



A habitat island approach to conserving birds in urban landscapes: case studies from southern and northern Europe

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Abstract. Wildlife conservation in urban habitats is increasingly important due to current urbanization trends. We review the different approaches to studying birds in urban landscapes, and point out the importance of the habitat island ecological theory as a research framework for the management and conservation of urban birds. Based on two comprehensive research projects conducted at urban parks in Spain (Madrid) and Finland (Oulu and Rovaniemi), several different issues related to bird conservation in cities are discussed, main findings of these projects are presented, and future research needs are suggested. Urban parks are important biodiversity hotspots in cities. Fragmentation conditions have the same deleterious effects to urban birds as in other fragmented landscapes. Park size accounts for species accumulation in urban parks; this pattern being highly nested. Urban parks of 10–35 ha would contain most of the species recorded in cities, but other indicators related to the probabilities of persistence of the target species should be obtained. Wooded streets can increase urban landscape connectivity by providing alternative habitat for feeding and nesting during the breeding season. Because increasing the size of parks is difficult in cities, enhancement of habitat diversity and resource availability for birds within parks (e.g. nest boxes, winter feeding tables, etc.) appears to be a straightforward way of increasing urban bird diversity. However, human disturbance (pedestrians) should be controlled since it can negatively influence many urban birds. We present a conceptual model for urban bird conservation, which includes three aspects (management, environmental education and research) and new alternatives to promote the involvement of different sectors of the society.

Key words: bird conservation, fragmentation, habitat island theory, urban birds, urban ecosystems

Introduction

Approximately 80% of the human population from industrialized countries is concentrated on urban areas (World Resources Institute 1996). In the next decade, urban sprawl will reach such a magnitude that several natural areas surrounding cities will give way to buildings and residential areas. Although some long-term efforts to understanding wildlife dynamics in cities are under way (Grimm et al. 2000), very little has been done in order to foretell the influence of urban expansion on wildlife and to develop management strategies aimed at diminishing these impacts (Hadidian et al.

1997). There is a need to study urban biodiversity and to include ecological knowledge in urban planning (Niemelä 1999a; Savard et al. 2000). There are several special features of urban ecosystems like mosaic phenomena, specific disturbance regimes, and the ‘heat island’ phenomena that are expected to influence the dynamics and structure of urban populations and communities (Rebele 1994).

Most studies on urban wildlife have focused on birds, and the information gathered up to now allows the comparison of different cities in relation to bird abundance and diversity patterns (e.g. Clergeau et al. 1998). At the community level, the study of urban birds has been approached from different perspectives (Table 1). The most used approach has been the application of the gradient paradigm, whereby environmental variation is assorted spatially, and these spatial patterns are supposed to affect the structure and function of ecological systems (McDonnell et al. 1997). Comparisons have generally included variations in community attributes (richness, diversity, evenness) between low and highly urbanized areas. Temporal changes in urbanization levels were successfully used to determine variations in urban bird populations in periods greater than 10 years. Other studies have tried to evaluate structural differences in communities (guilds, food consumption, etc.) between urban and non-urban settings. Another interesting approach, unfortunately not usually employed, included the comparison of urban communities at different latitudes and even in different continents, thereby improving our knowledge of the general processes that affect urban bird species.

One particularly useful approach has been the application of the habitat island ecological theory to urban landscapes (Table 1). In this paper, we refer to the island ecological theory in its broadest sense (see Doak and Scott Mills 1994; Whittaker

Table 1. Approaches to studying birds in urban habitats.

Approach	References
Urban gradients	Hohtola (1978), Gavareski (1976), Lancaster and Rees (1979), Cousins (1982), Sasvári and Moskät (1988), Jokimäki and Suhonen (1993), Blair (1996), Bolger et al. (1997), Rolando et al. (1997), Jokimäki (1992), Huhtalo and Järvinen (1977)
Temporal changes in urbanization levels	Batten (1972), Aldrich and Coffin (1980), Horn (1985), Jokimäki (1992)
Comparison between urban and non-urban communities	Tomialojc (1970), Emlen (1974), Tomialojc and Profus (1977), Beissinger and Osborne (1982)
Comparison between urban communities from different latitudes	Jokimäki et al. (1996)
Comparison between urban communities at different continents	Clergeau et al. (1998)
Habitat island approach	Lussenhop (1977), Davis and Glick (1978), Luniak (1983), Sasvári (1984), Vizyová (1986), Soulé et al. (1988), Suhonen and Jokimäki (1988), Natuhara and Imai (1999), Jokimäki (1999), Fernández-Juricic (2000c)

1998, 2000 and references therein), which encompasses a series of models that analyse the processes affecting species richness in islands and habitat islands. Urbanization generally leaves natural settings transformed into fragmented landscapes, with urban parks, gardens, etc., as the sole refugees for many bird species (Davis and Glick 1978). Urban parks are more isolated from their surroundings by an 'urban ocean' of buildings (urban matrix) than woodlots surrounded by clear cuts or farmland, but less isolated than real islands. Nevertheless, the overall difference between urban parks and their surroundings is quite sheer. The island approach allows focusing on the patterns and processes underlying bird distributions in urban parks as well as deriving a series of strategies for the management of urban birds (Adams and Dove 1989).

In the last years, comprehensive projects have been conducted in the cities of Madrid (Spain) (Fernández-Juricic and Tellería 1999, 2000; Fernández-Juricic 2000a–d, 2001a, b; Fernández-Juricic, in press), Rovaniemi (Finland) and Oulu (Finland) (Jokimäki 1992, 1996, 1999) to study urban birds under the umbrella of the habitat island theory. These projects have generated a great deal of information that can improve our knowledge of habitat selection processes of different bird species in urban settings. The goals of this paper are: (a) to resume the general findings of these projects, by re-analysing some results from published material; (b) to discuss relevant issues for the conservation of urban birds with the habitat island approach, stressing new research needs; and (c) to set up a conceptual framework for urban bird conservation, combining research, management, environmental education, and public participation.

