10

URBANIZATION, HABITAT LOSS AND BIODIVERSITY DECLINE

Solution pathways to break the cycle

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The interactions between urbanization with biodiversity and ecosystem services that take place defy simple generalizations. There is increasing evidence for the negative impacts of urbanization on biodiversity, most directly in the form of habitat loss and fragmentation. Recent forecasts suggest that the amount of urban land near protected areas is expected to increase, on average, by more than three times between 2000 and 2030 (from 450,000 km^2 *c.* 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. However, there is also ample evidence pointing to opportunities to shape urbanization strategies in a way to reconcile urban development and biodiversity conservation strategies (Elmqvist et al. 2013). While gaps in knowledge and practice remain, an increasing number of studies scrutinize the interactions of urbanization with biodiversity and ecosystem services at local, regional and global scales.

Urbanization and biodiversity

Urbanization and biodiversity interact in multifaceted and complex ways (McKinney 2002). Both the size and spatial configuration of urban areas matter for biodiversity (Alberti 2005; Tratalos et al. 2007). While some urban areas have high local species richness, this is typically at the cost of native species (McKinney 2002, 2006). Urban expansion may lead to habitat fragmentation, potentially resulting in genetic or demographic isolation of native species (Ricketts 2001). A major impact of the expansion of urban areas on native species is on their dispersal through changes in habitat configuration and connectivity (Bierwagen 2007). Urbanization is also a major threat to endemic species due to increased incidence of colonization by introduced species (McKinney 2006, 2008).

Urbanization impacts biodiversity and ecosystem services both directly and indirectly. Direct impacts primarily consist of habitat loss and degradation, altered disturbance regimes, modified soils and other physical transformations caused by the expansion of urban areas. Indirect impacts include changes in water and nutrient availability, increases in abiotic stressors such as air pollution, increases in competition from non-native species, and changes in herbivory and predation rates (Pickett and Cadenasso 2009).

Urban expansion and landcover changes

The most obvious direct impact of urbanization on biodiversity is landcover change due to the growth of urban areas. Although urban areas cover less than 3 percent of the Earth's surface, the location and spatial pattern of urban areas have significant impacts on biodiversity (Müller et al. 2013). Worldwide, urban expansion occurs faster in low-elevation coastal zones, which are biodiversity-rich, than elsewhere. Although urban land occupies less than 1 percent of land in the majority of terrestrial ecoregions, it most heavily impacts ecoregions along coasts and on islands, affecting about 10 percent of terrestrial vertebrates found therein. Likewise, more than 25 percent of all endangered or critically endangered species will be affected, directly or indirectly, by urban expansion by 2030 (Güneralp and Seto 2013; McDonald et al. 2013).

In a simple exercise, McDonald et al. (2014), assuming a linear species-area curve and using the expected amount of urban growth (and hence habitat loss) between 2000 and 2030, predict the expected number of endemic vertebrate species that might be lost due to urbanization (see Figure 10.1). They find that urban growth in 10 percent of all ecoregions would account for almost 80 percent of the expected loss in species. This implies that safeguarding species from urbanization in a relatively small number of ecoregions could have a disproportionately large benefit in terms of avoiding biodiversity loss. For example, cities have historically been concentrated along coastlines, on some islands and on major river systems, all of which are often areas of high species richness and endemism.

More than 25 percent of the world's terrestrial protected areas are within 50 km of a city (McDonald et al. 2009). This close proximity has multiple effects, positive and negative, not only on the protected area, but also on the neighboring human population. Negative effects include feral pets, vandalism, illegal dumping, poaching of animals and plants, land squatting and introduction of invasive species. Positive outcomes could include increased potential for recreational activities, eco-tourism and nature-based education thus potentially contributing to increased environmental awareness among residents and visitors. Establishing management practices such as biodiversity corridors in urbanizing regions is desirable, but will require coordinated efforts among administrative bodies within and among nations as well as local residents. The identification and implementation of such corridors may have additional significance considering the migration of species in response to shifts in their ranges with climate change (Forman 2008).

