) it is evident that the CBI has limitations that need to be addressed: (1) lack of data and the scale and boundaries need careful consideration, (2) the scoring represents a challenge as the bio-geographical differences or the profile of the cities varies largely, (3) the number and scope of ecosystems captured are limited and a broader range of ecosystem services should be included, and (4) the integrated social-ecological dimension of cities needs further development. However, it is also evident that CBI has some unique features, and can perhaps most importantly serve as both a tool that brings managers, scientists and other stakeholders together to act on the role of biodiversity and ecosystem services in the cities as well as a tool for assessing the impacts of different policies and land planning options on urban biodiversity.

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32.1 Introduction – History of Indicators

The development of environmental indicators dates back to the 1960s (OECD 1997). During the initial phase, environmental indicators were treated separately from other social and economic indicators, but since that time various frameworks have been designed to streamline different indicators in logical steps or in causal chains that include human dimensions. The PSR model (pressure-state-response) is one of the initial models from the 1990s. The framework later developed into the DPSIR model (Driving forces, Pressure, State, Impact, Responses), which has been widely used because of its logical structure and policy relevance (Kohsaka 2010). The Millennium Ecosystem Assessment (MA 2005) developed a framework to assess ecosystem change that integrated the concept of ecosystem services, thus emphasizing human well-being and allowing for the use of a wide range of indicators (Pereira et al. 2005). Within the intergovernmental process of IPBES (Intergovernmental Platform for Biodiversity and Ecosystem Services)[1] there is a development of a new comprehensive framework to assess ecosystem change.

Efforts to initiate such indicators have been taken by the Convention on Biological Diversity (CBD) which historically developed its own set of indicators for assessing the 2010 target of reducing the loss of biodiversity (Walpole et al. 2009). The failure to meet the 2010 target led the parties of the CBD to set new targets for 2020, the Aichi targets (CBD Decision X/2), and the development of indicators for these targets is an ongoing process (GEO BON 2011; SCBD 2011). Biodiversity indicators need systematic observations, both on the ground and from remote sensing, and these must be possible to aggregate, in order to provide accurate information on global biodiversity change (Pereira and Cooper 2006). A global biodiversity observation network to provide the data needed for biodiversity indicators, the scientific community, international conventions and IPBES is now being developed under the auspices of the Group on Earth Observations Biodiversity Observation Network (Scholes et al. 2012; Pereira et al. 2013). Indicators were originally designed to span national to global scales, and

[1] www.ipbes.net

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integrated into a DPSIR framework (Butchart et al. 2010; GBO3 2010) but it has been repeatedly emphasized that there is a need for a set of scalable indicators, which could be used for upscaling of observations from local to global scales as well as downscaling (SCBD 2011; UNEP 2011).

Other types of environmental indicators have, in a few cases, been designed at the scale of municipalities and cities (Mori and Christodoulou 2012). Such indicators have sometimes been framed as "quality of life" indicators (Chan et al. 2005); sometimes they have been developed in the context of "Local Agenda 21" initiatives or in association with a general "sustainability index" (e.g., Mori and Christodoulou 2012). Such indices may have been broken down into individual environmental, social and economic indicators, but in general lacked a connection to biodiversity and ecosystem services.

It has thus become apparent that in the emerging initiatives by local governments engaging in implementing the CBD Aichi targets and the Plan of action on subnational governments, cities, and other local authorities for biodiversity (CBD Decision X/22),[2] that a set of indicators specifically designed to the spatial scales of municipalities, rather than those of nations and larger regions was lacking and urgently needed (see CBI 2012).

32.2 The City Biodiversity Index (CBI)

32.2.1 The History of CBI

The City Biodiversity Index (CBI), also known as the Singapore Index on Cities' Biodiversity (SI) is a tool designed to allow cities to monitor and evaluate their progress and performance related to conserving and enhancing biodiversity and ecosystem services (CBI 2012). The idea for the CBI was proposed in 2008 and the development of CBI has been led by the Secretariat of the CBD, in collaboration with the Global Partnership on Local and Sub-national Action for Biodiversity, the Government of Singapore, and partners from academic institutions, international organizations and civil societies.

A first technical expert workshop on the CBI was held in Singapore in February 2009. Key considerations in developing the index were its ease of use by cities, scientific credibility, and objectivity. The draft CBI comprised 25 indicators divided into three components: (1) native biodiversity in the city, (2) ecosystem services provided by biodiversity in the city, and (3) governance and management of biodiversity in the city. The rationale for these components was the need by city officials and civil society to know what biodiversity exists in their city, and its importance in terms of providing ecosystem services (such as regulation of climate or water). Governance and management were also viewed as an important component of the index, as these are the means by which cities enhance their biodiversity efforts. A quantitative scoring methodology based on a scale of 1–4 points per indicator was developed. The first version of the CBI User's Manual was made available in September 2009 on the CBD website,

[2] www.cbd.int

and cities were invited to test the index. A second technical expert workshop, held in July 2010 in Singapore, reviewed the experience of cities that had tested the index. Participants made key revisions, including streamlining the number of indicators from 25 to 23 and fine-tuning the scoring, and a revised User's Manual was made available. On 29th of October 2010, the Plan of Action on Sub-national Governments, Cities, and other Local Authorities for Biodiversity, was endorsed by 193 CBD parties through Decision X/22 at COP11 in Nagoya. The plan included suggestions that CBI be used by local and sub-national authorities to support the local implementation of the Aichi targets. A third technical expert workshop was held in October 2011 in Singapore. As data were available from only 14 cities for the seven indicators that require scoring ranges to be determined, participants agreed that a larger sample size was required before an appropriate statistical methodology could be adopted and the scoring ranges determined. There is now a third revision of the CBI available (CBI 2012).

32.2.2 The Structure of the CBI

The CBI indicators (Box 32.1) are broad and designed to meet three important criteria: (1) to be a comprehensive tool for assessing not only biodiversity, but also ecosystem services, governance and management; (2) to be a self-assessment tool, as it is not intended for comparisons between cities; and (3) to be a simple but yet scientifically credible tool.

Box 32.1 City Biodiversity Index

List of Indicators:

1.	Proportion of natural areas
2.	Connectivity measures or ecological networks to counter fragmentation
3.	Native biodiversity in built-up areas (bird species)
4–8.	Change in number of native species (4. vascular plants, 5. birds, 6. butterflies, 7. and 8. optional)
9.	Proportion of protected natural areas
10.	Proportion of invasive alien species
11	Regulation of quantity of water
12	Climate regulation: carbon storage and cooling effect of vegetation
13–14.	Recreational and educational services
15.	Budget allocated to biodiversity
16.	Number of biodiversity projects implemented annually
17.	Rules, regulations and policy – existence of local biodiversity strategy and action plans
18–19.	Institutional capacity
20–21.	Participation and partnership
22–23.	Education and awareness

(continued)

Box 32.1 (continued)

The CBI's current 23 indicators are viewed as core indicators and optional or sub-indicators can be developed as necessary and tailored to specific monitoring needs of individual cities. For each indicator, the CBI manual (CBI 2012) proposes a scoring of 0–4 points, where 0 corresponds to poor performance and 4 points corresponds to excellent performance. Points can be summed to provide an overall score of the city's biodiversity performance. For some of the indicators, the conversion of the measurements to the score grade have been already proposed by experts, and for others a statistical analysis of incoming CBI data from the cities will be used to determine the scoring ranges.

32.3 Evaluation of Experiences with the CBI

32.3.1 Experiences in Japan

The background of application of the CBI in Japan was a new law, titled Basic Law on Biological Diversity (*Seibutsu Tayousei Kihon -ho*), that was introduced in 2008 as a parliamentary act. In Article 13 of the Law, municipalities (prefectures, cities and other local units) were called upon to develop their local biodiversity action plans. The Ministry of Environment has been leading the process with plans to develop a handbook for the municipalities including instructions on the use of specific indicators to promote development of local biodiversity strategy and action plans. The CBI has subsequently been applied in 15 cities, and in this chapter we report on two specific applications, in Yokohama and Kanazawa. We also provide a summary of applications in 13 mid to large Japanese cities (details given in Appendix I).

32.3.1.1 Experiences in Two Cities: Yokohama and Kanazawa

The City of Yokohama is the second largest city in Japan with a population of approximately 3.7 million. The steady population growth in the city has led to a decrease in green spaces from 50 % in 1970 to 30 % in 2009. Most parts of the city are dotted with forest and farmland (thus embracing dynamic water and green environments), and while the city has experienced a steady loss in green coverage, it has developed a variety of innovative, biodiversity-related measures and plans based on principles of multi-stakeholder engagement.

A study to draw experiences from the application of CBI was conducted by the United Nations University – Institute of Advanced Studies (UNU-IAS) in cooperation with the City of Yokohama. The Yokohama experience shows that one of the biggest challenges in applying the CBI was identifying key variables of biodiversity and

Table 32.1 Summary of experiences of the application of CBI in 13 cities

CBI indicators		Challenges
1–3	Areas	Definition of natural areas and fragmentation
4–8	Native species	Data availability
9	Protected areas	Definition of protected areas
10	Invasive species	Unavailable data/Unreported activities
11–13	Ecosystem services	Difficult to calculate
15–22		Distinctions between general greening and biodiversity-specific activities or budgets were unclear. These include planting trees (with non-native species), recycling, etc.
		Difficulties to capture activities in schools because information is not disclosed openly
General comments		For urban biodiversity, increase in conservation activities does not necessarily correspond to improvements in indicators and it is difficult to set benchmarks to measure impact and performance in urban contexts.
		Number of indicators too high and to limit to "core" indicators

Modified from information included in Kohsaka and Okumura (2014) and Inoue and Morimoto (2011)

ecosystem services for the city, along with data availability. Yokohama municipality has conducted extensive citywide extensive surveys of terrestrial species only twice in the past several decades, and due to budgetary constraints it was viewed as unrealistic to carry out such extensive surveys on a regular basis. A second challenge involved the governance indicators. Biodiversity-related activities and budgets are most often embedded in multiple other sectors of the city administration, and are difficult to separate out and report. An interesting initiative taken by Yokohama municipality is the incorporation of biodiversity into their environmental management system called ISO14001, thus aiming to minimize the impact of human activities. Through ISO14001, the issues of biodiversity are addressed in the agendas of each department and section in the city. However, it proved difficult to capture such an initiative through the current indicators. Also, while many of the current indicators may be able to report the magnitude of efforts (e.g., budgetary and personnel) the city has made for addressing the biodiversity issues, they fail to show if such efforts were successful, effective or influential. Nevertheless, it demonstrated, in particular, the validity of CBI as a tool to keep track of progress of the Yokohama's biodiversity action plan and facilitate discussions on a way of achieving its targets.

In Japan, the CBI was tested in 13 mid- to large-size urban areas with a qualitative approach analyzed by Kohsaka and Okumura (2014) and with quantitative methods analyzed by Inoue and Morimoto (2011). A summary of the main results of these studies is given in Table 32.1 (see Appendix I for further details). Some of the challenges faced in the application of the CBI were related to the need for clearer definitions of indicators for the following terms (cf. Table 32.1) *Indicator 1* – natural and semi-natural areas, *Indicator 2* – fragmentation, and *Indicator 9* – protected natural area. Additionally, methodological challenges included evaluation of *Indicator 2* – fragmentation and *Indicators 4–8* (native species). In some cities (Chiba, Kawasaki, Kyoto, and Osaka), basic data of native species were totally unavailable; this identified a need for an assessment and monitoring of the native species in these areas. For *Indicator 12*, the effects of heat-islands or cooling effects proved difficult to calculate in some cases. At the same time, positive remarks were expressed by city officials; they suggested that these data could be used for housing or city planning issues once the indicators are set in place (Kohsaka and Okumura 2014).

Further implementation challenges were presented in making distinctions between general environmental and biodiversity-specific city activities and budget allocation; this proved difficult, irrespective of city size (Kohsaka and Okumura 2014; Inoue and Morimoto 2011). The number of indicators was also viewed to be too high to handle for small- to mid-sized cities due to limitation of human resources.