Urban park birds in Madrid, Oulu, and Rovaniemi: main findings

In many studies of woodland bird communities, woodland area has been found to explain a high percentage of the variation in total species numbers (Opdam et al. 1985; van Dorp and Opdam 1987; Villard et al. 1999) or the occurrence of individual species in woodland fragments (van Dorp and Opdam 1987; Hinsley et al. 1995a, b). Bird species in Madrid appear to follow similar patterns of distribution to other fragmented environments (Saunders et al. 1991), with park area being the main factor accounting for the probabilities of park occupation at the community and individual species levels (Fernández-Juricic, in press, Figure 1). When considering only forest passerines in Madrid, park size also plays a significant role in the temporal distribution of species, with larger parks favouring species persistence and small parks having higher turnover rates (Fernández-Juricic, in press). Park area was also an important predictor of the number of species in Finland, explaining 39 and 68% of the variability in Oulu and Rovaniemi, respectively (Jokimäki 1999). Some species, mainly ground-nesters and typical forest species avoided small-sized urban parks (Suhonen and Jokimäki 1988; Jokimäki 1999). However, hole-nesting and deciduous forest species preferred urban parks in Finland. These relationships may be related to the area requirements of

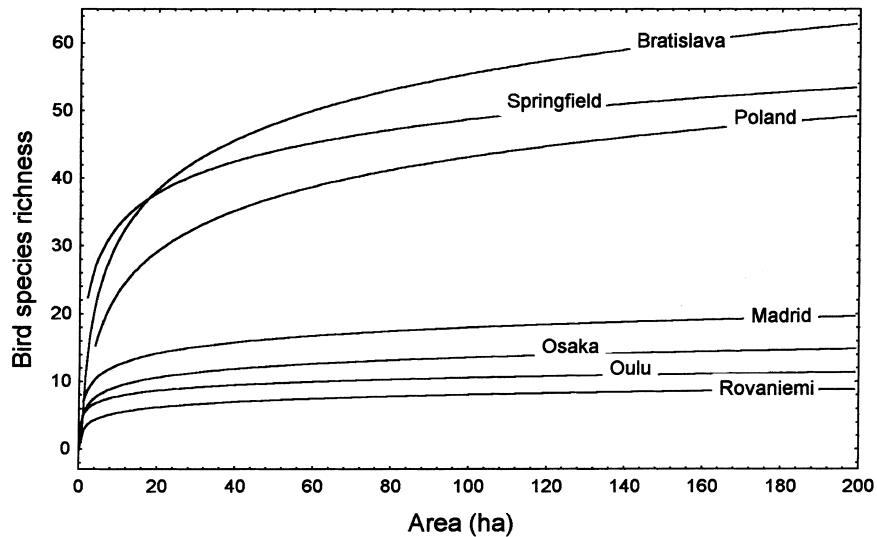


Figure 1. Species–area relationships for urban parks in Madrid (Spain), Oulu and Rovaniemi (Finland), Osaka (Japan), Springfield (USA), Bratislava (Slovakia), and several cities in Poland. For references see Table 1.

individual species (Hinsley et al. 1995a, b; Newton 1998), resource availability (e.g. food, nest sites, Suhonen and Jokimäki 1988), higher habitat diversity in large parks (Martin and Lepart 1989; Andrén 1994), nest predation (Jokimäki and Huhta 2000), and interspecific competition (Jokimäki 1999).

Patterns of species composition confirmed that park size also determines the accumulation of species in urban parks (Fernández-Juricic, in press). This pattern turned out to be nested; that is, small parks with n species are subsets of large parks with $n+1$ species (Patterson and Atmar 1986). In assessing the degree of nestedness among the three cities, we used the Nested Temperature Calculator (Atmar and Patterson 1993, 1995), which yields a measure of disorder in the community (T°), with a perfectly nested matrix having 0° , and a random one, 100° . The degree of nestedness was similar among Madrid ($T_{\text{obs}}^\circ = 16.26$, $T_{\text{rand}}^\circ = 64.05$, $SD = 4.05$, $P < 0.001$), Oulu ($T_{\text{obs}}^\circ = 18.01$, $T_{\text{rand}}^\circ = 52.32$, $SD = 4.55$, $P < 0.001$), and Rovaniemi ($T_{\text{obs}}^\circ = 16.13$, $T_{\text{rand}}^\circ = 46.5$, $SD = 5.52$, $P < 0.001$), suggesting similar patterns of species accumulation as park area increases in urban landscapes (see also Natuhara and Imai 1999). However, other factors also play relevant roles in shaping species distribution. Park age favours the colonization of new species, since old parks have a more complex habitat structure that allows species with specific habitat requirements (e.g. forest specialists) to make use of alternative substrates not found in young parks (Hohtola 1978; Fernández-Juricic 2000c). However, isolation effects appear not to exert any significant influence on species distributions at the community level, although they do influence the patterns of colonization of several individual species (Jokimäki 1999;

Fernández-Juricic 2000b). The lack of isolation effects at the community level probably stems from the low variation in isolation distances and the enhanced landscape connectivity provided by wooded streets (Fernández-Juricic 2000a).

In winter, urban areas may have more species and especially individuals than surrounding country sites or forests (Jokimäki et al. 1996), which may have to do with intensive winter feeding, especially in northern areas. Bird species commonly using feeding tables are more abundant in heavily urbanized areas than in less urbanized ones (Jokimäki and Suhonen 1998). Feeding tables located in urban parks may not only change bird density and species richness, but also species composition. Winter feeding may also help bird species to adapt to urban environments, as in the case of mallards (*Anas platyrhynchos*), greenfinches (*Carduelis chloris*) and blue tits (*Parus caeruleus*) in Finland (Pulliainen 1963; Jokimäki and Suhonen 1998).