By 2030, the urban lands near protected areas (PAs) are forecasted to increase substantially in almost all world regions (see Figure 10.2; McDonald et al. 2008; Güneralp and Seto 2013). Even in the highly urbanized US, urban land near protected areas may grow by almost 70 percent by the middle of this century (Martinuzzi et al. 2015). By 2030, China will most likely have more urban land within 50 km of their respective PAs than North America or Western Europe. The largest proportional change during the same time period, however, will likely be in Mid-latitudinal Africa, where urban land near PAs is forecasted to increase about 20 times.

Of 34 identified biodiversity hotspots (Mittermeier et al. 2004; Myers et al. 2000), the Mediterranean hotspot contains the most urban land, hugging the coastlines of three continents with different geographic, cultural, social and economic characteristics. In the Mediterranean, for a hotspot that is already diminished and severely fragmented, even relatively modest decreases in habitat can cause pressure on rare species to rise disproportionately (Tilman et al. 1994). By 2030, forecasts suggest that the Mediterranean Basin may become the only hotspot containing more than 100,000 km^2 of urban land (see Figure 10.3; Güneralp and Seto 2013). On the other hand, four biodiversity hotpots that were relatively undisturbed

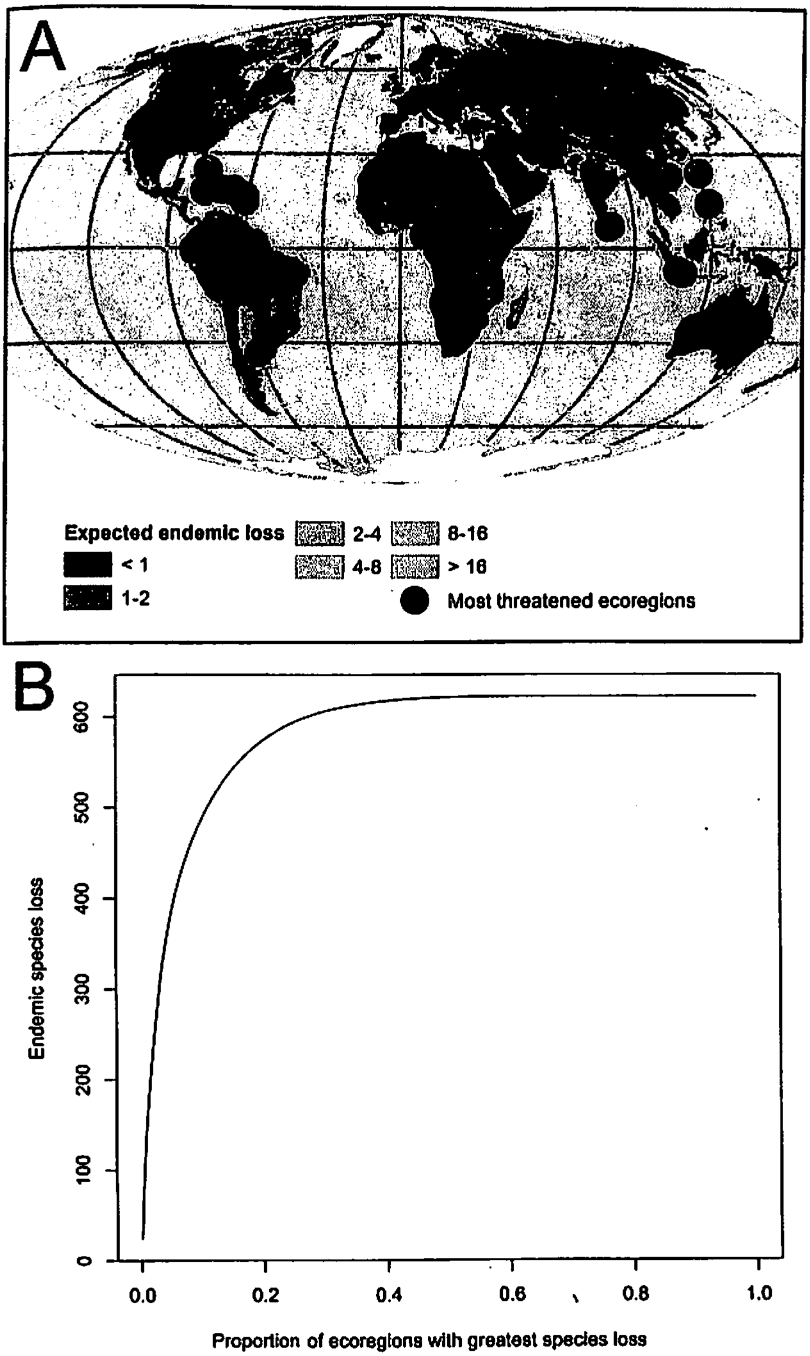

Figure 10.1 Endemic vertebrate species expected to be lost due to urban area expansion: (A) the 25 most threatened ecoregions are shown with dots; (B) the majority of species loss due to urbanization will be in a small fraction of ecoregions

Source: Reproduced under CC license from McDonald et al. (2014) with permission of the Solutions journal (2014).

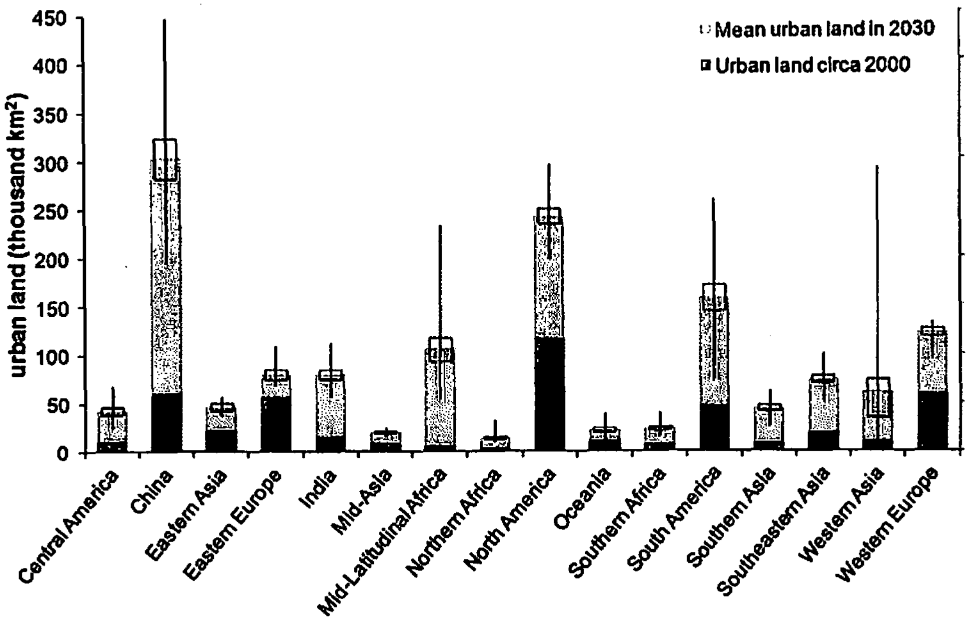

Figure 10.2 Urban extent within a distance of 50 km of PAs by geographic region *c.* 2000 and as forecasted in 2030

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by urban land change by 2000 are forecasted to experience the highest rates of increase—over ten times—in urban land cover by 2030. These hotspots are: Eastern Afromontane; Guinean Forests of West Africa; Western Ghats and Sri Lanka; and Madagascar and the Indian Ocean Islands (see Figure 10.3).

