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SHORT COMMUNICATION

Promoting and preserving biodiversity in the urban forest

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Abstract

Efforts at mitigating global biodiversity loss have often focused on preserving large, intact natural habitats. However, preserving biodiversity should also be an important goal in the urban environment, especially in highly urbanized areas where little natural habitat remains. Increasingly, research at the city/county scale as well as at the landscape scale reveals that urban areas can contain relatively high levels of biodiversity. Important percentages of species found in the surrounding natural habitat, including endangered species, have been found in the urban forest.

This contribution concisely highlights some examples of urban biodiversity research from various areas of the world. Key issues involved in understanding the patterns and processes that affect urban biodiversity, such as the urban–rural gradient and biotic homogenization, are addressed. The potential for urban areas to harbor considerable amounts of biodiversity needs to be recognized by city planners and urban foresters so that management practices that preserve and promote that diversity can be pursued. Management options should focus on increasing biodiversity in all aspects of the urban forest, from street trees to urban parks and woodlots.

Keywords: Green-space; Management options; Urban forestry; Urban planning

Introduction

Rapid loss of biodiversity is a global phenomenon. It is estimated that possibly half or more of all current species could be at risk of extinction in the foreseeable future (Myers, 1996; Sax and Gaines, 2003). This loss of biodiversity is of critical concern, given that an increasing amount of research indicates that diversity plays an important role in long-term ecosystem functioning (Groombridge and Jenkins, 2002). Many factors are contributing to biodiversity loss including habitat modification, competition from introduced species, human demand for certain species and products, and rapid environmental changes such as climactic fluctuations (Groombridge and Jenkins, 2002).

Preserving large, intact areas of natural habitat is a key means of preserving biodiversity. However, this may not be feasible in highly urbanized locations where there is little natural habitat remaining. One viable alternative is promoting biodiversity in the urban ecosystem. The loss of natural forest stands and the degradation of those stands that still remain, places great importance on preserving and promoting the biodiversity that is present in the urban forest.

The following sections will demonstrate how urban and suburban green-spaces can be biologically rich by providing examples addressing urban tree biodiversity. Key issues involved in understanding the patterns and processes that affect urban biodiversity, such as the urban–rural gradient and biotic homogenization, are also addressed. Management practices that preserve and promote urban biodiversity will be also discussed.

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Urban forests can contain relatively high levels of biodiversity

The urban forest, which includes vegetation along urban streets and in urban parks, woodlots, abandoned sites, and residential areas, can comprise a significant percentage of a nation's tree canopy. In the contiguous United States, trees in urban counties account for nearly 25% of the nation's total tree canopy cover (Dwyer et al., 2000). Given the current rapid rate of global urbanization, the percentage and value of urban forests will increase. Nowak and Walton (2005) note that, "…expanding urbanization increases the importance of urban forests in terms of their extent and the critical ecosystem services they provide to sustain human health and environmental quality in and around urban areas."

Biodiversity can be regarded as one such ecosystem service. Traditionally, urban areas have been regarded as locations of low biodiversity that are dominated by non-native species. However, evidence is mounting that urban and suburban areas can contain relatively high levels of biodiversity (Balmford et al., 2001; Jim and Liu, 2001; Araújo, 2003; Godefroid and Koedam, 2003a; Cornelis and Hermy, 2004; Kühn et al., 2004). It is important to establish that urban and suburban areas can be biologically rich so that urban foresters and city planners can actively manage to preserve and promote that diversity. Research has been conducted in numerous countries and across various spatial scales that lends support to this assertion.

Biodiversity research at the city/county scale

Most research on urban biodiversity has focused on the diversity within a city or county. Researchers have found that urban forests can contain a significant percentage of species that naturally occur in an area. Surveys of 15 urban and suburban parks in Flanders, Belgium, revealed that the 15 parks contained about 30%, 50%, 40%, and 60% of the total number of wild plant species, breeding birds, butterflies, and amphibians still occurring in Flanders, respectively (Cornelis and Hermy, 2004). Tree species with the highest cover were *Fagus sylvatica* L., *Acer pseudoplatanus* L., *Quercus robur* L., and *Fraxinus excelsior* L. Flanders is a highly urbanized and densely populated area, with forests making up only 10% and nature reserves only 1.7% of Flanders (Cornelis and Hermy, 2004). Urban parks therefore function as an important reserve of biodiversity in Flanders.