Conservation of urban birds

General considerations

Conservation goals differ markedly as to the needs of people and the degree of modification of habitats. In the case of urban birds, there is a full panoply of conservation goals to be pursued (Savard et al. 2000), such as increase of bird diversity, diversity of ecological processes, overall genetic heterogeneity of urban bird populations, etc. However, in this paper, we focus on the increase of bird diversity in urban parks following the habitat island approach (Table 1). Studies of bird communities in urban parks have shown that parks are considerably richer in bird diversity than other urban habitats (Tilghman 1987; Jokimäki and Suhonen 1993; Hadidian et al. 1997). Therefore, it is especially important to understand the factors affecting the occurrence of birds in urban parks in order to maintain or even increase the diversity of birds in these fragments. This is a feasible alternative taking into account the economic and logistic resources available in many urban wildlife management units. Moreover, it also entails ecological and social benefits, since the number and abundance of species are good indicators of ecological conditions, and diverse communities can draw more attention from the general public, as sources of rare or even threatened species (Webb and Foster 1991; Rohde and Kendle 1994).

All the studies undertaken with the habitat island approach point out that fragmentation conditions exert the same deleterious effects to urban birds as in other fragmented landscapes (Table 1). For instance, in Madrid, Blackbird (*Turdus merula*) juvenile recruitment increases with park area as well as with the amount of protective cover (shrub cover and height). Large parks appear to provide more diverse habitats for Blackbirds to breed, and the amount of shrub cover can be used not only as substrate for reproduction but also as cover from predators (magpies) and disturbance (pedestrians) (Fernández-Juricic and Tellería 1999). The amount of vegetation cover

is especially important, since, for example, in northern Finland, artificial ground-nests located in urban parks suffer from high nest predation (46–57% of predated nests) in comparison with surrounding forest areas (Jokimäki and Huhta 2000). The habitat island approach can then be a first appraisal of the potential scenarios to direct conservation efforts.

Minimum park area for the occurrence of species in urban parks

That in several cities (Madrid, Rovaniemi, and Oulu—this study; Osaka–Naruhara and Imai 1999) the distribution of urban birds follows a nested accumulation of species suggests that large parks are more suitable than many small parks, since they harbour all the species present in small parks besides those that are only found in large ones (Patterson and Atmar 2000). To increase urban bird diversity, large parks may provide a large diversity of habitats necessary to hold many species with different habitat requirements. A classical approach to determine minimum park area comprises finding out the point upon which the accumulation of species levels off in the relationship $S = C + z \log A$ (S , species richness; A , habitat area; C and z , constants) (Figure 1). Comparatively, the minimum park area for Madrid ranges between 10–20 ha, very similar to that in Osaka, Rovaniemi, and Oulu, but slightly lower than that in Springfield (Tilghman 1987), Bratislava (Vizyová 1986), and some Poland cities (Luniak 1983) (Figure 1). Thus, the minimum park area for birds in these cities can range between 10 and 35 ha. This conservative estimate permits to incorporate a biological criterion (in this case, based on birds) for future design of suitable urban parks.

Nevertheless, minimum park area estimates based solely on species–area relationships may lack another important attribute of bird communities: its composition (the identities of species). Each bird species has its own habitat and area requirements. Atmar and Patterson (1993) put forward a new procedure to determine minimum fragment size that takes into account the probabilities of colonization and extinction of individual species. Using the Nested Temperature Calculator (Atmar and Patterson 1995), these authors estimate a diagonal line called the occurrence boundary threshold which separates the occupied from the unoccupied area of a presence/absence matrix (Atmar and Patterson 1993; Patterson and Atmar 2000; Figure 2). Provided species distribute following a nested pattern, then the probability of fragment occupation decreases from the top left to the bottom right corner of the matrix (Patterson and Atmar 2000). The area above 2 SD from the occurrence boundary threshold implies a 97–100% probabilities of occurrence of a species in a given fragment (Atmar and Patterson 1993). Therefore, the minimum area for the occurrence of a given species can be assessed as the position of each species within the matrix; in this case, the range of park areas that supports the species above 2 SD from the occurrence threshold (Figure 2).