Biodiversity hotspots can span national borders creating jurisdictional challenges and issues for management and planning. These challenges and issues cannot solely be met by local level solutions and will require policy responses at a broader scale such as national and international levels. Appropriate strategies with sufficient breadth to protect biodiversity and ecosystem functioning in these multi-jurisdictional hotspots will need to be assessed and implemented through trans-border and regional cooperation among the countries involved (Chettri et al. 2007). Urban expansion will significantly impact freshwater biodiversity on a global scale. Direct and indirect impacts will be most critical in places where there is a confluence of large urban water demands relative to water availability and high freshwater endemism. For instance, Western Ghats of India is projected to have a population of 81 million people with insufficient access to water by 2050. The region also possesses 293 fish species, 29 percent of which are endemic (McDonald et al. 2008). As water resources in the region become limiting, potential species extinction could be substantial.

Urbanization and species patterns

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 'natural' elements in designing 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). 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 four factors: 1) site availability; 2) species availability (native and non-native); 3) species performance (how well a species does in urban landscape); and 4) site history (Pickett et al. 1987; Williams et al. 2009; Müller et al. 2013). Habitat loss and degradation and the 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. The net result is sometimes called 'biotic homogenization' (McKinney 2006). 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 finds that urban areas filter out or exclude, on average, about one-third of native species existing in the surrounding region (Aronson et al. 2014). While this loss of diversity is problematic, 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

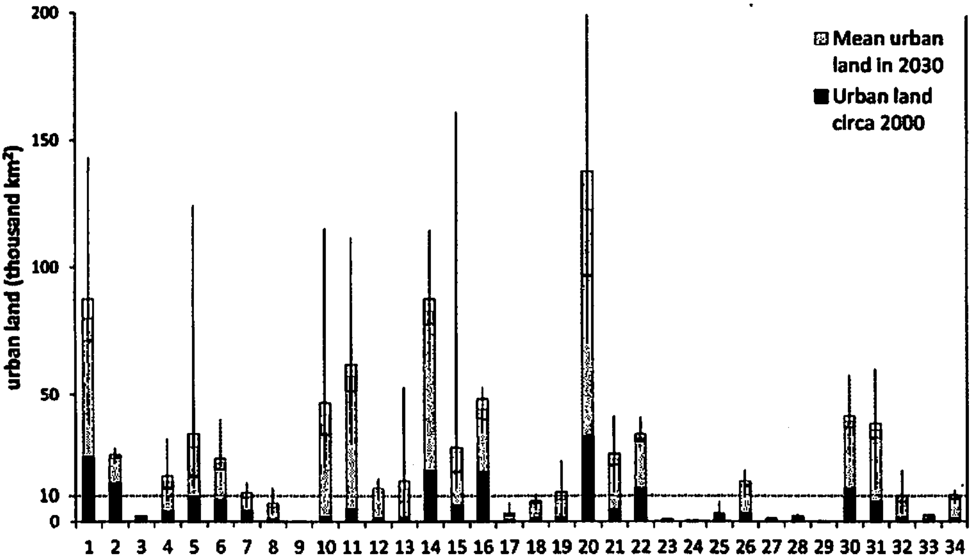

Figure 10.3 Urban extent in biodiversity hotspots *c.* 2000 and as forecasted in 2030[1]

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and region, especially if the surrounding landscapes have been simplified through agriculture or forestry (Barthel et al. 2005). For instance, native species richness declines and non-native species richness increases as one moves from the rural fringe to the urban core with approximately 30–50 percent 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 invasive and native generalists, which are tolerant to the urban conditions. However, the literature also provides evidence that contradicts these generalities. For example, Hope et al. (2003) report that species richness in Phoenix, Arizona, US, a city located in a desert environment, increases with urbanization because of human influences such as irrigation and ornamental landscaping.

Urbanization and ecosystem services

Urban areas affect many ecosystem services on scales ranging from local to global. One of the most critical services on a regional to global scale is the provision of freshwater (McDonald et al. 2013). 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 (Güneralp et al., 2015a; see Pfister et al., Chapter 17 in this volume). 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 shortages, defined as having less than 100 L/person/day of sustainable surface and groundwater flow within their urban extent. By 2050, this number will reach almost a billion people due to population growth. Furthermore, climate change is projected to cause water shortages for an additional 100 million urbanites (McDonald et al. 2011).