In Guangzhou City, China, Jim and Liu (2001) found over 250 species after surveying over 115,000 trees in parks, on university grounds, and along city streets. In this large, highly developed, subtropical city, the most dominant species were native broadleaves. The top three abundant species were *Ficus virens* Aiton, *Caryota mitis* Loureiro, and *Melaleuca leucadendra* L. (Jim and Liu, 2001). Plant species richness in Guangzhou is actually higher than it is in the degraded forests of the surrounding countryside (Zheng, 1995 cited in Jim and Liu, 2001, p. 105). This phenomenon has been observed in other areas of the world as well. In New Zealand, the second largest city, Christchurch, has higher floral diversity than its surroundings – a converted landscape of pastoral land with very few remnants of indigenous forest (Stewart et al., 2004).

Urban forests have also been shown to harbor endangered species and species of high conservation value. Urban green areas in Sweden are home to endangered species as identified on the Red List of Swedish species. It is estimated that densely populated Stockholm County contains two-thirds of red-listed species (Colding et al., 2003). Red-listed plant species such as *Dryopteris cristata* L. and *Buxbaumia viridis* (Moug.) Moug. & Nestl. were observed in Roslagen, Stockholm County (Gustafsson, 2002).

Biodiversity research at the landscape scale

Biodiversity research on a broader landscape scale has also revealed that urban areas can contain a relatively high level of biodiversity. Kühn et al. (2004) examined the landscape of Germany by dividing the country into city and non-city grid cells. Non-native as well as native plant species richness was significantly higher in the city grid cells. They suggest that this may be due to geological diversity. Both the location of German cities and the locations of high native plant diversity positively correlated with locations that were geologically diverse. The authors conclude that cities may be disproportionately situated in areas of naturally occurring high biodiversity (Kühn et al., 2004).

Not only Germany, but urban areas across all of Europe seem to contain higher levels of biodiversity than unpopulated areas (Araújo, 2003). Araújo (2003) found a positive correlation between human population density and plant, mammal, and reptile and amphibian species richness throughout Europe. These findings are largely consistent with those obtained from similar studies conducted in Africa (Araújo, 2003). Balmford et al. (2001) used data from across sub-Saharan Africa and found that human population density positively correlated with bird, mammal, snake, and amphibian species richness. This association was true for widespread, narrowly endemic, and threatened species.

Araújo (2003) suggests two possible explanations for this relationship. First, factors causing an area to be suitable for people may be similar to the factors causing an area to be suitable for other species. Secondly, human actions directly and indirectly increase the total number of species, for example, through introduced species and increased landscape heterogeneity (Araújo, 2003). Whatever the cause, this association further emphasizes the importance of biodiversity conservation in and around dense human settlement.

Understanding the patterns and processes that affect urban biodiversity

The urban–rural gradient

Despite the evidence that urban and suburban areas can harbor relatively high levels of biodiversity, it is generally agreed upon that the heavily built-up urban core does not support nearly as many species, especially those that are native, as compared to less urbanized areas. Many studies have shown that the loss and fragmentation of natural habitat has reduced the richness of taxa including plants, birds, insects, and mammals in the urban core to less than half of that found in rural areas (McKinney, 2002). Competition from exotic-invasive species further reduces native species diversity.

As one moves from rural areas towards the city core, human population density, road density, air and soil pollution, average air temperature, soil compaction, and soil alkalinity have been shown to increase (McKinney, 2002). As these changes occur, natural habitat is lost and is replaced by the built environment, managed vegetation, and ruderal vegetation. Many analyses rely on this urban–rural gradient approach, where biodiversity is analyzed along a transect from the inner city to surrounding, less-altered ecosystems. Long-term biodiversity management goals may be implemented more effectively by applying our understanding of the urban–rural gradient. For example, McKinney (2002) suggests that restoring managed and ruderal habitats may be more feasible in the urban core and highly urbanized suburbia than acquiring remnant habitat.