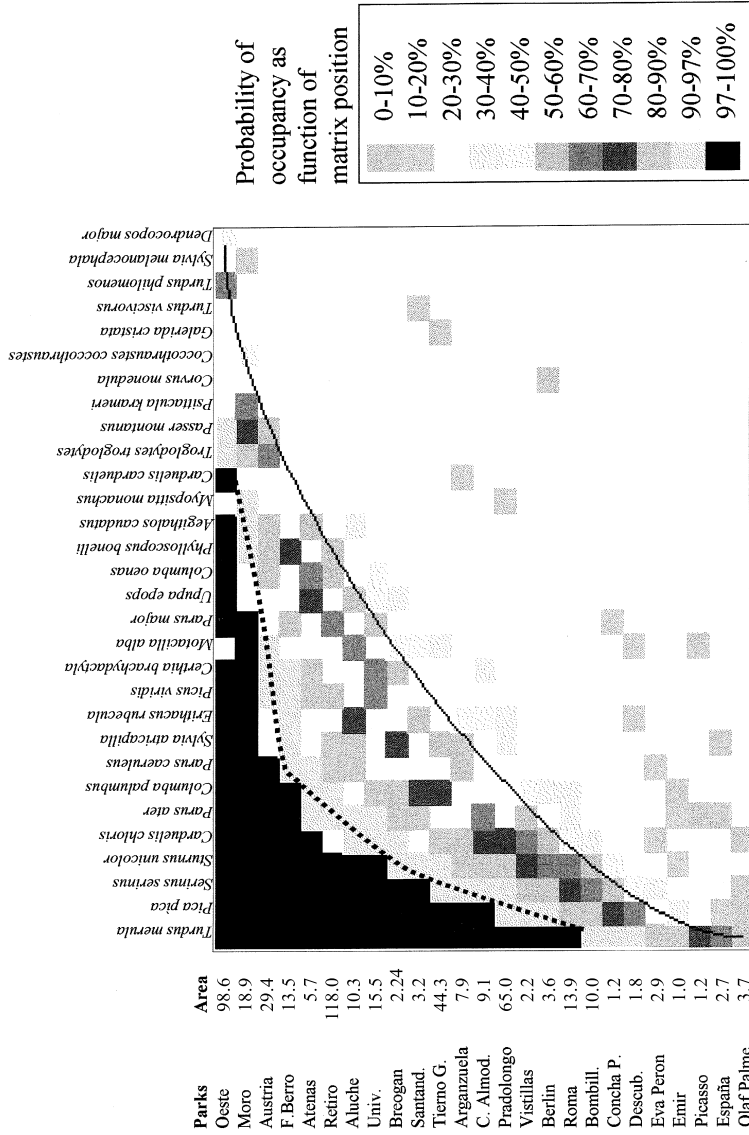


Figure 2. Determination of minimum area for species occurrence in Madrid. Presence/absence matrix, in which species and urban parks are ranked in decreasing order of species occurrence. Each box denotes the occurrence of a species in a given park. The lower line is the occurrence boundary threshold (see text). The dotted upper line shows 2 SD of the occurrence boundary, which indicates a 97–100% probability of species occurrence (Atmar and Patterson 1993). Also shown are the probabilities of species occurrence as a function of matrix position, taking into account that occurrence increases from the bottom right to the top left corner of the matrix (for the calculation of these probabilities see Atmar and Patterson 1993; Patterson and Atmar 2000).

For instance, in Madrid, the minimum area for a widely abundant species (*Turdus merula*) ranges between 2.2 and 3.2 ha (Figure 2). For other species (*Columba palumbus*, *Carduelis chloris*, *Parus caeruleus* and *Parus ater*) parks should have a minimum of 5.7–13.5 ha (Figure 2). This method also underscores a group of 10 species (in the upper-right corner of the matrix) whose probabilities of persistence in this landscape are <97–100%. Minimum area estimates may be especially valuable for the conservation of certain vulnerable species. However, park area in big cities is sometimes difficult to be enlarged, so in management perspective other alternatives are necessary; such as recovery plans aimed at species with restricted ranges (Morrison et al. 1992) as well as increasing habitat connectivity among high-quality parks (Adams and Dove 1989; Clergeau and Burel 1997). In addition to occurrence data, it is important to study reproductive success and mortality patterns of urban birds as well (Cowie and Hinsley 1987; Hůrak 1993; Fernández-Juricic and Tellería 1999).

Landscape connectivity

Connectivity is an important issue in urban landscapes (Adams and Dove 1989), since these habitats have very harsh limits between landscape elements, and the urban matrix may greatly restrain bird movement. There is some evidence that in cities, vegetation strips act as corridors for some birds (Clergeau and Burel 1997; Sodhi et al. 1999; Fernández-Juricic 2000a; Fernández-Juricic 2001b).

In Madrid, wooded streets proved to be efficient linear landscape elements to be used as corridors (Fernández-Juricic 2000a; Fernández-Juricic 2001b). Those wooded streets with complex habitat structure positively influence the number of species present within wooded streets, as well as species persistence, guild densities, and the probabilities of occupation by individual species (Fernández-Juricic 2000a). Thus, wooded streets allow certain species (particularly those feeding on the ground and breeding in trees or tree holes) to fare well by providing alternative habitats during the breeding season.

Wooded streets in Madrid also appear to enlarge park size and change its shape (Figure 3). Such area modification influences the process by which species enter into wooded streets (Fernández-Juricic 2001b). As population density increases in urban parks, their suitability appears to decrease due to intra- and inter-specific competition processes, and individuals begin to occupy wooded streets as alternative habitats in terms of foraging and breeding by a density-dependent habitat selection process (Fernández-Juricic 2001b) (Figure 3). Consequently, the implementation of high quality wooded streets with greater habitat complexity can be regarded as a powerful tool to augment the connectivity of urban habitats and increase their suitability for bird species (Savard et al. 2000). However, when networks of urban corridors are planned, it would be also important to preserve the existing diversity of native habitats surrounding corridors, which would eventually enhance the biodiversity value of corridors (Sodhi et al. 1999).