Kanazawa, the capital of Ishikawa Prefecture (population 460,000) located in the northwest of Japan has experienced a high rate of urban development since the 1970s. In general, a dichotomy between humans and nature is not at all evident in Japanese traditional thinking and landscape management (Duraiappah et al. 2012), and the suggestion from the Kanazawa experience was that local versions of CBI could developed with locally adapted forms of the indicators, reflecting the uniqueness of individual cities in different ecological and cultural contexts (UNU-IAS OUIK 2011). In Kanazawa, unique conditions include the longstanding, traditional agricultural activities that are part of the ecosystem, such as ponds and marshes used for agriculture or charcoal production activities. The richness of agro-biodiversity was perceived as particularly important and the biodiversity of the social-ecological production landscape of *satoyama*, was thought to be inadequately captured in the CBI. For more information on *satoyama* landscapes, see the local assessment of Chap. 8 (Fig. 32.1).

32.3.2 Lisbon, Portugal

Lisbon is the capital of Portugal, located on the Atlantic Ocean coast in Southwestern Europe. The city has a resident population of 550,000 in an area of 85 km^2, but the greater metropolitan area has a population of approximately three million people. Due to the relatively small number of green areas inside the city and dense urbanization, Lisbon has been classified as a brown city in a green background (EEA 2010). However, the metropolitan region is composed of several Natura 2000 sites, including one of the most important bird areas in Europe (Tejo Estuary), and agricultural and forest areas.

To celebrate the 2010 International Year of Biodiversity, the municipality of Lisbon decided to set an aspirational target for 2020 of increasing the biodiversity in the city by 20 % relative to its 2010 levels. The establishment of this target set in motion two important processes: (1) the definition of indicators to assess the target (operationalizing the target into measurable indicators, such as the proportion of semi-natural areas in the city or the number of native species commonly seen in the

Fig. 32.1 In Hakusan, a suburb of Kanazawa, Japan, forests serve as a place for environmental education (Photographed by and published with kind permission of © Ryo Kohsaka 2013. All Rights Reserved)

city), and (2) the development of a municipal biodiversity strategy. To develop these processes, an expert group was established, composed of representatives of the Municipality of Lisbon (CML), the Institute for the Conservation of Nature and Biodiversity (ICNB), the Municipal Environmental Agency (Lisboa E-NOVA), and scientists from the University of Lisbon. The expert group decided to base its indicator framework on the CBI, in order to build on the work being done by other cities, and to facilitate indicator harmonization in global assessments. The expert group worked for 1 year to estimate values for the 23 indicators of the CBI, mainly from compilation and GIS analysis of existing data (Appendix II). It was found that the CBI addressed most of the dimensions that the expert group wanted to cover, but there were several challenges in its application.

The first challenge was related to the concept of naturalness. There are no natural areas left inside the municipality of Lisbon (with the possible exception of the mud intertidal areas in the river front), but there are areas in the process of renaturalization. These areas include large portions of the city forest park of Monsanto (with significant areas still covered by exotic trees, despite forestry practice changes in the last 20 years that promote native tree recruitment), and abandoned areas and other semi-natural areas (that are in some cases planned for future development). The second challenge was related to the use of species number as an indicator.

Fig. 32.2 An urban garden near the historic center of Lisbon (Photographed by and published with kind permission of © Henrique M. Pereira 2013. All Rights Reserved)

Species number has been shown to have limitations as a biodiversity indicator, and it has been suggested that indices based on species abundance such as the geometric mean abundance have better statistical properties (van Strien et al. 2012). Another problem is that species lists tend to be cumulative, so the expert group restricted species counts to species occurring between 2005 and 2010 (Appendix II). A third challenge was that the ecosystems service indicators and the connectivity indicators are in an early stage of methodological development. In response, the Lisbon expert group proposed several sub-indicators that can inform on the condition of biodiversity and ecosystem services, and which can be adopted by other cities applying the CBI (Appendix II). A fourth issue was that the governance and management indicators were relatively numerous and sometimes hard to assess precisely. For example, the city statistics and reports do not always make the distinction between general public parks investment or other environmental activities and biodiversity-specific activities. Finally, the Lisbon expert group did not apply the 4-point CBI scores to each indicator, as the experts felt it was subjective and did not further the monitoring goals. Instead, the numerical values of each indicator were calculated and reported (Appendix II) (Fig. 32.2).

Nonetheless, beyond the numerical value of the indicators, the implementation of the CBI in Lisbon fostered collaboration between several institutions and experts on monitoring biodiversity change and management of biodiversity. It also led to the development of a Biodiversity Strategy for Lisbon and a Local Action Plan, which hopefully will contribute to achieve the broad target set by the municipality for 2020.

32.3.3 Helsinki, Finland

The city of Helsinki is located in southern Finland by the Baltic Sea. Numerous green areas enrich the scenery of Helsinki and the structure of the city is widely dispersed. The city's government has made a decision to maintain the city's biodiversity even as the city grows rapidly. To support and monitor this goal, the city is searching for standardized indicators for biodiversity assessment. The CBI is one potentially useful set of indicators. A study on the availability of data for calculating CBI indicators – a feasibility study – concluded that it is possible for Helsinki to participate in the CBI, but required data are incomplete. Data exist for some of the indicators, such as *Indicator 9* (proportion of protected natural areas), *Indicator 19* (number of city agencies involved in inter-agency cooperation) and *Indicator 21* (number of organizations with which the city is partnering in biodiversity activities). However, for many indicators (e.g., *Indicators 2*, *4–8* and *10–12*), collection of new data is required.

Scores for the indicators have not been calculated in Helsinki yet, but a rough estimate has been produced for *Indicator 1* showing that the proportion of natural areas in the city is about 40 %, which is well above the highest score (4 points: >20 %) for the indicator. However, the value of the indicator depends very much on exactly how 'natural area' is defined and whether the total area (including sea area) or only the terrestrial area of Helsinki is considered.

Another problem is that for many indicators it is unrealistic – for the reason of limited resources – to monitor changes in the whole city, but the CBI requires that samples need to be taken (e.g., *Indicators 4–8* on changes in number of native species). In such cases an alternative would be to use the gradient approach, i.e., select sampling sites along a gradient from the city center through suburban areas to the outskirts of the city (see Chap. 10). This would also enable the cities to use reference areas outside the city to find out whether observed biodiversity changes take place within the city only or in larger geographical areas. The gradient approach would also enable studies comparing changes along the gradients between cities without comparing the cities directly. For example, this kind of an approach has been successfully used to study changes in carabid beetle assemblages along urban-rural gradients in several cities across the world (Niemelä and Kotze 2009).

The assessment of the use of the CBI in Helsinki also highlighted some more general issues regarding the index. For example, the temporal span of measurements of certain indicators pose challenges. For example, the time span of 3 years for monitoring change in the number of native bird species (*Indicator 5*) was considered by the city's biologists too short to show significant changes in population sizes and ranges. A longer time span of 5–10 years was suggested. Corresponding increase in the time frame of other similar indicators (*Indicators 4–8*) was suggested to show changes in populations sizes and ranges. Moreover, most of the administrative area of the City of Helsinki is, in fact, water (Baltic Sea), which impacts the scores of the area-related indicators. A specific indicator for cities with considerable sea areas (for example, an indicator measuring marine biodiversity) should be considered. It also became apparent that the flow of information between the cities participating in CBI should be enhanced for useful comparisons, and information about how different cities have tested and provided their preliminary scores should be made available for participants and potential participants of the CBI.

32.3.4 Mira Bhainder, India

Mira Bhainder is a small but rapidly expanding city to the north of Mumbai, India. Due to its proximity to Mumbai, India's commercial capital, this formerly peri-urban area has grown into a city in the past decade and now has its own administrative municipal body. Many of Mira Bhainder's residents travel to neighbouring Mumbai for work. The built-up areas are concentrated around the center of town, while the periphery is dotted with settlements surrounded by secondary growth deciduous forest patches and plantations. Mira Bhainder spans an area of 91.9 km^2, more that 40 % of which includes part of a national park and stretches of mangrove forests.

Terracon[3] introduced the City Biodiversity Index to the city administration of Mira Bhainder with a proposal to apply the Index to the city, and Mira Bhainder became the first city in India to apply the CBI. Terracon required about 2 months for conducting this exercise with multiple personnel from various fields ranging from biodiversity experts, GIS specialists and planners. Most of the raw baseline data required for spatial analyses was available from the city municipal corporation. However, the data did not clearly define boundaries of natural areas such as those between mangroves and saltpans, forest patches, etc. Terracon defined these boundaries with the help of open source Google images and also from results of previous projects (Fig. 32.3).

There were multiple challenges in applying the CBI to Mira Bhainder. One was the paucity of baseline data on biodiversity. The difficulties with calculating *Indicators 3–8* led the indicator team to suggest to the city administration the need for more detailed baseline biodiversity surveys. Making the city administration conduct more biodiversity surveys would also help to mainstream biodiversity in the planning process, as well as indirectly help raise awareness about biodiversity.

[3] Terracon Ecotech™ is an ecological solutions provider based in Mumbai.

Fig. 32.3 Live and Let Live: It is remarkable to see great egrets (*Casmerodius albus*) nesting atop a raintree (*Albizia saman*) in the center of town, as seen from the terrace of Mira Bhainder Municipal Corporation's Garden Department office. Surrounded by residential complexes and offices, it is symbolic of human populace and biodiversity living side by side (Photographed by and published with kind permission of © Salil P. Kawli 2013. All Rights Reserved)

32.3.5 Edmonton, Alberta, Canada

Edmonton is the capital of the Province of Alberta in western Canada in the northern part of the Great Plains of North America. Edmonton is a relatively young city and still has a significant area of agricultural land, but significant growth pressure is resulting in the conversion of farmland and natural patches to urban development. Approximately 10 % of Edmonton's area is in a natural state (i.e., a predominance of native vegetation in naturally occurring patterns) (City of Edmonton, 2007, Natural Connections Strategic Plan).

Edmonton's relatively low biodiversity is related to its climate. It is one of the coldest cities using the CBI, and the scores for the biodiversity and ecosystem services components are low when compared to most other cities – particularly cities located in tropical and Mediterranean ecosystems. This highlights the fact that the index is a primarily a self-assessment tool and that caution is necessary when comparing cities. Nevertheless, the CBI is an important tool locally to provide feedback to local decision makers in Edmonton on the effect of city policies on biodiversity over time.

In contrast to the biodiversity and ecosystem services component of the CBI, the sub-scores of the governance component of the index provide meaningful insight when compared to other cities and are useful for benchmarking programs and initiatives. However, there are some caveats. For example, the area of protected natural spaces can vary greatly between cities, depending on whether the local authority has sufficient enabling legislation to protect nature or must protect nature with its own budget from the tax roll. In addition, some cities have federal and provincial/state protected areas within its boundaries, which can boost the scores significantly. In addition, regional governments have a higher probability of better scores than single cities because the catchment area is much larger and often includes undeveloped lands.

Populating the CBI with data has proven to be a catalyst for accelerating innovation in Edmonton. The CBI is a potent community engagement tool. In order to gather data for the species indicators, The City of Edmonton brought together many citizens and groups with specialized knowledge of the number of individual species in the area to provide the first comprehensive species list in the city. These relationships have developed and grown. In order to meet the challenge of calculating the impervious area of the city, the Office of Biodiversity acquired its first satellite imagery, which has yielded positive results in other areas as well.

Although Edmonton has found some limitations to the index, these limitations can be overcome with the addition of indicators to supplement the CBI. The Office of Biodiversity also maintains an additional suite of indicators to manage the effectiveness of policies and programs. Other limitations of the index include:

- The species indicators do not register change until a species has been lost. Edmonton is working on a finer grain estimate of species change.
- The number of formal educational visits to natural areas is not tracked in Edmonton and many neighborhoods have been designed to include natural areas and schools that are adjacent to each other, so formal visits can be frequent.
- The budget allocated to biodiversity annually is extremely hard to estimate for local authorities like Edmonton where biodiversity functions exist in a highly integrated management system.
- It is only possible to get a rough estimate of the number of outreach and public awareness events held in the city each year because of the large number of non-profit organizations and other institutions involved in this work. The recently created Edmonton Biodiversity Network should help Edmonton in the future.