Urbanization also affects regulatory hydrological services (Gómez-Baggethun and Barton 2013), which are usually defined as public goods. Consider an expanding city where new residential areas are replacing forests. This increases the impermeable surface area, which leads to increased volumes of surface water runoff, and thus increases the vulnerability to flooding of downstream communities. Depending on the rain event, vegetation reduces surface runoff following precipitation events by increasing infiltration. Urban landscapes with 50–90 percent impervious cover can lose 40–83 percent of rainfall to surface runoff compared to 13 percent in forested landscapes (Bonan 2002). Urban areas thus both depend on upstream natural habitats for regulating water flows and impact provision of this ecosystem service to downstream communities.

Ecosystem services provided by the urban forest and especially parks within cities are vital for human health and well-being, yet these are not always adequately considered during urban planning processes. These services include recreational value, aesthetic benefits and benefits to human physical and mental health (Gómez-Baggethun and Barton 2013; McDonald, Chapter 29 in this volume). Because city environments may be stressful for inhabitants, the recreational aspects of urban ecosystems are among the highest valued ecosystem services in cities. While these cultural ecosystem services are very important to the well-being of urban dwellers, they are often ignored by urban planning authorities; moreover, unplanned urban developments do not usually set aside any land for natural habitat to provide these services.

Although land is a scarce resource in a city and most will be used for development, the role of urban planning is to create an urban form that provides for the well-being of residents, which requires that some natural habitats are set aside as protected areas.

Urbanization, biodiversity, ecosystem services and governance

Minimizing habitat and biodiversity loss and limiting degradation of ecosystem services also require cities to integrate ecological knowledge into their urban planning practices (Niemelä 1999; Puppim de Oliveira et al. 2011). Specifically, urban planning practices need to become more attuned to conservation of biodiversity and preservation of ecosystem services that are of critical importance for the inhabitants of the urban areas (McDonald et al. 2014; Puppim de Oliveira et al. 2011). In this respect, the dissemination of information and connection of science to practitioners is an important aspect of formulating sound urbanization strategies that explicitly acknowledge and consider conservation of biodiversity (Güneralp and Seto 2013). However, one of the critical prerequisites to ensure this integration is that urban planners are equipped with the requisite institutional capacity (Sandström et al. 2006; Blicharska et al. 2011).

Novel ecosystems and communities composed of both native and non-native species may give us insights into how future ecosystems in urban landscapes may function. Novel plant and animal communities are continuously assembled in urban areas, either on abandoned land or with active manipulation and management. These communities can play an important role in the generation and maintenance of ecosystem services including water, fuel and food as well as recreation within the urban areas. 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. 2013).

Many biodiversity hotspots threatened by urban growth are located in developing countries (countries forecasted to have the greatest increase in population), and may have fewer financial resources to devote to land protection than cities in developed countries. Moreover, since the attention of municipal governments in developing countries is often understandably focused on regulatory services such as providing clean drinking water and sanitation to their burgeoning residents, biodiversity protection may not be seen as a municipal priority. However, globally there is substantial interest by many people in preventing massive biodiversity loss in these biodiversity hotspots that face continuing urbanization. This spatial disconnect between those making the decisions in cities in biodiversity hotspots and those who care about the biodiversity losses elsewhere might only be overcome by a global effort to protect these biodiversity hotspots from further urban encroachment. This effort must include focusing conservation funding from organizations and governments in the developed world in these hotspots (McDonald et al. 2014).