However, the processes and patterns that shape biodiversity within and around cities need to be more fully understood (Kinzig et al., 2005). Our traditional generalizations about the urban–rural gradient may not be valid in all situations. Godefroid and Koedam (2003a) studied the spatial variation of plant assemblages in the remnant Sonian Forest, which is partially situated within the administrative limits of the Brussels Capital Region, Belgium. As expected, forest stands that were close to urban areas supported more pioneer and exotic species than the forest interior. Surprisingly though, these stands were also more likely to be colonized by species classified as ancient forest species and rare species. Up to 23% of rare species were found in the forest edge, while less than 5% were found in the forest interior (Godefroid and Koedam, 2003a). Studies such as this reveal that we may need to update our traditional assumptions about the urban–rural gradient and forest edge effects.

The role of socioeconomics and culture

As we further understand the processes that affect biodiversity, it becomes apparent that socioeconomic status and culture of the resident people are also a shaping force for urban biodiversity. Kinzig et al. (2005) suggest augmenting the traditional gradient approach discussed above with socioeconomic and cultural measures as the preferences, desires, and wherewithal of the people in the landscape matter. Common sense tells us that if household behaviors such as deciding what to plant in the yard, are strongly structuring the urban forest, and these household behaviors vary based upon socioeconomic status and culture, then we would expect to see biodiversity vary across different neighborhoods (Kinzig et al., 2005).

This hypothesis was tested by Kinzig et al. (2005) when they assessed bird and plant diversity within neighborhood parks and residential areas in Phoenix, Arizona. Their results indicate that socioeconomics *do* play an important role; as economic status increased, species diversity generally increased. There was a significant effect of adding socioeconomic status to the model for plant diversity in neighborhoods, but the effect was not significant for plant diversity in parks (Kinzig et al., 2005).

A study of the urban and periurban areas of Santiago, Chile revealed similar results. Santiago is composed of 36 metropolitan boroughs with a total of about six million inhabitants (Luz de la Maza et al., 2002). Indices of species richness and evenness were correlated with income level – as income level rose, so did richness and evenness (Luz de la Maza et al., 2002).

Biotic homogenization

Central to the issue of urban biodiversity is biotic homogenization which may be defined as the process of replacing localized native species with increasingly widespread non-native species (McKinney, 2006). The establishment of the rock dove (*Columbia livia* L.), house mouse (*Mus musculus* L.), and feral house cat (*Felis catus* L.) in ecosystems throughout the globe and their direct impact on the loss of native species are some prime examples of biotic homogenization. Faced with no natural enemies, exotic species may out-compete native species for resources leading to reduced numbers of native species and/or local extinctions.

The introduction of exotic flora and fauna has greatly affected native and overall species richness and abundance throughout the globe. The problem with biotic homogenization is that although diversity at the local or regional scale may increase with the introduction of exotics, overall biodiversity at the global scale decreases. Global biodiversity is expected to continue to decline for at least the next few centuries (Sax and Gaines, 2003).

Urbanization promotes biotic homogenization by increasing the importation of non-native species through accidental and intentional importation. Urban areas also provide a favorable habitat for the establishment of non-native species by providing food resources, by reducing the threat of natural enemies, and/or by altering the physical environment in favor of the non-native species, for example, through the urban heat island effect (McKinney, 2006). Cities also serve as the main source from which introduced species can further spread into an area (Tait et al., 2005). Therefore, native species restoration programs and exotic-invasive species management plans are critically important in urban areas.

Biotic homogenization cannot be fully understood without having documented, long-term, historical data. Understanding the interactions of exotics and natives over time within cities is crucial. Unfortunately, particularly for urban ecosystems, there is a general lack of such temporal biodiversity studies (Tait et al., 2005).