32.4 Challenges Ahead

The experiences from these cities show that there are multiple potential benefits of the CBI in promoting conservation and sustainable use of biodiversity at the local level. For example in the Japanese cities, the application of the CBI promoted intersectional dialogue across different departments in the cities, which otherwise would not have communicated.

There may be a general pattern here, whereby sharing, interpreting and reflecting on the results among different departments for the improvement of their daily administration work may facilitate internal communications and improve the capacity of the local government. Also, through quantifying biodiversity and ecosystem services, and evaluating their changes over time, the CBI may motivate various stakeholders to recognize their connections with biodiversity, register concern, and take action for stewardship. In addition, the CBI may enable the local government to establish a system to address urban sustainability more generally, particularly when indicators are linked to numerical targets in plans or strategies of the city (see Chap. 33 for further discussion on the future implications for sustainability).

The practical application challenges of the CBI are many, but could be summarized as relating to: (1) the lack of data; (2) the scale, boundaries, and definitions; (3) the scoring that needs to capture the vast bio-geographical differences among cities; and (4) the number and scope of ecosystem services are limited. The lack of data is a challenge but also a motivation: the CBI can provide incentives for municipalities to start making inventories and monitoring programs of their biodiversity. For example, it is today possible to integrate remote sensing data and *in situ* observations to monitor several essential biodiversity variables such as habitat structure and phenology (Pereira et al. 2013).

In this context, municipalities should explore the possibilities of launching citizen science projects (see Chap. 30) and consider the possibility in general that within cities local knowledge on biodiversity and ecosystem services may reside in many different groups within civic society (for a general overview, see Chap. 30). Another general issue reported by many of the cities analyzed here was that the number of indicators was too large. We feel that a revision of the CBI should try to reduce or merge indicators, particularly in the governance section because institutional arrangements such as the budget, number of activities, and existence of departments overlap with one another.

The challenges related to scaling, boundaries, locally adapted indicators and scoring can be met by each municipality developing their interpretation of what scale and what boundary is the most appropriate, what definitions to use, and what set of sub-indicators may best reflect the local ecological and cultural context. However, there are some challenges that are not easily addressed at the municipal level and need input from the research community. One important challenge is related to the development of indicators that could complement or even replace some of the species-richness-based indicators. Recent work on the identification of essential biodiversity variables (Pereira et al. 2013) suggests that important variables to measure are species abundances, species traits, and ecosystem structure. Monitoring of how urbanization and changes in habitat structure may result in changes in species abundances (Pereira and Cooper 2006) and losses and gains of functional traits (Cornelissen et al. 2003; Lavorel et al. 2007) will be very important. Grouping species according to functional type characteristics builds on the assumption that these groupings share similar resource-use patterns and ecosystem roles, and are responding in similar ways to environmental conditions or disturbance. Thus, functional types could potentially be extremely useful management tools where indicator types could be generated and predictive models on changes in generation of ecosystem services could be developed. Furthermore, a functional type approach allows for regional comparisons with the formation of a common language through which taxonomically distinct and complex systems can be effectively compared. So far, such analyses have been carried out in a large number of habitat types, except in the urban landscape (Chap. 10).

Another challenge relates to expanding the part of ecosystem services in the CBI, and here much further research needs to be done (for a further discussion on urban ecosystem services, see Chap. 11). In contrast with indicators that emphasize the biological component of ecosystems, such as species extinction risk or trends in invasive species, indicators for ecosystem services have to include a social dimension, as ecosystem services are produced by an interconnected social-ecological system rather than by ecosystems alone (Reyers et al. 2013). Measuring ecological properties and functions alone will not provide an adequate picture of ecosystem service status and trends; rather, a significant input of additional social and economic data will also be required. These elements are reflected in the conceptual framework of the CBI, which aims to capture changes in benefits of services, impacts in human well-being and effects of policy, but needs to be further developed. A second challenge with ecosystem service indicators is related to the interactive characters of bundles of ecosystem services (i.e., a tight positive or negative correlation among sets of services). Such correlations mean that when managing for the increase of a particular service, others may increase (synergy) or decrease (trade-off) simultaneously. Such synergies and trade-offs are poorly documented, and the evaluation of trends in ecosystem services in the CBI over longer periods of time is of special interest (because patterns of trade-offs among services and different trends in the responses of services to certain management schemes may be revealed). Furthermore, it has been stressed by many applying the CBI that indicators capturing the flow of ecosystem services from more distant ecosystems beyond the city would be desirable to include, in order to assess the impact that cities and their inhabitants and policies have on ecosystems elsewhere (cf. Seto et al. 2012; Seitzinger et al. 2012).

Despite these challenges, the CBI is a powerful tool for increasing the importance of biodiversity in city management. The CBI can bring managers, scientists and other stakeholders together to think about the role of biodiversity in the city. The impacts of different policies and land-planning options on biodiversity can be assessed with the CBI. We hope that as more cities develop local action plans and strategies in response to the call of the CBD (Decision X/22), the CBI will be further developed and enriched with experiences around the world, and biodiversity management will come to the forefront of city planners' concerns and help improve the well-being of all urban dwellers.

Appendices

Appendix I

General outcome of CBI application to Japanese cities

CBI	No.	Indicator		Sapporo	Sendai	Chiba	Tokyo Wards	Yokohama	Kawasaki	Nagoya	Kyoto	Osaka	Kobe	Hiroshima	Kitakyusyu	Fukuoka
1.Native biodiversity	1	Propotion of Natural areas in city		60.7	56.8	19.4	0.0	9.0	5.6	3.7	73.8	0.0	42.2	68.7	40.1	33.3
	3	Native Biodiversity in Built-up areas	(Number of Bird species)	275	89	413	259	65	338	100	300	175	279	278	316	316
	4-8	Changes in Nmber of Native Species	vascular plant	1365	315	1898	1475	1375	2316	1020	2460	2436	2396	2288	1528	2270
			birds	275	89	390	259	65	338	272	300	175	279	278	316	316
			mammals	40	32	25	15	9	40	29	48	33	28	41	26	40
			reptles	8	9	14	12	9	20	14	16	20	19	16	11	16
			amphibians	6	11	12	6	7	16	12	22	18	17	16	14	15
			freshwater fish	46	15	74	55	48	83	63	114	130	62	89	77	
			insect	2902	166			206		3524	6413	5567	3807	3767		1100
	9	Propotion of Protected Areas	(%)	8.49	42.77	3.04	0.14	2.28	0.63	0.63	40.51	0.00	17.33	0.86	18.53	0.34
	10	Propotion of Invasive Alien Species	vascular plant	2.9	15.2	3.0	3.9	4.1	2.5	5.7	2.4	2.4	2.6	2.4	3.5	
			birds	0.0	1.1	0.3	1.2	6.2	1.2	0.7	0.6	0.8	0.4	0.4	0.6	0.6
			mammals	5.0	3.1	8.0	20.0	44.4	7.5	3.4	4.2	9.1	7.7	2.4	3.8	3.0
			reptles	12.5	11.1	14.3	16.7	22.2	10.0	14.3	12.5	10.0	11.8	12.5	18.2	11.1
			amphibians	16.7	9.1	8.3	16.7	14.3	6.3	8.3	4.5	5.6	6.3	6.3	7.1	7.1
			freshwater fish	19.6	40.0	13.5	12.7	14.6	8.4	11.1	5.3	5.4	14.8	9.0	7.8	8.5
			insect	0.0	0.0			1.5		0.0	0.0	0.0	0.1	0.0		0.0
2.Ecosystem Service.	11	Regulation of quantity of water		77.0	78.6	49.0	24.0	35.1	37.9	25.0	83.0	25.0	69.0	75.0	65.0	60.0
	12	Climate regulation:Carbon strage and Cooling effect of vegetaion.		73.8	62.0	33.8	22.8	23.9	19.1	20.6	79.7	5.0	60.8	71.5	59.7	53.6
	12	Recreational and Educational Service		6.1	33.8	1.7	0.3	0.7	0.4	0.8	23.2	0.4	7.9	1.5	10.4	1.0
3.Governance and Management	15	Budget allocated to biodiversity		2.2	2.8	3,2	0.9	2.5	4.3	3.8	2.3	2.6	3.0	0.6	3.0	3.8
	17	Rules,Regulations and Policy		×	×	×	×	×	×	○	×	×	×	×	×	×
	18	Institutional capacity		16	18	9	35	12	4	15	6	9	7	3	6	5
	20	Participation and Partnership		○	○	○	○	○	○	○	○	○	○	○	○	○
	22	Education and Awareness		○	○	○	○	○	○	○	○	○	○	○	○	○

score	
	4
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	2
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Appendix II

The City Biodiversity Index for the municipality of Lisbon in 2010

Indicator	Interpretation note		Value
1. Proportion of Natural Areas in the City	In the municipality there are no pristine areas. Based on areas naturalized by abandonment (921 ha) and forested areas where the long-term goal is renaturalization (936 ha)		22 %
2. Connectivity			
3. Native biodiversity in built-up areas	Number of species	Birds	76
4–8. Native biodiversity in the city	Number of native species with confirmed occurrence between 2005 and 2010	Vascular plants	342
		Fungi	140
		Birds	126
		Mammals	19
		Amph. & reptiles	28
		Fish	45
9. Proportion of protected areas	These are the areas in Lisbon that have to be managed as forest areas		16 %
10. Invasive species	Number of species	Vascular plants	32
		Birds	4
11. Water cycle regulation	Soil permeability is used as a proxy for this ecosystem service		39 %
12. Climate regulation	Forest cover		1,352 ha
	Street trees		190 km
	Proportion of tree canopy cover		18 %
	Carbon sequestration		5,144 t CO2/year
13. and 14. Recreation and education	Recreation was calculated based on all green areas in the city (3,369 ha)	Green area per inhabitant	27 m^2
	No available data for educational services	Population lacking neighborhood green areas	380,000
15–23. Governance and management	Annual budget allocated to the municipal department of environment and public spaces (only a part of which is spent on biodiversity management)		46 M€
	Number of institutions related to biodiversity		102
	Number of information and educational actions promoted by the municipality on biodiversity		811

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Chapter 33
Stewardship of the Biosphere in the Urban Era

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33.1 Introduction

We are entering a new urban era in which the ecology of the planet as a whole is increasingly influenced by human activities (Turner et al. 1990; Ellis 2011; Steffen et al. 2011a, b; Folke et al. 2011). Cities have become a central nexus of the relationship between people and nature, both as crucial centres of demand of ecosystem services, and as sources of environmental impacts. Approximately 60 % of the urban land present in 2030 is forecast to be built in the period 2000–2030 (Chap. 21). Urbanization therefore presents challenges but also opportunities. In the next two to three decades, we have unprecedented chances to vastly improve global sustainability through designing systems for increased resource efficiency, as well as

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through exploring how cities can be responsible stewards of biodiversity and ecosystem services, both within and beyond city boundaries.

A social-technological approach has, up until now, been a traditional way of analyzing urban complexity (e.g., Geels 2011; Hodson and Marvin 2010), and in this context many have struggled to define exactly what is meant by a city. In this volume, we have expanded on an emerging framework of cities as complex social-ecological systems, since cities include much more than a particular density of people or area covered by human-made structures. A social-technological approach will continue to be important in the urban sustainability discourse. However, an urban social-ecological approach (Berkes and Folke 1998) will be increasingly

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necessary to succeed in enhancing human well-being in urban areas in the face of new and complex challenges such as climate change (Ernstson et al. 2010a; Chelleri and Olazabal 2012), migration (Seto 2005), and shifting and globalized economic investment (Childers et al. 2013). Furthermore, the research and application of urban sustainability principles have until now rarely been applied beyond city boundaries and are often constrained to either single or narrowly defined issues (e.g., population, climate, energy, water) (Marcotullio and McGranahan 2007; Seitzinger et al. 2012) (Chap. 27). Although local governments often aim to optimize resource use in cities, increase efficiency, and minimize waste, cities can never become fully self-sufficient. Therefore, individual cities cannot be considered "sustainable" without acknowledging and accounting for their dependence on ecosystems, resources and populations from other regions around the world (Folke et al. 1997; McGranahan and Satterthwaite 2003; Seitzinger et al. 2012). Consequently, there is a need to revisit the concept of sustainability, as its narrow definition and application may not only be insufficient but can also result in unintended consequences, such as the "lock-in" of undesirable urban development trajectories (Ernstson et al. 2010a). We suggest that an appropriate conceptualization of urban sustainability is one that incorporates a complex social-ecological systems perspective of urban areas and their global hinterlands, and one that recognizes that urban areas are embedded in, and are significant parts of, the operation of the biosphere. The focus is not just on sustainability goals or aspirations, but also on resilience and transformations as components of the urbanization process.