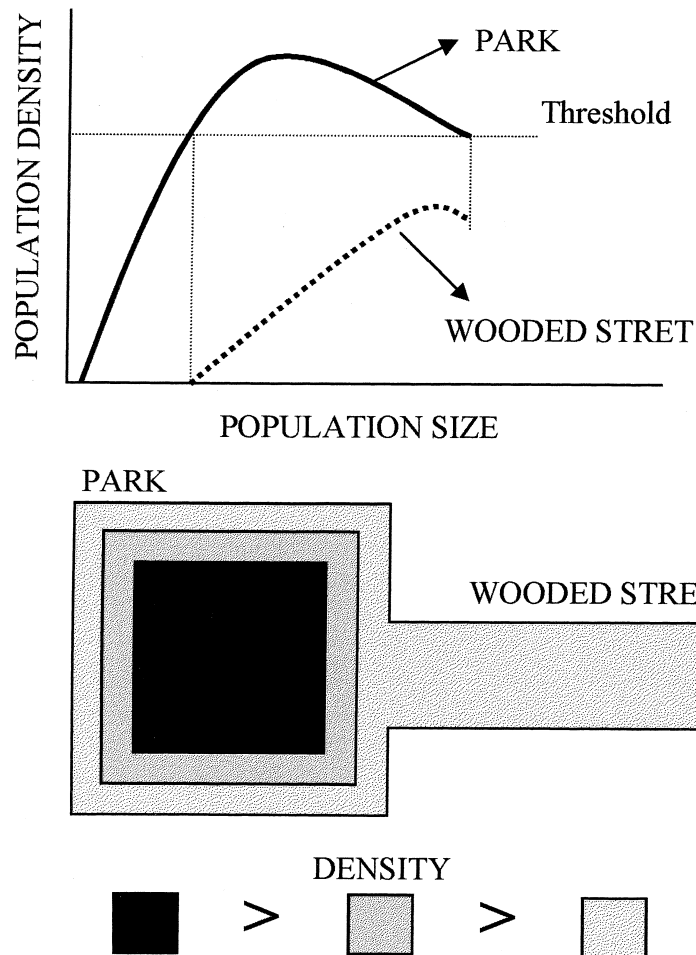


Figure 3. Density-dependent habitat selection of wooded streets (corridors) in Madrid. When abundance in urban parks rises above a certain threshold, individuals begin selecting wooded streets as alternative habitats for foraging or breeding. Based on Newton (1988), and Haddad and Baum (1999).

More research is required on the role of wooded streets as connectivity elements, particularly on how forest specialists use these landscape elements. In Madrid, although forest species were recorded in several wooded streets, they were not as common as generalist species (only 42% of forest species in urban parks were seen in wooded streets, compared to 83% of generalists, Fernández-Juricic 2000a). Such pattern raises some concerns as to whether wooded streets are useful for habitat specialists or simply for habitat generalists. On the one hand, it may be possible that habitat specialists might not fare well in urban corridors since their minimum habitat requirements are seldom reached. Conversely, habitat generalists could take advantage of corridor implementation, namely by filling hastily a broader variety of

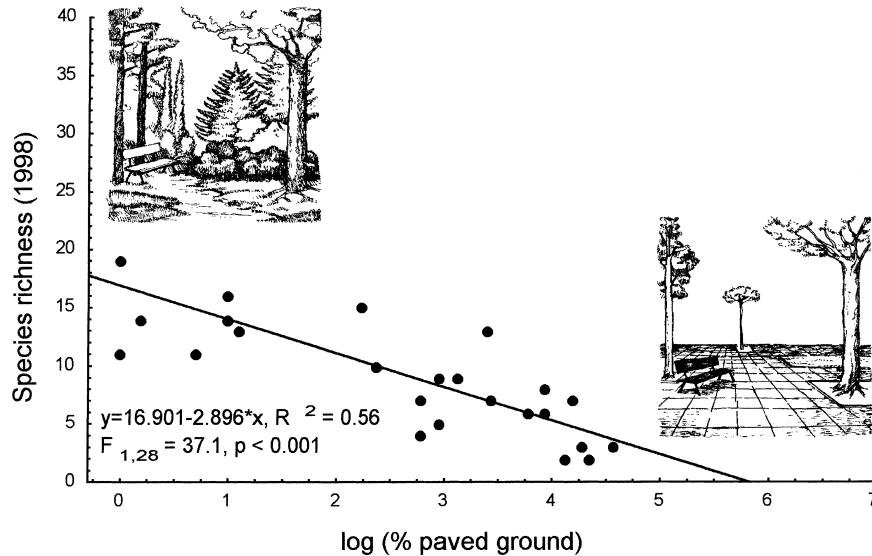


Figure 4. Bird species richness in Madrid is negatively associated with the amount of paved ground within urban parks. This pattern typifies two kinds of urban parks usually seen in several cities.

niches, decreasing habitat suitability to later colonizers by virtue of a preemptive use of sites (Newton 1998). With increasing amounts of built area within and surrounding corridors, higher abundance of human-associated bird species are expected in corridors (Sodhi et al. 1999). As a result, any attempt by habitat specialists to occupy the available habitat within corridors may turn unsuccessful because of inter-specific interactions, and could, in the end, restrain their landscape movements and increase isolation effects. On the other hand, an enlargement of a network of corridors could also intensify the presence of predators, such as Magpie *Pica pica* (Groom 1993), leading to increasing mortality and reproductive costs within corridors, and turning them into ecological traps. The relevance of both processes in urban landscapes requires proper evaluation concurrent with the goals of management programs.

Small parks could play a significant role in the connectivity of urban landscapes. As some authors rightfully pointed out (Lancaster and Rees 1979; Vízová 1986) the most important factor to increase species diversity in small parks is to enlarge their habitat complexity (vegetation structure in all layers). This management strategy could render small parks as high quality stepping stones (Rosenberg et al. 1997), which could be temporarily used by different species in their way through urban habitats.

Habitat structure

Generally, urban parks have lower total vegetation cover as compared to more natural areas, even lacking some vegetation layers at all (Erz 1966). Several studies

underscore the importance of preserving habitat complexity to increase bird diversity in urban parks (Lancaster and Rees 1979; Natuhara and Imai 1996; Jokimäki 1999; Savard et al. 2000). This appears to be a somewhat unpracticed lesson by urban wildlife units in some cities, albeit very effective. It is commonplace to hear people complain about the lack of enough green cover in urban parks and the superabundance of paved ground, when the latter clearly diminishes bird diversity (Figure 4) and increases bird abundance fluctuations (Lancaster and Rees 1979). Trees of different ages as well as multiple layers of vegetation are the most simple and direct tools to increase the suitability of urban parks due to a higher availability of food, shelter and breeding substrates (Leedy and Adams 1984). Primary habitats are quite valuable for bird conservation, since they allow the presence of certain native species without special adaptations for the urban life. Moreover, mixed species forests (deciduous and coniferous) enlarge bird diversity in urban settings more than single species forest stands (Thompson et al. 1993): more species with different habitat requirements can make use of a greater diversity in floristic composition.