A notable exception is Adelaide, Australia, an isolated city with over a million residents (Tait et al., 2005). By comparing current biodiversity data with historical data that has been regularly and systematically collected since 1836, Tait et al. (2005) found a dramatic change in species composition. The total number of plant and animal species had increased by nearly 30%, with a minimum of 648 species that had been introduced. However, at least 132 species had become locally extinct (Tait et al., 2005).

More studies like this are needed. By studying historical patterns of change, one may be better able to predict species that are at risk of local extinctions and be better equipped to establish long-term biodiversity management plans.

The complexity of biodiversity

Despite our best efforts to understand the processes and patterns that affect urban biodiversity, biodiversity itself is a complex concept. Biodiversity comprises both richness and evenness and is measured on three different organizational levels – genome, assemblage or species, and landscape (Angermeier, 1994). A study will typically measure biodiversity at only one of these levels. Incorporating results from various studies is therefore complex, making biodiversity difficult to compare across space and time (Angermeier, 1994). Furthermore, due to the complexity of biological systems, even expert measurements of biodiversity may be relatively rough estimations that rely on indicators or extrapolation (Gyllin and Grahn, 2005).

In order to measure and manage biodiversity more efficiently, a consensus needs to be reached on its meaning. Defining biodiversity in an unambiguous way is especially important in the urban context where residential areas are often incorporated into biodiversity planning (Gyllin and Grahn, 2005). To gain public input and support, the concept of biodiversity must be defined so that is understandable and acceptable to local residents (Gyllin and Grahn, 2005).

Managing for urban biodiversity

Once urban foresters and city planners recognize that urban forests are capable of supporting considerable amounts of biodiversity and have a basic understanding of the patterns and processes that affect biodiversity, addressing the question, "What can be done to actively preserve and promote that diversity?," becomes crucial. Many management options are available and may or may not be applicable depending on the spatial and political contexts of the urban forest. Management should focus on increasing biodiversity in all parts of the urban forest – street trees, parks, woodlots, abandoned sites, and residential areas.

Tree inventories

Conducting a tree inventory is often the first step towards managing for urban biodiversity. An inventory of the urban forest establishes a baseline for setting management objectives by determining what you have and where you have it. Urban tree inventories may collect various information including tree species, location, and overall health. Depending on financial resources, information may be gathered on all trees in a municipality, a sample of trees, or only a specific component of the urban forest such as street trees. Once field data are gathered and analyzed, they may be used for tree species diversity and tree age diversity management as well as other management objectives. Tree inventories can also be used in public relations, as an educational tool, and to advocate for increased funding and urban forest program support.

Rapid advances in technology have made urban tree inventories a popular and efficient management tool. The ability to use geographic information systems on hand-held computers has revolutionized tree inventory data collection. The GIS unit can be brought into the field and data may be inputted quickly, accurately, and may be geo-referenced. A number of urban forest assessment tools are currently available, including American Forests' CITYgreen® and the United States Forest Service's i-Tree®. Geospatial analyses such as that described by Löfvenhaft et al. (2002) can also be very useful. Using aerial photographs, Löfvenhaft et al. (2002) created a model for the city of Stockholm, Sweden that identifies ecologically valuable biotopes. The model has been used effectively as a planning and design tool by the National Urban Park in Stockholm (Löfvenhaft et al., 2002).

Planting for biodiversity

Much emphasis has recently been placed on native species restoration in the urban environment. As previously explained, biotic homogenization decreases global biodiversity, and the importance of planting native species while reducing the impact of invasive species has been recognized. Many municipalities have set up invasive species management programs and do not actively plant invasive species.

When planting trees in the urban context, native species should always be preferred, but cultivars and non-native species that are not invasive, should also be given due consideration. Many tree cultivars such as *Platanus x acerifolia* and *Acer x freemanii* have been hybridized and bred to perform well under unfavorable urban conditions. Urban environments typically have more in common with one another than with their surrounding natural environment (McKinney, 2002). Urban stresses such as restrictive soil volume and crown space, soil compaction, soil and air pollution, high salinity, and vandalism limit the number of species that are capable of successfully growing in urban environments. Selection programs in various countries are under way to find better suited tree species that are capable of performing well under urban stresses (Sæbø et al., 2003).