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As human dominance of ecosystems spread across the globe, humankind must become more proactive not only in trying to preserve components of earlier ecosystems and services that they displace, but also in imagining and building whole new kinds of ecosystems that allow for a reconciliation between human development and biodiversity. Populations and assemblages of species that evolve under urban conditions may well represent what holds for much of Earth's terrestrial biodiversity in the future.

We hope this book stimulates a continued debate and knowledge generation about beneficial and necessary responses to urbanization, as well as provides support for a process that moves from knowledge to action. In this chapter we will: (1) summarize the main insights from the preceding chapters of the book; (2) outline current gaps in knowledge; and (3) discuss local, regional, and global strategies and actions for the urbanization process to become more sustainable, and; (4) a new framework for increasing our understanding of urbanization, sustainability, resilience, and transformation.

33.2 Summary: Global Urbanization and Impacts on Biodiversity and Ecosystem Services

33.2.1 Urban Trends

If current trends continue, by 2050 the global urban population is estimated to be 6.3 billion, up from the 3.5 billion urban dwellers worldwide in 2010 (Chap. 21). This is likely to have major impacts on biodiversity and ecosystem services (Chap. 22). We have, in this context, identified five trends in the global urbanization process (Chap. 1):

1. Urban areas are expanding faster than urban populations (Chap. 21). If current trends continue, between 2000 and 2030 urban land cover is expected to triple, while urban populations are expected to nearly double. Most of the growth is expected to happen in small and medium-sized cities, not in megacities.
2. Urban areas modify their local and regional climate through the urban heat island effect and by altering precipitation patterns, which together will have significant impacts on local and regional net primary production, biodiversity and ecosystem functions (Chap. 25).
3. Urban expansion will heavily draw on natural resources, including water, on a global scale, and will often consume prime agricultural land, with knock-on effects on biodiversity and ecosystem services elsewhere (Chaps. 3, 10, 23 and 26).
4. Urban land expansion is occurring rapidly in areas adjacent to biodiversity hotspots, and faster in low-elevation, biodiversity-rich coastal zones than in other areas (Chaps. 3 and 22).
5. Future urban expansion will mainly occur in regions of limited economic and institutional capacity, which will constrain management of biodiversity and ecosystem services. Half the increase in urban land across the world over the next 20 years will occur in Asia, with the most extensive patterns of change expected to take place in India and China (Chaps. 4, 21, 22 and 27).

All these projections, however, have uncertainties and are susceptible to several factors or events—for example, a deep and protracted world economic crisis, accelerating fossil-fuel prices, or a global pandemic—that could considerably decrease the projected rate of global urbanization. On the other hand, successful development of alternative energy sources might enhance urbanization processes and growth rates. Furthermore, mapping physical expansion of urban areas is not sufficient to calculate the full range of effects of urbanization on biodiversity, ecosystem services and human well-being. There are many indirect effects of urbanization due to the resource demands of residential, commercial, and industrial activities in urban areas that need to be considered. Urbanization also transforms consumption patterns, as well as alters how people value biodiversity and the social norms related to its sustainable use. Land-use change models that better reflect the complexity, diversity, and intensity of human influence on land systems and the feedback mechanisms are needed to more completely account for these effects (Cadenasso et al. 2007; Seto et al. 2010) (Chap. 22, see also Sect. 33.4).

33.2.2 Urbanization and Global Trends in Biodiversity

Urbanization impacts biodiversity both directly and indirectly. Direct impacts primarily consist of habitat loss and transformation through physical expansion of urban areas. Indirect impacts include changes in water and nutrients, increased colonization by introduced species as urban areas expand (Pickett and Cadenasso 2009) and the auxiliary effects of land use and human behaviors within urban space (Clergeau et al. 2006; Szlavecz et al. 2011; Lepczyk and Warren 2012).

The most obvious direct impact of urbanization on biodiversity is the land cover change due to the growth of urban areas. Although urban areas cover less than 3 % of the Earth's surface, the location and spatial pattern of urban areas have a significant impact on biodiversity:

1. Cities have historically been concentrated along coastlines and some islands as well as on major river systems, which also are areas of high species richness and endemism. The spatial correlation between urban growth and endemism means that urban growth already has significantly impacted biodiversity (Chaps. 2, 3, 9, 10 and 23).
2. Urban expansion is now occurring faster in low-elevation, biodiversity-rich coastal zones than in other areas (Chap. 22). While the majority of terrestrial ecoregions are currently less than 1 % urbanized, about 10 % of terrestrial vertebrates are in ecoregions along coasts and on islands that are heavily impacted by urbanization (Chaps. 3 and 22). More than 25 % of all endangered or critically endangered species will be affected to varying degrees from urban expansion by 2030. This will be most pronounced in coastal areas and islands where endemism tends to be particularly high (Chap. 22).
3. The urban land cover in biodiversity hotspots around the world is expected to increase by more than 200 % between 2000 and 2030 with substantial variations

in the rate and amount of increase across individual hotspots. The hotspots in South and Central America as well as in Southeast Asia will experience both high rates and high amounts of urban expansion by 2030 (Chaps. 22, 4 and 28).

4. Urban population expansion is also significant in tropical dry and moist forests, deserts and tropical grasslands. However, the largest increases in terms of urban population per habitat area will be in mangrove habitats, flooded grasslands, and temperate broadleaf forests (Chap. 22).
5. Urban expansion will significantly impact freshwater biodiversity on a global scale. Freshwater biodiversity impacts will be largest in places with large urban water demands relative to water availability as well as high freshwater endemism. Of particular conservation concern is the Western Ghats of India, which will have 81 million people with insufficient access to water by 2050, but is also a region with 293 fish species, 29 % of which are endemic to the ecoregion (Chap. 22).
6. More than 25 % of the world's terrestrial protected areas are within 50 km of a city (Chap. 3). This close proximity will have multiple effects on these protected areas, and signifies a need for urban residents and local governments to explore how to co-exist with protected areas. However, close proximity between urban populations and protected areas can have positive outcomes, such as increased potential for recreational activities and nature-based education. Urban land expansion is likely to take place near protected land at approximately the same rate as elsewhere (Chap. 22). Being near a protected area does not necessarily slow, and can in fact accelerate, the rate of urban land conversion. Taken together, these results imply that due to impacts of continuing urbanization, alteration of ecosystem function of protected areas is taking place in most of the world's urban regions (Laband et al. 2012). Establishing management practices such as biodiversity corridors in regions with high likelihood of becoming urban is desirable, but will require coordinated efforts among administrative bodies within and among nations. Such corridors may take on additional significance considering the migration of species in response to shifts in their ranges with climate change (Forman 2008) (Chap. 25).

33.2.3 Biodiversity in Urban Areas

Since cities represent a complex, interlinked system shaped by the dynamic interactions between ecological and social systems, preserving and managing urban biodiversity means going well beyond the traditional conservation approaches of protecting and restoring what are often considered "natural ecosystems." Indeed, there is an imperative to infuse or mimic such elements in the design of urban spaces. Although the basic ecological patterns and processes (e.g., predation, decomposition) are the same in cities and more natural areas, urban ecosystems possess features that distinguish them from other non-urban ecosystems (Niemelä 1999) (Chaps. 10 and 11). Such ecological features include the extreme patchiness of urban ecosystems, prevalence of introduced species, and the high degree of disturbances in urban settings. Which species occur in any given urban area depends upon two factors: the extent to which the urban habitats support native species (i.e., the strength of the urban biotic filter) (Williams et al. 2009), and the introduction of non-native species. Introduction of non-native invasive species may lead not only to the loss of "sensitive" species dependent on larger, more natural blocks of habitat but also to the establishment of "cosmopolitan" species, i.e., generalists that are present in most cities around the world (Chaps. 10 and 28). The net result is sometimes called "biotic homogenization". The flora and fauna of the world's cities indeed become more similar and homogeneous over time but there is evidence that the proportion of native species remains high in spite of this (Pickett et al. 2011). A recent global analysis of urban plant and bird diversity found that urban areas filter out or exclude about a third of the native species of their surrounding region on average (Chap. 10). While this loss of diversity is worrying, it is worth noting that two-thirds of the native plant and bird species continue to occur in urban areas that are not designed with biodiversity protection in mind (although their population sizes and distribution ranges may be impacted by urbanization). In some cases, urban areas may host cultural and biodiversity-rich green spaces that serve as remnants of biodiversity of the broader landscape and region, especially if the surrounding landscapes have been simplified through agriculture or forestry (Barthel et al. 2005). Novel plant and animal communities are continuously assembled in urban areas, often with active manipulation and management by human society. These communities can play an important role in the generation and maintenance of ecosystem services within the urban area, as well as for surrounding habitats. Biodiversity-conscious urban design therefore has the potential to support a larger proportion of functional biodiversity within urban landscapes, as well as to maintain the density, structure and distribution of the plant and animal communities (Pickett et al. 2013b).

33.2.4 Urbanization and Ecosystem Services

Urban areas affect many ecosystem services on scales ranging from local to global (Chap. 11). One of the most critical services on a regional to global scale is the provision of freshwater (for details on local urban ecosystem services, see Chap. 11). Urban areas depend on freshwater availability for residential, industrial, and commercial purposes; yet, they also affect the quality and amount of freshwater available to them. Water availability is likely to be a serious problem in most cities in semiarid and arid climates. More than a fifth of urban dwellers, some 523 million, live in climates that would at least be classified as semiarid. Moreover, currently 150 million people live in cities with perennial water shortage, defined as having less than 100 L/person/day of sustainable surface and groundwater flow within their urban extent. By 2050, population growth will increase this number to almost a billion people. Furthermore, climate change is projected to cause water shortage for an additional 100 million urbanites (Chap. 3). Globally, urban areas and the resources consumed by urban inhabitants, are responsible for somewhere between

40 and 71 % of all anthropogenic greenhouse gas (GHG) emissions. The majority of global urban GHG emissions are from cities in the developed world. Within cities, energy service production accounts for the largest share of GHG emissions. A large percentage of GHG emissions are those from the largest cities (mega-cities). While there are smaller urban areas that have high per-capita emissions, these centers account for a much lower share of total emissions (Chap. 25).

General trends in the provisioning of more local urban ecosystem services are difficult to assess, but with current types of urban development they are likely to decrease on most continents (Chaps. 21 and 22). The picture is, however, complex; while in many places in Europe a tendency to move to more compact city development may reduce the area of green space, in other places shrinking cities free up space for establishment of new green areas (Chap. 12). Also, ecological restoration of old industrial areas and brown fields, and investment in green infrastructure is on the rise in both Europe and North America (Chap. 31).

While global-scale analyses and projections of the effects of urbanization on biodiversity and ecosystem services are valuable for exposing broad trends, studies at the regional and country scales may allow for additional depth and insight about more local processes. However, such regional and country-level studies are sparse. In a study of 25 EU countries (Chap. 22), average biodiversity appears to decline in almost all countries and all future development scenarios, with exceptions for Germany, Latvia, Estonia, and Malta. Most of the decline is due to urbanization, increase in nitrogen deposition, and disturbance in densely populated areas. Projected urbanization in Britain from 2006 to 2016 and effects on ecosystem services such as freshwater flood mitigation services, carbon storage, and agricultural production suggest that the way ecosystem services will be impacted depends largely upon the patterns of urbanization. There are complex trade-offs between densification and sprawl scenarios. Under the densification scenario, much less land becomes urban which limits the impacts on carbon storage and agricultural production. However, at the same time, more people would be affected by fluvial flooding. Collectively, the findings of these studies suggest the need for policies that consider urban growth at local as well as regional and global scales.