The availability of different kinds of nest-boxes may increase the colonization of urban parks by a great variety of cavity-nesting birds (Jokimäki 1999). Moreover, permanent water sources are important to foster bird diversity (Tilghman 1987; Jokimäki 1992); they attract a group of rare species not usually seen in urban parks. For example, in Madrid, the Grey Wagtail (*Motacilla cinerea*), a species whose habitat is close to mountainous water streams, is lured in winter to urban parks with artificial streams and small ponds. Benign neglect would be one inexpensive management method of maintaining biodiversity in urban landscapes (Haila and Levins 1992). However, people and birds do not see habitats the same way. Semi-natural areas that attract many bird species are not necessarily liked by humans.

Research should focus on identifying those habitat features at different spatial and temporal scales that may attract rare or threatened species to cities (Savard et al. 2000), as a first step towards setting directions to managing landscape elements (urban parks, wooded streets, etc.). Moreover, it would also be necessary to understand the inter-specific relationships between rare and overabundant species with the current habitat structure, to find out key factors for controlling the latter.

Human disturbance

Besides fragment size, isolation, and habitat structure, urban bird species may also be affected by human disturbance (Blair 1996; Sauvajot et al. 1998; Fernández-Juricic and Tellería 2000; Fernández-Juricic 2000b). At a regional scale, the rate of visitors to urban parks in Madrid diminishes species richness as well as the temporal persistence of breeding pairs (Fernández-Juricic 2000b). Low disturbed parks have lower species turnover, suggesting that they were more suitable for enhancing site-fidelity of individuals that bred in previous years. In Oulu, recreation activity affects negatively breeding bird species richness, but not at the level of individual species (Jokimäki

1999). Perhaps the intensity of recreational use is lower in small-sized northern parks than in southern parks. Moreover, recently, Magpies, Hooded Crows (*Corvus corone cornix*) and Fieldfares (*Turdus pilaris*) have become more accustomed to human presence because of the lack of persecution of people (Jokimäki 1999). This is one of the main reasons why corvids have increased their numbers in urban landscapes.

In Madrid, disturbance effects are also apparent at lower scales (within parks), decreasing species richness and abundance in plots limited by trails as disturbance levels increased (Fernández-Juricic 2000b). Such disturbance effects also reduce the breeding densities of individual species in highly visited parks (Fernández-Juricic 2000b). Human disturbance has the same effects as predator disturbance (Gill et al. 1996). For instance, Blackbird's (*Turdus merula*) responses to visitors entail more time being vigilant and moving away from people, and less time searching for food, thus decreasing food intake (Fernández-Juricic and Tellería 2000). Individuals appear to trade-off their feeding opportunities, foraging as much as possible when disturbance levels decreased throughout the day.

The interplay of human disturbance in each park and its habitat structure also modifies tolerance levels of breeding species in Madrid (Fernández-Juricic et al. 2001). Flushing distances (the distance at which an individual flee from a visitor, Gutzwiller et al. 1998) decrease with higher habitat complexity, increase with species' size (total length), and augment in parks with low disturbance levels (Fernández-Juricic et al. 2001). Therefore, larger species are less tolerant to human presence, and these tolerance levels rise with the degree of habituation to visitors in each urban park. Such patterns of bird tolerance have relevant conservation implications. Flee distances of different species may serve to establish the minimal distance that a pedestrian may approach a bird before it is disturbed. These set-back distances may lead the movement of pedestrians within pathways with minimal disturbance to birds (Rodgers and Smith 1997). One alternative is to use tolerance levels of the largest species. In Madrid, set back distances of about 11 m (corresponding to Magpies) would avoid human perturbation for smaller species with shorter flee distances, such as Blackbirds (8.6 m), and Woodpigeons (9.9 m).

The pervasive effects of human disturbance at local and regional scales deserves a careful examination of ways for soothing impacts to bird species, but at the same time for maintaining or even increasing visitor rates. That urban parks are recreational areas should be borne in mind so that the protection of urban birds is accomplished without too many changes in visitor habits to circumvent people-wildlife conflicts. Future research on the effects of human disturbance on urban birds should be directed to the analyses of the relationships between human disturbance and tolerance levels for more skittish and rare species, which are usually the target of conservation efforts; the determination of visitors loads in urban parks according to species composition, habituation levels, and type of human activities; and the study of how the temporal dynamics of visitors (daily and seasonally) may influence bird species tolerance levels, population persistence, and breeding success.

Edge effects

Edge effects have been regarded as changes occurring at the transition between adjacent habitats (Saunders et al. 1991). Cities have several different kinds of edges, and total edge length is very high in urban landscapes (Rebele 1994). Therefore, edge effects may play an important role in urban landscapes. Some species are attracted to forest edges, which may function as 'ecological traps' (Gates and Gysel 1978), where certain species may have very low breeding success due to increased predation (see references in McCollin 1998). However, a recent study in the city of Sydney shows that artificial nest predation rates in edges do not differ significantly from that in core areas of urban parks (Matthews et al. 1999). The authors argue that this pattern may stem from the generalist requirements of bird species. In general, nest predation rate may be either higher or lower in urban parks than in surrounding areas depending on the general structure of habitats or the predator community (Jokimäki and Huhta 2000). As a result, negative edge effects due to increased nest predation at edges may be absent in some cities.