The importance of increasing the variety of species planted, whether native or non-native, is more apparent now than ever before. The frequency of exotic pest introductions is increasing, often with devastating results. For example, in North America, the emerald ash borer or EAB (*Agrilus planipennis* Fairmaire), an exotic pest from Asia, was first identified in Detroit, Michigan and Windsor, Ontario in 2002 (Poland and McCullough, 2006). It attacks many North American ash species including *Fraxinus pennsylvanica* Marsh, *F. Americana* L., *F. nigra* Marsh, and to a lesser extent *F. quadrangulata* Michx. Extensive feeding of EAB larvae disrupts the host tree's translocation, girdling the tree and resulting in tree death in 1–3 years. Ash trees that can serve as EAB hosts have been planted extensively in the United States as a popular shade tree. It is estimated that up to 15 million ash trees in urban and forested areas have been killed by EAB (Poland and McCullough, 2006). Despite eradication efforts, EAB is rapidly spreading throughout Michigan and has been detected in Ohio and Indiana (Poland and McCullough, 2006).

The importance of planting for diversity becomes apparent when pest outbreaks of this magnitude occur. The urban forests in the two largest cities in Norway will fare worse than those in Michigan if a pest outbreak occurs on their linden trees. In Oslo and Bergen, 70% of the street trees planted were *Tilia x europaea* 'Pallida' (Sæbø et al., 2003). Such dominance of a single species predisposes the urban forest to potentially devastating effects from pest and disease outbreaks.

Urban parks and woodlots

Another option to promote urban biodiversity is to focus on urban parks and woodlots. Research has shown that the larger the park and/or woodlot size, the greater the species richness. After surveying 15 parks in Flanders, Cornelis and Hermy (2004) found that park area was the main factor explaining the variation in biodiversity indicators. Similarly, Godefroid and Koedam (2003b) found that one very large woodlot (1666 ha) in Brussels had greater species richness than 11 small woodlots (2–123 ha).

In some situations, forest stand management practices may be appropriate to increase biodiversity in urban woodlots. Incorporating silvicultural management systems lends scientific credibility to urban forest management and may improve management (von Gadow, 2002). One example is taken from Expo '70 Commemorative Park, Osaka, Japan. The forested park was planted 30 years ago and had been experiencing high stem density. Artificial gap creation and the incorporation of topsoil from a nearby forest increased floral diversity in the park (Nakamura et al., 2005). The artificial gaps increased light penetration and the topsoil served as a seed bank. Six species that were not found previously in the park, including *Cocculus trilobus* DC. and *Celastrus orbiculatus* Thunb., had germinated 1 year after the treatments (Nakamura et al., 2005).

A less intensive management option is the use of natural regeneration to promote biodiversity in urban parks. Natural regeneration of native species has been observed in the city of Christchurch, New Zealand (Stewart et al., 2004). Originally founded as a colonial English town, non-native European and North American species like *Quercus robur* L. and *Hedera helix* L. had been heavily planted. Yet over the past 20–30 years, a cultural awareness of indigenous New Zealand has emerged, and along with that, an increasingly common trend to plant native species. Natural regeneration of native species is occurring in the urban parks and gardens of Christchurch likely due to increased seed sources and less intensive management. Species such as *Coprosma robusta* Raoul and *Pittosporum tenuifolium* Gaertn. are becoming quite ubiquitous. If the most invasive non-native species are controlled, then there is potential for a gradual transformation to an urban forest dominated by native species (Stewart et al., 2004).