33.3 Gaps in Knowledge

In this section we will address some of the important knowledge gaps that have emerged from the analyses in various chapters of this book.

33.3.1 Gaps in Our Modeling of Global Urban Dynamics

Even though research has advanced considerably in the field of modeling global dynamics (Harris 1985; Wegener 1994; Wilson 1998; Agarwal et al. 2002; Batty et al. 2004), the traditional modeling of urban dynamics is still faced with significant challenges (Chaps. 21 and 22). We still fail to fully grasp the trajectories of urban systems, the planetary scale of impacts of urbanization, social-ecological feedbacks, and the ways through which changes in global environment will affect the urbanization process itself. General land-use change models began in the 1950s and were concerned primarily with local areas or regions, and the majority of the research conducted in this field remains a narrowly focused activity within specific urban regions. It was not until 2011 that the first global models of urban land-use change emerged, providing a fuller picture of global urbanization patterns (Chap. 21). There are still today very few models that adequately capture the coupled dynamics of social-ecological systems, and address important feedback loops and non-linearities in the systems.

A key feature to address in the integration of human and natural systems arises from the fact that we are faced with feedback loops between cities and the global environment (Sánchez-Rodríguez et al. 2005). These feedback loops occur in a parallel, simultaneous fashion: while urban systems and processes have an effect on local environments on a massive scale (leading to global environmental change), changes in the global environment (through a variety of natural cycles) affect urban areas differentially. Furthermore, responses (within and around urban areas) to global environmental changes eventually impact urbanization processes themselves. Models that only partially address components of this wider feedback loop are destined to capture an incomplete picture of a coupled natural-human system and will be limited in their capacity to project urban futures.

For example, while many models today address the impacts of urbanization (in terms of size, form and function) on climate change and biodiversity loss, very few models (if any) attempt to close the feedback loop by addressing the effects of changes in ecological systems on the urban system. Our most heavily utilized urban dynamics models are not currently designed to address fundamental questions regarding how humans can and will adjust their behavior through expectations about future risks and impacts related to climate change and biodiversity loss (e.g., Tidball and Stedman 2013). This lack of understanding adversely affects our capacity to realistically project change in urban systems; the vast majority of existing models ignore this dimension of the urban-environment feedback loop, thus assuming that urban populations do not respond with actions to new information about expected or actual impacts of environmental change. Typically, urban growth models will make projections about future urban population growth, physical expansion or GDP growth without paying attention to information about unintended costs, and risks and uncertainties that arise from projected environmental change. This is a significant paradox, especially in cases where a model employed is an integrated model, focusing on some of the connections between social and ecological systems. Unless we develop integrated models that address multiple scales of interactions and responses, non-linear trajectories, thresholds, the importance of economic agency, and the role of incentives and prices (among other factors important for system dynamics), our capacity to fully explain and realistically project how climate change and biodiversity loss will eventually affect urban growth, form, and function globally will remain extremely limited. The lack of such capacity is primarily due to conceptual and methodological challenges involved in creating integrated models of social and ecological systems (Holling 1993) but it is not insurmountable. A new generation of dynamic models, emerging from promising studies that offer ways to overcome the challenge of full integration (Alberti and Waddell 2000; Güneralp and Seto 2008; Haase and Schwarz 2009; Wilson 2010; Zhang et al. 2010), lies at the forefront of research on urbanization and biodiversity.

33.3.2 Knowledge Gaps Related to Biodiversity and Ecosystem Services

As indicated by the chapters in this book, there is no scarcity of research questions related to urbanization and its relationship to biodiversity and ecosystem services. Alongside challenges of understanding and forecasting patterns of land-use change and urbanization, there are also gaps in knowledge regarding many aspects of biodiversity and ecosystem services, such as connections between various ecosystem processes across spatial and temporal scales (Colding 2007; Niemelä 2013). The interactions between urban and rural regions (Larondelle and Haase 2013) and feedback mechanisms among ecosystem processes within and near cities are still poorly understood, as is the impact of urbanization on values, norms and institutions related to the consumption and/or sustainable use of biodiversity and ecosystem services. Furthermore, a major driver of change that likely will affect urban biodiversity and ecosystem services is climate change (Chap. 25). The broad questions integrating natural and social sciences in studying the effects of climate change on urban ecosystems, and the way these changes impact people's well-being, were identified by James et al. (2009) as requiring urgent research attention addressed through interdisciplinary collaboration between ecologists, geographers, urban scholars, social scientists, economists, together with urban land-use planners and conservation practitioners (Niemelä et al. 2011).

With respect to ecosystem services, we have identified several specific knowledge gaps:

Supply-demand gap: An increasing body of knowledge exists on the provisioning of ecosystem services at many different scales, but there is little information on needs and demands of ecosystem services in cities. In particular, we know little about the negotiated interactions that lead to trade-offs and synergies in the demand for particular bundles of ecosystem services accessed by different socio-economic or livelihood groups in urban environments (but see Colding et al. 2006; Andersson et al. 2007). This will play a major role in impacting outcomes of equity, particularly for the urban poor as well as for traditional livelihood users, such as fishers and livestock grazers in peri-urban areas (D'Souza and Nagendra 2011). When focusing on demands placed upon ecosystem services, we are in need of interdisciplinary approaches (see James et al. 2009; Niemelä et al. 2011; Kabisch and Haase 2012).

Geographical gap: There is a geographical gap in knowledge—most scientific studies of ecosystem services in cities are carried out and published in Europe, North

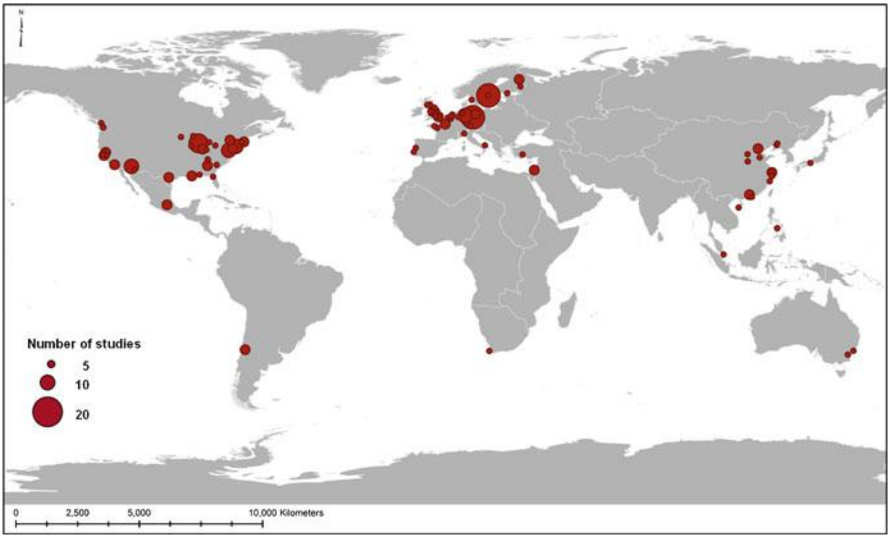

Fig. 33.1 The distribution of 217 urban ecosystem services case studies appearing in peer-reviewed literature during the period 2000–2012 (Reproduced from Haase et al. 2014, submitted. Published with kind permission of © Dagmar Haase 2014. All Rights Reserved)

America and China (Fig. 33.1, Haase et al. 2014) (Chap. 27). Thus, judging from what is available in the peer reviewed literature, we have a poor understanding of the supply, needs and management of urban ecosystem services in large regions in South Asia, Africa and Latin America, which critically are those areas that are developing most rapidly and face some of the greatest threats to protected areas and biodiversity hotspots in their boundaries (Chaps. 4, 21, 22, 23 and 28). For example, the novel structures and human fluxes associated with urbanization in Africa are especially under-studied (McHale et al. 2013). This gap is also reflected in the local assessments in this book, where Cape Town is the only African city represented (Chap. 24). However, this does not necessarily mean that local knowledge is non-existent. Most likely there is much ecological knowledge at the local level being used everyday in more informal management of urban ecosystems. Indeed, this is known to be the case in many places in Asia and Africa. For instance, comparisons of residential gardens in different continents indicate that most plant species in home gardens in Europe and North America are chosen for their ornamental value, while in contrast a large proportion of species in gardens in India and South Africa are chosen for their medicinal, food or cultural properties (Jaganmohan et al. 2012) (Chap. 7). Local knowledge and practices could be mobilized in multiple ways through,for example, citizen science initiatives (Chaps. 18, 29, 30 and 32), and thus could support more formal governance and management of urban ecosystem services.

Valuation gap: Many tools for monetary valuation of ecosystem services are already available, but these need to be complemented with non-monetary valuation methods and with planning tools based on multiple criteria (Chap. 11) (Gomez-Baggethun and Barton 2013). The total value of multiple services generated by ecosystems can be divided into different parts, depending on whether or not there is a market for the service and whether the value can be expressed in monetary or only in non-monetary terms. Ecosystem service science still lacks a robust theoretical framework that allows for consideration of social and cultural values of urban ecosystems on an equal basis with monetary values in decision-making processes. Developing such a framework involves synthesizing the large but scattered body of literature that has dealt with non-monetary values of the environment, and articulating this research into ecosystem service concepts, methods, and classifications. (Chan et al. 2012; Luck et al. 2012)

Insurance value gap: We are in particular need of new valuation techniques that utilize a resilience and inclusive wealth perspective to better capture the value of biodiversity and ecosystems in reducing urban vulnerability to shocks and disturbances (Sect. 33.4). The insurance value of an ecosystem is closely related to its resilience and self-organizing capacity, and to what extent it may continue to provide flows of ecosystem service benefits with stability over a range of variable environmental conditions. The economic approaches to insurance values are still poorly developed (Pascual et al. 2010).

Cultural value gap: While much attention has been focused on provisioning and regulating ecosystem services provided by urban ecosystems, cultural services have been poorly researched (e.g., Daniel et al. 2012). While such services may not be apparent in a global synthesis, they can play an extremely important role in place-based conceptualizations of urban ecosystem services, for instance, on continents like Asia and Africa, where many sacred conceptualizations of nature persist in cities (e.g., protection of sacred keystone species such as *Ficus religiosa* across cities in India (Chaps. 6 and 7)). There are numerous equity and environmental justice issues related to cultural ecosystem services, but these are often poorly documented (Alfsen et al. 2010; D'Souza and Nagendra 2011). Also lacking is careful articulation and analysis of urban land ethics (Boone et al. 2013) that might link with evolving conservation and cultural landscape ethics (Rozzi 2012).

Overall, a research agenda covering the above issues could assist in pinpointing where more understanding is needed for supporting urban sustainability and resilience. An ecosystem services and social-ecological approach to urban sustainability and resilience, could form a central and unifying approach (Niemelä et al. 2011), and will be discussed in the last section of this chapter.

33.3.3 Knowledge Gaps Related to Governance

Despite its relative youth, the field of urban biodiversity and ecosystem services research has generated a good deal of peer-reviewed material on issues that are explicitly linked to questions of urban governance (Chap. 27). However, knowledge gaps on governance issues are large and it is particularly challenging for a global analysis that the scientific literature on urban ecosystem governance is biased towards some parts of the world and largely missing from Africa, Latin America and parts of Asia (reflected in Fig. 33.1) (Chaps. 23 and 28). Furthermore, ecosystem *services* governance is extremely complex since the environmental agenda of cities is intertwined with a number of issues and competing priorities, as well as multiple temporal and spatial scales of ecosystem processes and their relation to multiple influencing and impacted actors (Sendstad 2012). Several studies indicate that public institutions have a lack of cooperation across departments or levels of authority, and have an inadequate capacity to handle diverse information and deal with change to respond to environmental problems (Alfsen-Norodom et al. 2004; Blaine et al. 2006; Andersson et al. 2007; Ernstson et al. 2010b). Strategies and regulations tend typically to focus only on a few ecosystem services at the local scale (Ernstson et al. 2010b), assume stability in their supply (Asikainen and Jokinen 2009), and show a lack of provisions connecting urban consumers of ecosystem services and the people managing the services that the urban consumers depend on, originating from outside the city boundaries (Blaine et al. 2006; Puppim de Oliveira et al. 2011). There may often be a mismatch between scales of ecosystem processes on the one hand, and scales of management on the other (Borgström et al. 2006). Multi-level governance can be critical to address issues of sustainable urban ecosystem management, taking into consideration the entire range of spatial and temporal scales that impact resilience, but the importance of such approaches has been insufficiently recognized.