In their study on potential direct impacts of urban expansion on biodiversity conservation in China, Güneralp et al. (2015b) discuss in detail the obstacles in incorporating biodiversity and ecosystem services into land use planning in the country. Examining historical patterns of urban population growth and expansion, and using forecasts from the Intergovernmental Panel on Climate Change (IPCC) on gross domestic product and projections by the United Nations on urban population growth, they predict that by 2030, urban land in China will reach over 400,000 km^2, which corresponds to a fourfold increase in urban land over 30 years. Such growth in urban areas will increase pressure on the already stressed protected areas and the biodiversity hotspots in the country. This poses a formidable challenge to the country's

goal of biodiversity conservation, and calls for more effective land use planning and regulation, in particular at the regional and provincial level. China's entire planning system encompasses various government agencies that formulate and approve land use plans. Under this system, the land and fiscal policy reforms in the 1980s and 1990s created an institutional environment in which local and municipal governments came to increasingly rely on land leasing to developers as a key source of revenue. This resulted in rapid expansion of urban areas which, despite further reforms to stem the trend, continues almost unabated.

To integrate the goal of well-functioning cities with that of well-functioning ecosystems, China needs to incorporate ecological considerations in the regional and provincial-level plans and decisions and effectively regulate development decisions at the municipal level. However, it will be challenging to overcome the entrenched governance practices and special interests (Güneralp et al. 2015b). For example, although China's recently unveiled urbanization plan recognizes the over-reliance of local governments on land leasing as a source of revenue (China State Council 2014), it is uncertain if the plan, despite its lofty goals, will lead to any significant changes in planning practices.

Of the actions proposed to ameliorate urban effects on biodiversity, setting aside large parcels of native habitats in those parts of biodiversity hotspots facing urbanization pressure may provide the best opportunity for regional floral and faunal species to persist in the face of climate change and a surrounding urbanizing landscape. These protected areas would need to be large enough to contain the spectrum of natural disturbances as well as native habitats. With land conservation, a number of landscape designs are possible. For instance, one design for large parcels would make these areas composed of multiple-utilization zones (Noss and Harris 1986). The interior zone would be road-free and managed to conserve native flora and fauna. By comparison, the perimeter would serve as a buffer that is used for multiple benefits and linked to other areas. An example would be the Tijuca Forest in Rio de Janeiro, Brazil (Herzog 2013). Large parcels can, to some extent, buffer local climatic changes and contain more individuals of a single species thus enhancing its genetic breadth. Even these large areas, however, will not be immune to human intrusions; so, natural resource managers must also continually adapt to changing circumstances.

The global challenge that urbanization poses for ecosystem services and biodiversity demands a global response. McDonald et al. (2014) present three potential solutions to address this challenge: 1) treating ecosystem services as an urban utility; 2) a global effort to protect those biodiversity hotspots under urbanization pressure; and 3) international coordination for urban sustainability. In several cities around the world, these solutions are being experimented with at present with varying levels of success. Importantly, any solutions to reconcile ongoing urbanization and conservation would require policies that work in harmony across scales, from local to regional to global (McDonald et al. 2014). In particular, establishing effective biodiversity conservation strategies in regions that are expected to undergo significant urban expansion require coordinated efforts among multiple cities, provinces and even countries. Such coordination, however, has been hard to achieve even among conservation bodies under existing regional and global governance mechanisms (Larigauderie and Mooney 2010). The recently formed Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES; www.ipbes.net) aims to remedy this lack of coordination by, among others, conducting periodic sub-regional, regional and global assessments on the state of the planet's biodiversity, its ecosystems and the essential services they provide to society (Larigauderie and Mooney 2010). Established in April 2012, the IPBES will act as an independent intergovernmental body, much like the IPCC; it will be open to all member countries of the United Nations. Clearly, the impacts of urbanization on biodiversity are critical enough to be included in these

assessments (McDonald et al. 2014). In this vein, the Cities and Biodiversity Outlook (CBO) that is endorsed by the Convention on Biological Diversity (CBD) is the first ever comprehensive assessment of the interaction of cities and biodiversity and ecosystem services (Elmqvist et al. 2013).

Gaps in knowledge and practice

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). 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, climate change is a major driver of change that likely will affect future urban biodiversity and ecosystem services.