Natural regeneration has also shown promise in Finland in the understory of urban woodlots. After surveying 30 urban woodlots in Helsinki and Vantaa, Lehvävirta and Rita (2002) concluded that the number of saplings was sufficient to maintain forest continuity and was comparable to the number in rural woodlands. The authors do note that a larger proportion of saplings consisted of deciduous trees, including *Acer platanoides* L. and *Populus tremula* L., potentially leading to changes in mature forest composition (Lehvävirta and Rita, 2002).

Others have found that natural regeneration on abandoned, derelict sites – also known as natural colonization – does *not* lead to increased biodiversity. Millard (2000) studied naturally colonized sites in Leeds, England, and found that vegetation was dominated by only a few species, such as *Betula* sp. and *Salix* sp., and that growth rates were less than optimal. Seed bank establishment and soil improvement may be necessary on these derelict sites. Nevertheless, vegetation on derelict sites does provide environmental benefits such as amelioration of air pollution and reduction of water runoff (Millard, 2004).

Residential areas and people

Biodiversity should also be preserved and promoted in urban residential areas. Preserving trees during development projects can protect native biodiversity. In the long run, costs for construction and maintenance of green areas are minimized when tree preservation occurs (Florgård, 2000). Only vigorous, healthy trees should be selected for preservation because unhealthy trees may present a hazard and unnecessarily increase development costs.

Additionally, public education about biodiversity is crucial because the success of biodiversity conservation ultimately hinges on broad-based public support (Miller and Hobbs, 2002). The importance of biodiversity and its relevance to individual people's lives needs to be addressed. There is promise in community-based projects that foster an appreciation for the nature that is in city-dwellers' own backyards (Miller and Hobbs, 2002). Fostering a well-informed public may be the most important application of urban ecology (McKinney, 2002).

Conclusion

Promoting and preserving biodiversity within urban green-space is one way to decelerate the rapid rate of biodiversity loss. As our world becomes more and more urbanized, the urban forest will increasingly become an important reserve of biodiversity. We need to recognize the potential of urban areas to contain important amounts of biodiversity and work to promote that diversity.

In the future we will likely see increasing importance given to preserving and promoting biodiversity in the urban forest. City planners and urban foresters will have the opportunity to expand their traditional roles by incorporating a more ecological perspective into their management plans. Social as well as ecological benefits will be gained through biodiversity protection, such as increased tree health and greater aesthetic appreciation. New management options should be tested and incorporated into city plans, which will eventually lead to more sustainable and biologically rich urban forests.

References

Angermeier, P.L., 1994. Does biodiversity include artificial diversity? Conservation Biology 8, 600–602.

Araújo, M.B., 2003. The coincidence of people and biodiversity in Europe. Global Ecology and Biogeography 12, 5–12.

Balmford, A., Moore, J.L., Brooks, T., Burgess, N., Hansen, L.A., Williams, P., Rahbek, C., 2001. Conservation conflicts across Africa. Science 291, 2616–2619.

Colding, J., Elmqvist, T., Lundberg, J., Ahrné, K., Andersson, E., Barthel, S., Borgström, S., Duit, A., Ernstsson, H., Tengö, M., 2003. The Stockholm Urban Assessment (SUA-Sweden). Millennium Ecosystem Assessment Sub-Global Summary Report, Stockholm.

Cornelis, J., Hermy, M., 2004. Biodiversity relationships in urban and suburban parks in Flanders. Landscape and Urban Planning 69, 385–401.

Dwyer, J.F., Nowak, D.J., Noble, M.H., Sisinni, S.M., 2000. Connecting people with ecosystems in the 21st century, an assessment of our nation's urban forests. General Technical Report, PNW-GTR-490. USDA Forest Service, Pacific Northwest Research Station, Portland OR, p. 483.

Florgård, C., 2000. Long-term changes in indigenous vegetation preserved in urban areas. Landscape and Urban Planning 52, 101–116.

Godefroid, S., Koedam, N., 2003a. Distribution pattern of the flora in a peri-urban forest: an effect of the city–forest ecotone. Landscape and Urban Planning 65, 169–185.