Although studies are emerging that describe how institutions can be formed to connect stakeholders managing, impacting and depending on certain ecosystems and their services (e.g., Colding et al. 2006; Ernstson et al. 2010a, b; Barthel et al. 2010), such studies are scarce and there are few measures that have been implemented or tested. Similarly, there are several studies recognizing cities as having a global impact on ecosystem services provision (e.g., Hagerman 2007; Hutton 2011), but few have investigated policy mechanisms connecting multiple cities and ecosystems at the global level (Sendstad 2012; but see Folke et al. 1997; Wackernagel et al. 2006).

33.4 Local Action and Policy for Urban Biodiversity and Ecosystem Services

In the first publication of the Cities and Biodiversity Outlook (CBO) project, the *Cities and Biodiversity Outlook – Action and Policy* (CBO A&P, see Preface), the main message was that urbanization and biodiversity challenges will require improved governance responses across multiple scales. Particularly at the city level, a lack of financial and human resources, as well as technical capacity, can prevent issues on biodiversity and ecosystem services from being recognized or addressed. This was also illustrated by a range of examples from cities around the world, and has been further discussed in the more extensive local and regional assessments in this book. The CBO A&P is organized around ten Key Messages, of which number one sets the framework of challenges and opportunities, and the remaining nine explore the opportunities inherent in urbanization:

1. Urbanization is both a challenge and an opportunity to manage ecosystem services globally.
2. Rich biodiversity can exist in cities.
3. Biodiversity and ecosystem services are critical natural capital.
4. Maintaining functioning urban ecosystems can significantly enhance human health and well-being.
5. Urban ecosystem services and biodiversity can help contribute to climate-change mitigation and adaptation.
6. Increasing the biodiversity of urban food systems can enhance food and nutrition security.
7. Ecosystem services must be integrated in urban policy and planning.
8. Successful management of biodiversity and ecosystem services must be based on multi-scale, multi-sectoral, and multi-stakeholder involvement.
9. Cities offer unique opportunities for learning and education about a resilient and sustainable future.
10. Cities have a large potential to generate innovations and governance tools and therefore can—and must—take the lead in sustainable development.

The implementation of many of these key messages will depend on governance efforts characterized by collaboration of multiple jurisdictions as well as involvement of stakeholders to address the multiple drivers of biodiversity loss. Some approaches to implementation and successful examples of collaborations are presented in, amongst others, the local assessments of New York, Bangalore, Cape Town and Stockholm (Chaps. 19, 7, 24 and 17). Research on planning emphasizes the importance of assessment and valuation of a broad spectrum of urban ecosystem services. However, while such evaluations are useful for measuring progress towards sustainability, they rarely motivate or support the innovations required to provide ecosystem services as an intentional part of urban planning (Ahern et al. 2013). As the local assessments of Shanghai and Istanbul (Chaps. 9 and 16) highlight, there is a dichotomy between (a) knowledge on the importance of services provided by urban ecosystems for the cities (and assessments as tools for safeguarding the ecosystems), and (b) the actual urban development trajectory with its associated impacts on the ecosystems. In this context, urbanization and development of new urban infrastructure represent a unique opportunity for "learning-by-doing." Although advances in urban sustainability have been made through transdisciplinary collaborations among researchers, professionals, decision-makers and other stakeholders, these advances have limited transferability due to the uniqueness of the city in which they originated. The promise of practicing "learning-by-doing," therefore, is not yet fully integrated with urban development. Ahern et al. (2013) propose a model for "safe to fail" adaptive urban design to provide a framework to integrate science, professional practice, and stakeholder participation. The framework includes experimental design guidelines, and monitoring and assessment protocols for realizing urban ecosystem services integral with urban development.

Cooperation is important in order to synchronize and to harmonize actions "vertically" (i.e., at international, national, sub-national, and local levels) and "horizontally" (e.g., across divisions such as environment, planning, transportation, education, finance and nutrition). As the broad scope of local assessments in this book shows, there is significant diversity in the way local governments in different countries can approach vertical and horizontal governance of biodiversity and ecosystem services. The local assessment of Istanbul (Chap. 16) shows how tools for policy-makers and planners to assess and value urban ecosystems hold the potential to increase focus on ecosystems in urban planning. The local assessment of Chicago (Chap. 18) presents the initiative of *Chicago Wilderness*, an organization in which researchers, policy-makers and the public alike participate in restoring and conserving nature in the region. The local assessment of Stockholm (Chap. 17) highlights parallel but largely separate management practices in support of urban ecosystems in the city, such as conservation of the city's green wedges in municipal planning, and the active maintenance of allotment gardens by private initiatives. Federally-managed governments such as that in the UK decentralize many of the mandates on biodiversity governance to their national and sub-national authorities, and these in turn commission much of the implementation at lower government levels. This is also the case in Germany and Canada. Other nations, such as Japan, South Africa, Switzerland, Mexico, and Brazil, provide guidelines for biodiversity governance and encourage their sub-national and local governments to develop strategies and action plans in line with their national ones. For example, the assessment of *satoyama* and *satoumi* landscapes (Chap. 8) discusses how pressure is increasing on local urban policies to actively support adaptation of traditional management systems and landscapes to the urban environment, as the rapid influx of people to urban areas causes ecosystem productivity in the surrounding rural areas to decrease.

As many of the solutions to global concerns such as biodiversity emerge at the local level, we need local and global efforts to create the capacity to innovate locally and diffuse those innovations globally. As the local assessments of Bangalore and Chicago (Chaps. 7 and 18) highlight, local groups have to be able to adopt the best solutions for their local needs, absorb new practices, and create the institutional mechanisms to support these efforts. New governance structures for land management of biodiversity have emerged that do not rely solely on traditional market and government interventions, but on other institutional arrangements. Local citizens often make these arrangements themselves, which involve private, common, and public land to protect ecosystem services that cannot always be assessed by monetary values. The local assessment of Rio de Janeiro (Chap. 29) found that in the case study area where people were predominantly low-income earners, they had great knowledge of the local biodiversity, and actively managed the local urban greens in their neighborhoods by maintaining native plants that could be used, for example, as food. However, the settlements were often informal, and the people were thus vulnerable to changes in the official planning of the area. These are governance mechanisms that can provide new forms of thinking about spatial planning and interventions from different perspectives. They are particularly useful for understanding the role of different actors. They can also address concerns that local populations may be losing control of their landscape to higher levels of governance. Empowering local people is one step toward finding sustainable solutions to manage resources. However, as important as it is to build the capacity of local communities to define their own challenges and means to implement solutions, it should be recognized that emergent governance such as this may not always direct the community along pathways deemed desirable by the broader community of scholars, planners, and resilient theorists. The key is to incorporate positive normative values into the capacity-building exercises and incorporate safeguards against pathways leading to undesirable states (Wiek et al. 2012).

33.5 A Global Framework for Urbanization, Sustainability, Resilience and Transformation

33.5.1 The Need for a New Framework

In spite of the remarkable progress made in urban ecosystem studies over the last few decades, dynamic interactions and resilience of ecosystem functions in these social-ecological systems are still poorly understood (Andersson 2006; Alberti 2010). Sustainability and resilience in urban systems require a new framework that explicitly addresses the question of scale and the multiple-scale interactions, feedbacks, tradeoffs, and synergies between specific and general resilience (Cumming et al. 2013). The challenge to advancing our understanding of coupled urban dynamics is to integrate diverse scientific approaches and knowledge domains grounded in multiple epistemologies, but engaged with the sustainability challenge. Sustainability science serves as an inspiring arena for such integration.

Sustainability science is a field defined by the problems it addresses rather than by the disciplines it employs; it focuses on improving society's capacity to use the earth in ways that simultaneously meet the needs of a much larger (but stabilizing) human population, sustain the life support systems of the planet, and substantially reduce hunger and poverty (PNAS 2007). Resilience thinking is part of sustainability science, and has two central foci: one is strengthening the current social-ecological system to live with change by enhancing the ability to adapt to potential external pressures in order to retain its essential functions and identity; the other is the ability to shift development pathways from those that are less desirable and/or unsustainable, to ones that are more desirable and/or sustainable—also referred to as transformability (Walker et al. 2004; Folke et al. 2010).

The complexity of urban coupled human-natural systems or social-ecological systems poses enormous challenges in identifying causal mechanisms because of the many confounding variables that exist. At the same time, scientific findings from empirical studies are difficult to generalize due to variation in socio-economic and biophysical contexts, and the great heterogeneity that characterizes urban regions (Grimm et al. 2008). Key challenges are scale mismatches, cross-scale interactions, and limited transferability across scales (Cumming et al. 2013). Furthermore, limited predictability of system behavior over the long term requires a new consideration of uncertainty (Polasky et al. 2011).

Special attention will also need to be given to the translation of the emerging knowledge in urban practice and governance through sustainability and resilience planning. While planning theory thus far has paid surprisingly little attention to human-nature relations (Wilkinson 2012a), planning practitioners see insights from resilience thinking as providing a new language and metaphors for the dynamics of change, and new tools and methods for analysis and synthesis. Furthermore, a resilience approach confronts modes of governance based on assumptions of predictability and controllability (Wilkinson 2012b) with a mode based on dynamics and non-linearity. This is an emerging field in which new, innovative means of planning that deal with urban complexity and sustaining urban ecosystem services are needed. However, resilience thinking and social-ecological theory provide planning with little guidance in prioritizing or addressing tradeoffs between different strategies; this highlights the inherently political character of urban governance (Wilkinson 2012a, b).

What actually constitutes urban sustainability—particularly in relation to various spatial scales—needs rethinking, but so do the concepts of resilience and transformations (Folke et al. 2002; Childers et al. 2013; Westley et al. 2011; Pickett et al. 2013a). In this part of the chapter, we will explore these concepts and also address some misconceptions. In general, both the sustainability and the resilience concepts (particularly general resilience, see below and Table 33.1) are not easily applicable to the city scale. Cities are centers of production and consumption, and urban inhabitants are reliant on resources and ecosystem services—including everything from food, water and construction materials to waste assimilation— secured from locations around the world. Although cities can optimize their resource use, increase their efficiency, and minimize waste, they can never become fully self-sufficient (Grove 2009). Therefore, individual cities cannot be considered “sustainable” without acknowledging and accounting for their teleconnections (Seto et al. 2012)—in other words, the long-distance dependence and impact on ecosystems, resources and populations in other regions around the world (Folke et al. 1997). Sustainability is commonly misunderstood as being equal to self-sufficiency, but in a globalized world, virtually nothing at a local scale is self-sufficient. To become meaningful, urban sustainability therefore has to address appropriate scales, which always will be larger than an individual city. The same logic is also true for the concept of general resilience; a narrow focus on a single city is often counterproductive (and may even be destructive) since building resilience in one city often may erode resilience somewhere else, thus producing multiple negative effects across the globe. Also, while from historical accounts we learn that there are some cities that have gone into precipitous decline or actually failed and disappeared, such as Mayan cities (Tainter 2003), our modern era experience is that contemporary cities are much less likely to collapse and disappear (Chap. 2). Instead, they may enter a

Table 33.1 Definition of concepts

Sustainability	Manage resources in a way that guarantees welfare and promotes equity of current and future generations
Resilience	The capacity of a system to absorb disturbance and reorganize while undergoing change so as to still retain essentially the same function, structure and feedbacks, and therefore identity, i.e., capacity to change in order to maintain the same identity
General resilience	The resilience of a system to all kinds of shocks, including novel ones
Specified resilience	The resilience "of what, to what"; resilience of some particular part of a system, related to a particular control variable, to one or more identified kinds of shocks
Coping strategy	The ability to deal effectively with, e.g., a single disturbance, with the understanding that a crisis is rare and temporary and that the situation will quickly normalize when the disturbance recedes
Adaptive strategy	Adjustment in natural and human systems in response to actual or expected disturbances when frequencies of disturbances tend to increase
Transformative strategy	The capacity to transform the stability landscape itself in order to become a different kind of system, to create a fundamentally new system when ecological, economic, or social structures make the existing system untenable

Modified from information included in Folke et al. (2010) and Tuvendal and Elmqvist (2012)

spiral of decline, becoming less competitive and losing their position in regional, national and even global systems of cities. However, through extensive financial and trading networks, cities have a high capacity to avoid abrupt change and collapse. Applying the resilience concept at the local city scale in a global context is thus not particularly useful. Rather, the utility of the resilience concept may lie in thinking about diverse development pathways or basins of attraction in cities, such as smart growth versus a less dense cityscape with green areas and ecosystem services.