Even though most research has been focused on predation at edges, there are many causal factors that may bring about edge patterns, such as species-specific habitat requirements, biotic interactions, microclimate, habitat structure, etc. (McCollin 1998). Differences in bird composition and density in edge versus core habitats are supposed to reflect differences in food supply (Gates and Gysel 1978). In Madrid, large parks are particularly affected at their edges by car traffic and pedestrians, which raise noise and disturbance levels. People-tolerant species (House Sparrows *Passer domesticus* and Rock Doves *Columba livia*) are more common at edges, whereas less tolerant species are mainly encountered at core areas (Fernández-Juricic 2001a). Such bimodal pattern of habitat selection at edges point out the role of people as 'food sources', since many of the species that inhabit edge areas take advantage of human leftovers for food (Fernández-Juricic 2001a).

Differential density between edge and core areas may bring about differences in food selectivity (E. Fernández-Juricic, pers. obs.). Then, it may be expected that birds influence the spatial distribution of viable seeds with differential preference for native and introduced species, which could have important implications for native plant species. Such possibility certainly merits more research. Moreover, since edges may be rarely exploited by less-tolerant species, buffer zones could be useful to reduce disturbance effects (e.g. traffic noise, concerts, markets, dog-walking, etc.) on bird species.

Guild representation

Local rather than regional factors appear to play a significant role in the structure of urban communities (Davis and Glick 1978; Rebele 1994; Jokimaki et al. 1996;

Fernández-Juricic 2000c); that is, urban communities are composed of species not common in the surrounding landscape (Clergeau et al. 1998). In this case, the role of people becomes prevalent since there are many species highly adapted to human presence. In Madrid, House Sparrows and Rock Doves comprise an important guild in itself because its representation in the community (194 indiv./10 ha) is slightly higher than all other guilds (186 indiv./10 ha, E. Fernández-Juricic, pers. obs.). In Rovaniemi, the proportion of the House Sparrow and Rock Doves is over 40% of the breeding bird community (Jokimäki 1992). A similar pattern takes place in other European and North American cities (Savard et al. 2000). Both species are highly dependent on human activities as sources of food (e.g. refuse) and breeding substrates (e.g., buildings) (Rolando et al. 1997). Therefore, urban communities are usually characterized by the dominance of a few species (Huhtalo and Järvinen 1977; Beissinger and Osborne 1982; Jokimäki 1992; Rolando et al. 1997).

Even though in several cities throughout Great Britain there has been a significant decrease in the abundance of House Sparrows (Easterbrook 1999), the pattern of species dominance in many other cities poses several problems for urban planners (Lancaster and Rees 1979). Following the approach of increasing bird diversity in urban areas, a feasible goal would be to restrain the abundance of these species and favor rare ones. Habitat specialists (e.g. insectivores, old forest species, etc.) conform a group of species whose representation may be enhanced in urban habitats. For example, food provision may increase their recruitment, over wintering survival, breeding numbers, and reduce winter mortality (van Balen et al. 1987; Thompson et al. 1993; Jokimäki and Suhonen 1998). Nest box provision could also increase the availability of nest-sites and the abundance of these species in urban areas (Jokimäki 1999). However, in Madrid nest-boxes are utilized not only by small passerines but also by other species. Tree Sparrows (*Passer montanus*) take over these boxes, constructing nests that fill their whole capacity, thereby making them useless for the future breeding attempts of other species. Competition for nest-sites and the effectiveness of artificial habitat improvements should be experimentally assessed as to the target species before setting up any management measure.

A framework for future conservation efforts

People are a great resource to tackle conservation problems. This is particularly the case of urban landscapes, where people are the most prevalent biological element. In fact, urban landscapes represent a huge opportunity to increase public active participation in urban wildlife conservation as well as to promote conservation of other threatened habitats (Savard et al. 2000). Several interesting projects currently under way intend to incorporate social approaches in order to get an integrated understanding of urban ecosystems (Morgan Grove and Burch 1997; Pickett et al. 1997; Grimm et al. 2000).

We present a framework to encourage the links between conservation and public involvement (Figure 5), with the proximate goal of promoting urban bird conservation, and the ultimate purpose of improving the coexistence between people and birds in cities. This conceptual model is highly dynamic, in that it represents an interactive entity in perpetual change. People needs and inquiries about the wildlife surrounding them are the generator of such dynamic process. Another property of the model is its circularity, which means that the actions taken at any level would affect the other levels sequentially. As a result, the model can begin at any stage according to current conservation needs; in some cities social factors need to be worked out more thoroughly, whereas in others there is a need to begin simply by characterizing what species are present. For the sake of simplicity, we start at the management stage (Figure 5).

The conclusions obtained through the habitat island perspective can be regarded as a first approach to improving the status of urban birds. Bird management in cities can be conducted at two complementary levels: local and regional. For instance, locally (within parks), buffering human disturbance to increase park quality for birds, increasing the abundance of rare species by provisioning their main habitat requirements, etc. Regionally (urban landscape scale) management could be aimed at increasing habitat connectivity by implementing a network of wooded streets that soothe urban matrix effects, increasing the habitat complexity of small-medium parks to be used as stepping-stones, etc. Habitat restoration in urban areas is another