Godefroid, S., Koedam, N., 2003b. How important are large vs. small forest remnants for the conservation of the woodland flora in an urban context? Global Ecology and Biogeography 12, 287–298.

Groombridge, B., Jenkins, M.D., 2002. World Atlas of Biodiversity: Earth's Living Resources in the 21st Century. University of California Press, Berkeley, CA.

Gustafsson, L., 2002. Presence and abundance of red-listed plant species in Swedish forests. Conservation Biology 16, 377–388.

Gyllin, M., Grahn, P., 2005. A semantic model for assessing the experience of urban biodiversity. Urban Forestry and Urban Greening 3, 149–161.

Jim, C.Y., Liu, H.T., 2001. Species diversity of three major urban forest types in Guangzhou City, China. Forest Ecology and Management 146, 99–114.

Kinzig, A.P., Warren, P., Martin, C., Hope, D., Katti, M., 2005. The effects of human socioeconomic status and cultural characteristics on urban patterns of biodiversity. Ecology and Society 10. Available from 〈http://www.ecologyandsociety.org/〉 (accessed September 2006).

Kühn, I., Brandl, R., Klotz, S., 2004. The flora of German cities is naturally species rich. Evolutionary Ecology Research 6, 749–764.

Lehvävirta, S., Rita, H., 2002. Natural regeneration of trees in urban woodlands. Journal of Vegetation Science 13, 57–66.

Löfvenhaft, K., Björn, C., Ihse, M., 2002. Biotope patterns in urban areas: a conceptual model integrating biodiversity issues in spatial planning. Landscape and Urban Planning 58, 223–240.

Luz de la Maza, C., Hernández, J., Bown, H., Rodríguez, M., Escobedo, F., 2002. Vegetation diversity in the Santiago de Chile urban ecosystem. Arboricultural Journal 26, 347–357.

McKinney, M.L., 2002. Urbanization, biodiversity, and conservation. BioScience 52, 883–890.

McKinney, M.L., 2006. Urbanization as a major cause of biotic homogenization. Biological Conservation 127, 247–260.

Millard, A., 2000. The potential role of natural colonization as a design tool for urban forestry – a pilot study. Landscape and Urban Planning 52, 173–179.

Millard, A., 2004. Indigenous and spontaneous vegetation: their relationship to urban development in the city of Leeds, UK. Urban Forestry and Urban Greening 3, 39–47.

Miller, J.R., Hobbs, R.J., 2002. Conservation where people live and work. Conservation Biology 16, 330–337.

Myers, N., 1996. The biodiversity crisis and the future of evolution. Environmentalist 16, 37–47.

Nakamura, A., Morimoto, Y., Mizutani, Y., 2005. Adaptive management approach to increasing the diversity of a 30-year-old planted forest in an urban area of Japan. Landscape and Urban Planning 70, 291–300.

Nowak, D.J., Walton, J.T., 2005. Projected urban growth (2000–2050) and its estimated impact on the US forest resource. Journal of Forestry 103, 383–389.

Poland, T.M., McCullough, D.G., 2006. Emerald ash borer: invasion of the urban forest and the threat to North America's ash resource. Journal of Forestry 104, 118–124.

Sax, D.F., Gaines, S.D., 2003. Species diversity: from global decreases to local increases. Trends in Ecology and Evolution 18, 561–566.

Sæbø, A., Benedikz, T., Randrup, T.B., 2003. Selection of trees for urban forestry in the Nordic countries. Urban Forestry and Urban Greening 2, 101–114.

Stewart, G.H., Ignatieva, M.E., Meurk, C.D., Earl, R.D., 2004. The re-emergence of indigenous forest in an urban environment, Christchurch, New Zealand. Urban Forestry and Urban Greening 2, 149–158.

Tait, C.J., Daniels, C.B., Hill, R.S., 2005. Changes in species assemblages within the Adelaide metropolitan area, Australia, 1836–2002. Ecological Applications 15, 346–359.

von Gadow, K., 2002. Adapting silvicultural management systems to urban forests. Urban Forestry and Urban Greening 1, 107–113.