When most people think of urban resilience, it is generally in the context of response to sudden impacts, such as a hazard or disaster recovery (see Alberti et al. 2003; Alberti and Marzluff 2004; Pickett et al. 2004; Vale and Campanella 2005; Cutter et al. 2008; Wallace and Wallace 2008). However, the resilience concept goes far beyond recovery from single disturbances. Resilience is a multidisciplinary concept that explores persistence, recovery, adaptive and transformative capacities of interlinked social and ecological systems and subsystems (Holling 2001; Walker et al. 2004; Brand and Jax 2007; Biggs et al. 2012). A distinction is often made between general resilience and specified resilience (Table 33.1) (Carpenter et al. 2012). General resilience refers to the resilience of a system to all kinds of shocks, including novel ones, whereas specified resilience refers to the resilience "of what, to what"—in other words, resilience of some particular part of a system (related to a particular control variable) to one or more identified kinds of shocks (Walker and Salt 2006; Folke et al. 2010). While sustainable development is inherently normative and positive, this is not necessarily true for the resilience concept (Pickett et al. 2013a). For example, development may lead to traps that are very resilient and difficult to break out of (e.g., Walker et al. 2009). The desirability of specified resilience in particular, depends on careful analysis of resilience "of what, to what" (Carpenter et al. 2001) since many examples can be found of highly resilient systems (e.g., oppressive political systems) locked into an undesirable system configuration or state. It also may refer "to whom" as a recognition of environmental inequity (Boone 2002; Pickett et al. 2011).

One of the basic principles in resilience thinking is that a slow variable may invisibly push a larger system closer and closer to a threshold (beyond which there would be radical change towards a new equilibrium) and that disturbances that previously could have been absorbed now result in abrupt change (e.g., Gunderson and Holling 2002). Urbanization may be viewed as a slow variable, which through, for example, changing land cover, pollution and nutrient depositions, may increase vulnerabilities to disturbances. At the same time, urbanization itself may lead to higher intensity/frequency of disturbances through impacts on both global and regional climate change. Urbanization therefore represents a complex interaction between slow and fast variables, which need to be addressed in order to understand how different urban responses link to resilience. Conventional urban responses to disturbances (such as coping and adaptive strategies) may not only, over time, be insufficient at the city-scale, they may also be counterproductive when it comes to maintaining resilience at the global scale.

The concept of coping with disturbance is here used to describe the ability to deal effectively with, for example, a single disturbance, with the understanding that a crisis is rare and temporary and that the situation will quickly normalize when the disturbance recedes (see also Fabricius et al. 2007) (Table 33.1). Adapting to change is defined here as an adjustment in natural and human systems in response to actual or expected disturbances when frequencies of disturbances tend to increase (e.g., Parry et al. 2007) (Table 33.1). In contrast, a transformation is defined as a response to disturbance that differs from both coping and adaptation strategies in that the decisions made and actions taken change the identity of the system itself (Table 33.1). Folke et al. (2010) defined transformability as the capacity to become a different kind of system, to create a fundamentally new system when ecological, economic, or social structures make the existing system untenable. It is important to consider disturbance as a part of a social-ecological system, having temporal and spatial dimensions (Peters et al. 2011), and note that changing social, climatic, and connective relationships may shift disturbance regimes.

It is important to note that transformations of urban contexts or urban sustainability transitions are not only triggered by disturbances, but may also be stimulated by innovative responses to challenges that progressively build up systems' transformative capacity towards a new configuration of drastically altered structures (i.e., infrastructures), cultures (i.e., institutions) and practices (i.e., routines) (Frantzeskaki et al. 2012; Nevens et al. 2013). Although at a first glance transformations often seem counterintuitive for building resilience, multiple transformations on lower scales may be necessary to maintain resilience on a larger scale (Allen and Hoekstra 1992; Wu and Loucks 1995). Implementation of transformation strategies for cities is therefore needed due to a number of reasons: (a) it is recognized that existing coping and adaptation strategies do not suffice and the suggested changes are perceived as highly undesirable (Tuvendal and Elmqvist 2012); (b) mitigation and adaptation strategies remain disconnected from each other and do not exploit synergies that may in return foster resilience (Jäger et al. 2012); and (c) current adaptation strategies do not consider emerging innovations and self-organized networks and initiatives experimenting with urban sustainability that can be the multipliers for transformative and innovative capacity of the cities (Van Eijndhoven et al. 2013; Maassen 2012). These sustainability experimentation spaces can be the examples to draw from and to scale up for achieving urban environmental stewardship via, for example, total re-design of resource production, supply and consumption chains through to stewardship (cf. Chapin et al. 2009) of ecosystem services within and outside city boundaries (Elmqvist et al. 2013).

33.5.2 Sources of Urban Resilience

In light of the aforementioned increased frequency and intensity of hazards and disasters as a result of climate change, and of the proposition of stewardship of ecosystem services in such contexts, it is notable that research focused particularly on hazards has generally settled on four themes in resilience: (1) resilience as a biophysical attribute, (2) resilience as a social attribute, (3) resilience as a social-ecological system attribute, and (4) resilience as an attribute of specific areas or places. Thus, scholars have begun to consider groupings such as these resilience themes, and to search for common linkages and mechanisms that may serve as sources of resilience in specific hazards contexts (Adger et al. 2005; Tidball 2012; Pickett et al. 2013b).

Examples of common linkages and mechanisms that may serve as sources of resilience, and that hit upon unique and novel combinations of biophysical, social, social-ecological, and area or place resilience include community-based natural resource management (greening) in urban landscapes that emerge in hazard and vulnerability contexts. Such “greening in the red zone” (Tidball and Krasny 2013) is defined as an active and integrated approach to the appreciation, stewardship and management of living elements of social-ecological systems. Greening can take place in cities, towns, townships and informal settlements in urban and peri-urban areas. Greening sites vary from small woodlands, public and private urban parks and gardens, urban natural areas, street tree and city square plantings, botanical gardens and cemeteries, to watersheds, whole forests and national or international parks. The contribution of neglected sites to greening should not be dismissed (Pickett 2010). Greening involves active participation of human or civil society in activities in ecosystems (Tidball and Krasny 2007), and can thus be distinguished from notions of “nature contact” (Ulrich 1993) that imply spending time in or viewing nature, but not necessarily active stewardship. Explorations of how greening embodies active community member participation in stewardship of ecosystems and the services provided by them (and which, in turn, may result in measurable benefits for individuals, their community, and the environment) (Svendsen 2013), may represent a kind of management "sweet spot" wherein multiple outcomes and benefits are derived.

A key assumption when considering urban social-ecological systems and hazards (and the potential of ecosystem stewardship within these contexts) is that while hazards are "natural", disasters are not (Bankoff 2010). There is a need to more fully understand the ways in which human systems, especially urban systems and a growing global system of cities, place people at risk in relation to each other and to their environment. There is also a need to continue to explore the interactions between humans and the rest of nature in the context of hazards, particularly how these interactions relate to multiple themes or kinds of resilience and urban sustainability.

33.5.3 Conclusions

Based on this overview, we argue that urban sustainability and resilience thinking, and policies derived from this thinking, must, to a much greater extent, address scales and consider urban teleconnections (Seto et al. 2012), i.e., urban dependence and impacts on distant populations and ecosystems. There is an apparent danger of applying too narrow an urban scale for these types of policies, since, for example, building (desired) resilience in one city may likely lead to erosion of resilience or create undesired resilience elsewhere. To build resilience, urban regions must take increased responsibility for motivating and implementing solutions that take into account their profound connections with, and impacts on, the rest of the planet. Collaboration across a global system of cities could and should provide a new component of a framework to manage resource chains for sustainability through resilience.

If we view sustainable development in a more dynamic way, we can define it as a form of development that fosters adaptive and transformative capabilities, and creates opportunities to maintain equitable, long-term prosperity and well-being in complex and interlinked social, economic, and ecological systems. However, with this definition it could be argued that there is a substantial overlap with the definition of resilience. One suggestion given to resolve this issue is that resilience can be seen as a necessary approach (non-normative process) to meet the challenges of sustainable development (normative goal) (Chelleri and Olazabal 2012; Pelling and Manuel-Navarrete 2011; Biggs et al. 2010; Childers et al. 2013).

Without such considerations, urban resilience may fail to find meaning in rapidly urbanizing areas, or worse: it may create oversimplified goals for building resilience in a too narrow sense and risking being counterproductive. Key contributions from urban research will include a greater understanding of what constitutes generic adaptive and transformative capacity, and finally, how governance might trigger and direct urban transformations. These are far from easy tasks that lie ahead, but as the scale of the global challenge associated with rapid urbanization and climate change grows, traditional conceptualizations of sustainability need to be extended through engagement with resilience.

33.6 Final Remark

Local Agenda 21 (LA21), launched in 1992 at the Earth Summit in Rio de Janeiro, attempted to assist local authorities in tackling many of the global sustainability challenges typically considered beyond their control. LA21 emphasized mainstreaming participatory processes in which local stakeholders set their own priorities while at the same time more effectively engaging higher levels of governments. Twenty years after the start of LA21, there is a perceptible tension between process and results—with an often stalled process at the national level while tangible results are being achieved at the local. In response to this tension, and to the mounting challenges that cities are beginning to face, initiatives by local municipalities to work together in global networks and in partnerships with the private sector are emerging and growing. Examples of this can be found across the world, including amongst others the Urban Biosphere Initiative (URBIS) (Alfsen et al. 2010); ICLEI—Local Governments for Sustainability; IUCN (International Union for the Conservation of Nature); and C40 Cities—Climate Leadership Group.

As centers of human innovation, and perhaps the most active frontier of our impact on the planet in shaping its landscapes and seascapes, cities offer arenas for enormous opportunities to reimagine and invent a different kind of future with room for humans and other species to thrive. Cities may well be the ground where we secure a globally sustainable future—one that builds on nature-based solutions and ecosystem-based adaptation, and establishes responsible environmental stewardship at the heart of public interest.

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CBO Inter-Agency Task-Force

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Glossary[1]

Adaptation Adjustment in natural or human systems to a new or changing environment. Various types of adaptation can be distinguished, including anticipatory and reactive adaptation, private and public adaptation, and autonomous and planned adaptation.

Adaptive capacity The general ability of institutions, systems, and individuals to adjust to potential damage, to take advantage of opportunities, or to cope with the consequences.

Adaptive management A systematic process for continually improving management policies and practices by learning from the outcomes of previously employed policies and practices. In active adaptive management, management is treated as a deliberate experiment for purposes of learning.

Adaptive strategy Adjustment in natural and human systems in response to actual or expected disturbances when frequencies tend to increase.

Anthropocentric perspectives Viewing humans as the most important entities.

Anthropocene An informal geologic chronological term that serves to mark the evidence and extent of human activities that have had a significant global impact on the Earth's ecosystems.

Anthropogenic impacts Impacts resulting from human activities.

Appropriation The process of capturing some or all of the demonstrated and measured values of ecosystem services so as to provide incentives for their sustainable provision.