The need for urbanization strategies that consider biodiversity conservation is especially acute for those in developing countries where most urban expansion near protected areas and in biodiversity hotspots are expected (Niemelä 1999; Puppim de Oliveira et al. 2011; see also Roberts, Chapter 35 in this volume, for a discussion of city action and biodiversity planning in Durban, South Africa). There are two challenges to overcome in this respect: first is building a firm theoretical foundation on which to ground the research on the relationship between urbanization and biodiversity; the second is building communication channels between the researchers and the stakeholders including citizens, community organizations, planners and government representatives alike. Building such channels of information between science and practitioners for communication of concerns and insights will be an important tool for formulating more robust urbanization strategies in regards to biodiversity.

With respect to ecosystem services, little is known about the negotiated interactions that lead to trade-offs and synergies in the demand for particular bundles of ecosystem services accessible to different socio-economic or livelihood groups in urban environments (but see Colding et al. 2006; Andersson et al. 2007). These interactions play a crucial role in shaping 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). Considering the multitude of services that ecosystems provide as well as the demand for these services, interdisciplinary approaches are needed to better safeguard, and benefit from, these services.

A better understanding is required of the supply, needs and management of urban ecosystem services in large regions in South Asia, Africa and Latin America, which are developing rapidly and face some of the greatest threats to protected areas and biodiversity hotspots in the future (Güneralp and Seto 2013). However, this does not necessarily mean that local knowledge is non-existent. Most likely traditional ecological knowledge, at the local level, is being used every day in more informal management decisions pertaining to ecosystem services. 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 are chosen for their medicinal, food or cultural properties (Jaganmohan et al. 2012). Local knowledge and practices could be mobilized in

multiple ways through, for example, citizen science initiatives, and thus could support more formal governance and management of urban ecosystem services.

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 services concepts, methods and classifications (Luck et al. 2012; Gómez-Baggethun and Barton 2013). For example, 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). Particularly across Asia and Africa, many sacred conceptualizations of nature persist in cities (e.g. protection of sacred keystone species such as *Ficus religiosa* across cities in India). There are also numerous equity and environmental justice issues related to cultural ecosystem services, but these are often poorly documented (D'Souza and Nagendra 2011). In addition, to better capture the value of biodiversity and ecosystems in reducing urban vulnerability to shocks and disturbances, there is a particular need of new valuation techniques that utilize a resilience and inclusive wealth perspective (Gómez-Baggethun et al. 2013). The insurance value of an ecosystem is closely related to its resilience and self-organizing capacity, and the extent to which it may continue to provide flows of ecosystem services benefits with stability over a range of variable environmental conditions.

Concluding remarks

Main messages from the chapter include:

- Notwithstanding uncertainties inevitable in any study on future trends, it is increasingly clear that urbanization will continue to affect biodiversity and ecosystem services around the world.
- Most of the effects on biodiversity and ecosystem services will take place in the developing world with limited means to address each and every challenge urbanization presents.
- As the world continues to urbanize, there is an increasing need for urban decision-makers and citizens to adopt policies and practices to integrate nature into daily lives; after all, cities may very well offer the key to a globally sustainable future underpinned by nature-based solutions and ecosystem-based adaptation.

Identified needs for future research and practice include:

- interdisciplinary approaches that examine the trade-offs and synergies of urban ecosystems and the interactions of user demand as it pertains to different stakeholders groups;
- mechanisms through which biodiversity and ecosystem-based solutions can be integrated into urban development and climate change adaptation decision-making and planning;
- cross scale understandings of ecosystem processes across rural–urban gradients and the resulting interactions and feedbacks from both current and future climate change and land use changes;
- more quantitative and qualitative studies on ecosystem services within developing regions, where both rapid urbanization and projected loss in protected areas and hotspots are expected to occur; and,

- the development of more robust theoretical frameworks that integrate the understudied social and cultural values of urban ecosystems with monetary values in decision-making processes.

Note

1 (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.

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