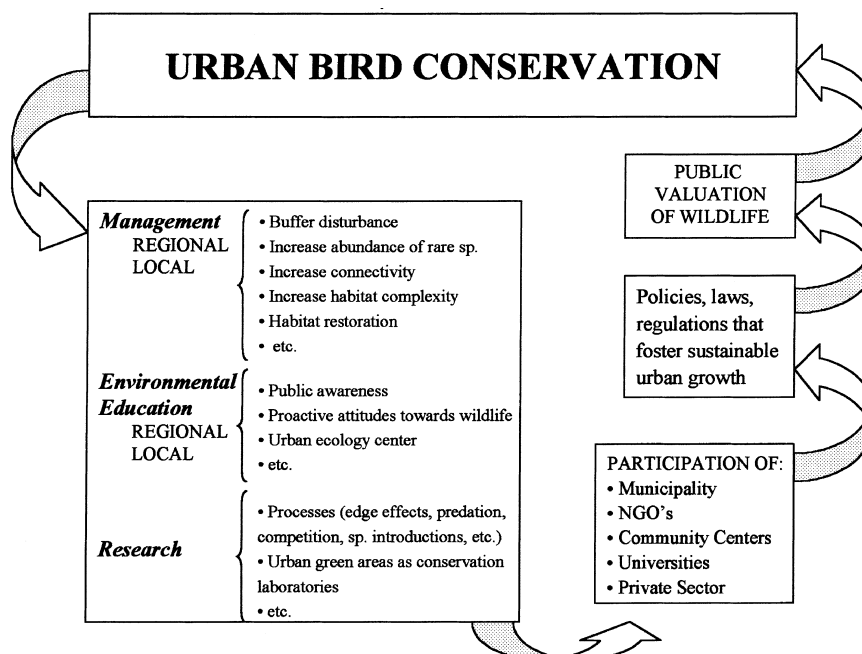


Figure 5. Conceptual model for improving bird conservation in cities. See details in text.

powerful tool that may be considered further. It is also an innovative way of involving people in the creation of suitable conditions for birds. One example comes from the selective plantation of native trees in urban areas, which are of benefit not only to birds (e.g. nesting cavities in mature trees) but also to people, because trees can enhance the value of property (Gilles 1999).

Management should be complemented by environmental education to increase public participation, and by research to get new information to correct and re-define management strategies. Environmental education approaches involving local (target public) and regional (indirect public) levels can tap people's resources more successfully (Fernández-Juricic 2000d). For example, if we are to encourage the protection of an endangered bird species in urban parks, we can focus the program on the neighbourhoods surrounding those parks in which the species is present (local). Meanwhile, the message can be handed out to the rest of the city (regional, e.g. down-town areas) so as to increase public awareness, embrace more people in proactive attitudes towards urban wildlife, and gain support to persuade local authorities. In this sense, several undertakings that link local and regional sectors of the society by means of the Internet are relevant examples of the power of new technologies oriented towards education and conservation purposes (e.g. The Nature Mapping Program; <http://www.fish.washington.edu/naturemapping/>; GLOBE, <http://globe.fsl.noaa.gov/>). Noteworthy is the role that teachers can play in such educational endeavours as a means to engage people's attention by cascade effects. The creation of urban ecology centres (as in many cities of the USA, England, China, Chile, etc.) can be an appropriate way of turning management, education and research into social priorities for the improvement of urban life quality.

Not only the general public are expected to participate in urban bird conservation; it is essential that other sectors of the society be represented to ensure success over the long run. That is the case of Municipalities, NGOs, community centers, and Universities, with which general agreements may increase the availability of logistic and economic support for management plans, not to mention the prospect of reaching consensus in controversial issues. Furthermore, the private sector can play another significant role within this framework, since it holds more economic power than public institutions and is usually keen on developing business with environmental roots (Shirley 1999). The wise lobbying with city-dwellers could end up in the development and approval of policies and laws that benefit both people and wildlife, and encourage the sustainable growth of urban areas.

Finally, an important step of this conceptual model is the assessment of how people perceive the surrounding wildlife. City-dwellers have come to undergo the "extinction of experience" (Goode 1990), which is simply a disassociation from nature in urban areas owing to the prevalence of urbanization and technological improvements that steer people's interest away from the wildlife dimension. Encouraging urban bird conservation is just one means to make people enjoy the many benefits of urban wildlife (Vandruff et al. 1995). But, how do people regard urban wildlife?

Do they feel awkward living with a few superabundant species that in some cases produce uncomfortable effects or do they prefer a city in which a great many species can be seen (or heard), thereby approaching them to the idea of inhabiting a rather natural area? The wildlife acceptance capacity may be a useful concept; it assesses the maximum population level for a certain species that is acceptable to people (WAC, Decker 1990). Given a particular management goal, the WAC can predict the sources of dissatisfaction among stakeholders. Thus, it provides a valuable source of information upon which to base environmental education efforts and management undertakings for soothing public differences. Once public valuation is incorporated into the model, its dynamic nature allows its own persistence due to the continuous inputs of and feedback from the different stages (Figure 5).

Conclusions

Because patterns of bird habitat use in cities have received a general characterization (Table 1), it is deemed advisable that research towards urban bird conservation focus on processes. For example, how edge effects modify habitat selection/reproductive output of low abundant species; how predation modify behavioral and population-level responses of prey; how disturbance and predation interact to allow the occurrence of low-tolerant species in highly visited urban parks; how superabundant species may increase competition for nest-cavities, reducing breeding densities of less abundant species (mostly forest passerines). An habitat island approach may be a good starting point for bird management in urban landscapes.

The prospect of regarding urban parks as natural laboratories for ecological research was set forth several decades ago (Erz 1966; McDonnell and Pickett 1990); nevertheless, little has been done in this direction mainly because ecologists have undermined the relevance of ecological studies conducted at urban centers. As other authors have pointed out, ecological patterns and processes in urban areas are similar to those of other natural settings (Sukopp and Numata 1995; but see Rebele 1994; Niemelä 1999b). Therefore, cities may also be of value in research oriented towards conservation. Urban parks can be thought of as laboratories for testing management goals and techniques as hypothesis that be confirmed or falsified. Many issues that may be difficult to test in natural grounds due to logistic reasons or just uncertainty, may well be suited for urban parks.

Bird conservation in urban and sub-urban habitats is no longer a fruitless task. Theoretical and empirical developments for understanding the functioning of wildlife in cities, along with the increasing interest placed on urban habitats as reservoirs of wildlife, open up new perspectives to direct conservation efforts with active public involvement. It is time for action and for offering urban people the possibility to learn to live in close proximity to their natural environment. Such endeavor will certainly benefit human well-being and wildlife conservation in general.

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