Benefits Positive change in wellbeing from the fulfilment of needs and wants.

Biodiversity (a contraction of **biological diversity)** The variability among living organisms from all sources, including terrestrial, marine, and other aquatic ecosystems and the ecological complexes of which they are part. Biodiversity includes diversity within species, between species, and between ecosystems.

[1] Based on definitions given in

MA. (2005). *Ecosystems and human well being: Current state and trends* (Vol. 1). Washington, DC: Island Press.

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Biodiversity may be described quantitatively, in terms such as richness, rarity, and uniqueness.

Biodiversity hotspot Refers specifically to 25 biologically rich areas around the world that have lost at least 70 % of their original habitat.

Biological diversity see Biodiversity

Biome The largest unit of ecological classification that is convenient to recognize below the entire globe. Terrestrial biomes are typically based on dominant vegetation structure (e.g., forest, grassland). Ecosystems within a biome function in a broadly similar way, although they may have very different species composition. For example, all forests share certain properties regarding nutrient cycling, disturbance, and biomass that are different from the properties of grasslands.

Biotope An ecological area that supports a particular range of biological communities.

Carbon sequestration The process of increasing the carbon content of a reservoir other than the atmosphere.

City see Urban area

Coping strategy The ability to deal effectively with, e.g., a single disturbance, with the understanding that a crisis is rare and temporary and that the situation will quickly normalize when the disturbance recedes.

Cost-benefit analysis A technique designed to determine the feasibility of a project or plan by quantifying its costs and benefits.

Cultural ecosystem services The nonmaterial benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experience, including, e.g., knowledge systems, social relations, and aesthetic values.

Direct driver A driver that unequivocally influences ecosystem processes and can therefore be identified and measured to differing degrees of accuracy.

Direct use value (of ecosystems) The benefits derived from the services provided by an ecosystem that are used directly by an economic agent. These include consumptive uses (e.g., harvesting goods) and non-consumptive uses (e.g., enjoyment of scenic beauty). Agents are often physically present in an ecosystem to receive direct use value.

Disservices Undesired negative effects resulting for the generation of ecosystem services.

Driver Any natural or human-induced factor that directly or indirectly causes a change in an ecosystem.

Ecological footprint An index of the area of productive land and aquatic ecosystems required to produce the resources used and to assimilate the wastes produced by a defined population at a specified material standard of living, wherever on Earth that land may be located.

Ecological infrastructure Any area that delivers services (such as fresh water, microclimate regulation, recreation, etc.) to a large proximate population, usually in cities. This is sometimes referred to as green infrastructure.

Ecological threshold The point at which the conditions of an ecosystem result in change to a new state.

Ecological value Non-monetary assessment of ecosystem integrity, health, or resilience, all of which are important indicators to determine critical thresholds and minimum requirements for ecosystem service provision.

Economic growth An increase in economic prosperity measured, for example, as an increase in per capita gross domestic product (GDP).

Eco-regional planning Planning that is undertaken on an eco-regional rather than national basis.

Ecosystem A dynamic complex of plant, animal, and microorganism communities and their non-living environment interacting as a functional unit. For practical purposes it is important to define the spatial dimensions of concern.

Ecosystem degradation A persistent reduction in the capacity to provide ecosystem services.

Ecosystem function A subset of the interactions between ecosystem structure and processes that underpin the capacity of an ecosystem to provide goods and services.

Ecosystem management An approach to maintaining or restoring the composition, structure, function, and delivery of services of natural and modified ecosystems for the goal of achieving sustainability. It is based on an adaptive, collaboratively developed vision of desired future conditions that integrates ecological, socioeconomic, and institutional perspectives, applied within a geographic framework, and defined primarily by natural ecological boundaries.

Ecosystem process Any change or reaction, which occurs within ecosystems, either physical, chemical or biological. Ecosystem processes include decomposition, production, nutrient cycling, and fluxes of nutrients and energy.

Ecosystem services The direct and indirect contributions of ecosystems to human wellbeing. The concept “ecosystem goods and services” is here used as synonymous with ecosystem services.

Ecosystem structure The biophysical architecture of an ecosystem. The composition of species comprising the architecture may vary.

Equity Fairness of rights, distribution, and access. Depending on context, this can refer to resources, services, or power.

Existence value The value that individuals place on knowing that a resource exists, even if they never use that resource (also sometimes known as conservation value or passive use value).

Externality A consequence of an action that affects someone other than the agent undertaking that action and for which the agent is neither compensated nor penalized through the markets. Externalities can be positive or negative.

Extinction The point at which a organisms within a species can no longer reproduce to create subsequent generations and the species dies out.

Functional diversity Value, range and abundance of functional traits of organisms in a given ecosystem.

Functional groups Groups of organisms that respond to the environment or affect ecosystem processes in a similar way. Examples of plant functional types include nitrogen-fixer versus non-fixer, stress-tolerant versus ruderal versus competitor, resprouter versus seeder, deciduous versus evergreen. Examples of animal functional types include granivorous versus fleshy-fruit eater, nocturnal versus diurnal predator, browser versus grazer.

Functional redundancy A characteristic of ecosystems in which more than one species in the system can carry out a particular process. Redundancy may be total or partial—that is, a species may not be able to completely replace the other species or it may compensate only some of the processes in which the other species are involved.

Functional traits A feature of an organism that has demonstrable links to the organism's function.

Genetic diversity The value, range, and relative abundance of genes present in the organisms in an ecological community.

Green infrastructure see Ecological infrastructure

Governance (of ecosystems) The process of regulating human behavior in accordance with shared ecosystem objectives. The term includes both governmental and nongovernmental mechanisms.

Habitat service The importance of ecosystems to provide living space for resident and migratory species (thus maintaining the gene pool and nursery service).

Human well-being A context- and situation-dependent state, comprising basic material for a good life, freedom and choice, health and bodily well-being, good social relations, security, peace of mind, and spiritual experience.

Indicator Information based on measured data used to represent a particular attribute, characteristic, or property of a system.

Indirect driver A driver that operates by altering the level or rate of change of one or more direct drivers.

Indirect use value The benefits derived from the goods and services provided by an ecosystem that are used indirectly by an economic agent. For example, an agent at some distance from an ecosystem may derive benefits from drinking water that has been purified as it passed through the ecosystem.

Institutions The rules that guide how people within societies live, work, and interact with each other. Formal institutions are written or codified rules. Examples of formal institutions would be the constitution, the judiciary laws, the organized market, and property rights. Informal institutions are rules governed by social and behavioral norms of the society, family, or community.

Intrinsic value The value of someone or something in and for itself, irrespective of its utility for someone else.

Management (of ecosystems) see Ecosystem management.

Mitigation (or restoration) cost The cost of mitigating the effects of the loss of ecosystem services or the cost of getting those services restored.

Natural capital An economic metaphor for the limited stocks of physical and biological resources found on earth.

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Opportunity cost The benefits forgone by undertaking one activity instead of another.

Poverty The pronounced deprivation of wellbeing. Income poverty refers to a particular formulation expressed solely in terms of per capita or household income.

Precautionary principle The management concept stating that in cases "where there are threats of serious or irreversible damage, lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environmental degradation," as defined in the Rio Declaration on Environment and Development.

Productivity Rate of biomass produced by an ecosystem, generally expressed as biomass produced per unit of time per unit of surface or volume. Net primary productivity is defined as the energy fixed by plants minus their respiration.

Provisioning services The products obtained from ecosystems, including, for example, genetic resources, food and fiber, and fresh water.

Public goods A good or service in which the benefit received by any one party does not diminish the availability of the benefits to others, and where access to the good cannot be restricted.

Range of tolerance The range of a given parameter within which an organism can function (e.g., temperature tolerance range).

Regulating services The benefits obtained from the regulation of ecosystem processes, including, for example, the regulation of climate, water, and some human diseases. (MA 2005)

Replacement cost The costs incurred by replacing ecosystem services with artificial technologies.

Resilience The capacity of a system to absorb disturbance and reorganize while undergoing change so as to still retain essentially the same function, structure and feedbacks, and therefore identity (i.e., capacity to change in order to maintain the same identity). *General resilience*: The resilience of a system to all kinds of shocks, including novel ones. *Specified resilience:* The resilience "of what, to what"; resilience of some particular part of a system, related to a particular control variable, to one or more identified kinds of shocks.

Resource Any physical or virtual entity of limited availability that provides a benefit.

Responses Human actions, including policies, strategies, and interventions, to address specific issues, needs, opportunities, or problems. In the context of ecosystem management, responses may be of legal, technical, institutional, economic, and behavioral nature and may operate at various spatial and time scales.

Scale The measurable dimensions of phenomena or observations. Expressed in physical units, such as meters, years, population size, or quantities moved or exchanged. In observation, scale determines the relative fineness and coarseness of different detail and the selectivity among patterns these data may form.

Social costs and benefits Costs and benefits as seen from the perspective of society as a whole. These differ from private costs and benefits in being more inclusive (all costs and benefits borne by some member of society are taken into account) and in being valued at social opportunity cost rather than market prices, where these differ. Sometimes termed "economic" costs and benefits. (MA 2005)

Social-ecological system An ecosystem, the management of this ecosystem by actors and organizations, and the rules, social norms, and conventions underlying this management.

Species diversity Biodiversity at the species level, often combining aspects of species richness, their relative abundance, and their dissimilarity.

Species richness The number of species within a given sample, community, or area.

Stakeholder A person, group or organization that has a stake in the outcome of a particular activity.

Substitutability The extent to which human-made capital can be substituted for natural capital (or vice versa).

Sustainability Management of resources in a way that guarantees welfare and promotes equity of current and future generations.

Sustainable use (of ecosystems) Using ecosystems in a way that benefits present generations while maintaining the potential to meet the needs and aspirations of future generations.

Threshold A point or level at which new properties emerge in an ecological, economic, or other system, invalidating predictions based on mathematical relationships that apply at lower levels. For example, species diversity of a landscape may decline steadily with increasing habitat degradation to a certain point, then fall sharply after a critical threshold of degradation is reached. Human behavior, especially at group levels, sometimes exhibits threshold effects. Thresholds at which irreversible changes occur are especially of concern to decision-makers.

Total economic value The value obtained from the various constituents of utilitarian value, including direct use value, indirect use value, option value, quasi-option value, and existence value.

Trade-offs of ecosystem services The way in which one ecosystem service relates to or responds to a change in another ecosystem service.

Transformative capacity The capacity to transform the stability landscape itself in order to become a different kind of system, to create a fundamentally new system when ecological, economic, or social structures make the existing system untenable.

Urban area (city) There is no general agreement on a definition of what is urban, and considerable differences in classification of urban and rural areas exist among countries and continents. In Europe and North America, the urban landscape is often defined as an area with human agglomerations and with >50 % of the surface built, surrounded by other areas with 30–50% built, and overall a population density of more than ten individuals per hectare. In other contexts, population size, the density of economic activity or the form of governance structure is used to delineate what is a town, city or city region, but there is significant variation in the criteria for defining what is urban.

Urbanization Urbanization is a multidimensional process that manifests itself through rapidly changing human populations and changing land cover. The growth of cities is due to a combination of four forces: natural growth, rural to urban migration, massive migration due to extreme events, and redefinitions of administrative boundaries.

Utility A measure of satisfaction.

Valuation The process of expressing a value for a particular good or service in a certain context (e.g., of decision-making) usually in terms of something that can be counted, often money, but also through methods and measures from other disciplines (e.g., sociology, ecology, etc.).

Value The contribution of an action or object to user-specified goals, objectives, or conditions.

Viable populations Organism populations that can survive in the wild.

Vulnerability Exposure to contingencies and stress, and the difficulty in coping with them. Three major dimensions of vulnerability are involved: exposure to stresses, perturbations, and shocks; the sensitivity of people, places, ecosystems, and species to the stress or perturbation, including their capacity to anticipate and cope with the stress; and the resilience of the exposed people, places, ecosystems, and species in terms of their capacity to absorb shocks and perturbations while maintaining function.

Willingness to pay The maximum amount that a person is willing to pay for a good they do not have.