

# Chapter 18

## Ecological and Social Factors Determining the Diversity of Birds in Residential Yards and Gardens

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*I value my garden more for being full of blackbirds than of cherries, and very frankly give them fruit for their songs.*  
(Joseph Addison, English essayist, poet, and politician, 1672–1719)

**Abstract** Residential landscapes with private yards and gardens are a major land cover in many cities, represent a considerable opportunity for bird conservation and enhance human experiences with wildlife. The number of studies of birds in residential landscapes is increasing worldwide, but a global-scale perspective on this research is lacking. Here we review the research conducted on birds in residential settings to explore how birds respond to this novel habitat and how private gardens can be designed and managed to enhance their value for bird populations and for human well-being. We examine the key ecological and social drivers that influence birds and draw particular attention to the importance of scale, the role of bird feeding, the predation risk from cats and the relationship between native vegetation and bird diversity. The success of bird conservation initiatives in residential landscapes hinges on collaboration between a range of stakeholders, and we conclude the chapter by making recommendations for urban planners and evaluating policy tools for incentivising householders and communities to conserve birds in their neighbourhoods.

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## 18.1 The Garden Resource

Although we may not all articulate it as eloquently as Addison, people have long sought to cater for birds in their gardens. Indeed, of all the places that we encounter birds in cities, it is in our private gardens and yards (hereafter gardens) that these interactions are often most meaningful. It is here that many of us spend vast amounts of time and money to interact with birds by providing vegetation, food, water or nest sites. Ultimately, whether birds or people (or both) benefit from these activities remains an active area of research, but when this resource provision is scaled up across neighbourhoods, it becomes clear that the collective impact of private gardens in our towns and cities is substantial (Goddard et al. 2010; Lerman and Warren 2011; Belaïre et al. 2014). This is especially true in many developed countries where private gardens are often a major urban land cover. For instance, an estimated 87% of UK householders have domestic gardens (Davies et al. 2009). Collectively, these gardens cover approximately 25% of the land area of UK cities, which equates to 35–47% of the total urban green space (Loram et al. 2007). Elsewhere, gardens are even more plentiful; in Dunedin, New Zealand, the vegetated garden area comprises over one third of the total urban area (Mathieu et al. 2007). Although they may differ in their management and form, gardens can also be important components of cities in the global south, e.g. residential gardens in Chile (Reyes-Paecke and Meza 2012), urban home gardens in Brazil (Akinnesi et al. 2010) and urban ‘patios’ in Nicaragua (Gonzalez-Garcia and Sal 2008), and their importance is likely to increase further as global urbanisation continues at a rapid pace (Fragkias et al. 2013).

In addition to the extent of gardens across our cities, the management of these residential habitats plays a significant role in their contribution to bird conservation. Over three quarters of US householders participate in some form of gardening (Clayton 2007) and, in 2013, spent an estimated \$34.9 billion on gardening supplies and activities (National Gardening Survey 2014). Moreover, 13.4 million Americans maintain vegetation or natural areas specifically for wildlife (US Government 2011). Birds are often the main target of such wildlife-friendly management, as evidenced by the popularity of bird feeding and nest box provision compared to other wildlife gardening activities (Davies et al. 2009; Goddard et al. 2013).

The upshot of the large number of gardens across cities and their potential bird-friendly management is that private gardens can have considerable ‘direct’ and ‘indirect’ value for bird conservation (Cannon 1999). On the one hand, residential landscapes within cities provide a direct habitat resource for birds. Early investigations in the USA revealed that residential areas were comparable in bird diversity to the natural habitats on which they were developed (Guthrie 1974; Emlen 1974). Residential developments in the UK have also been shown to have greater value than farmland for thrushes (*Turdus* spp.) as agricultural intensification depletes the

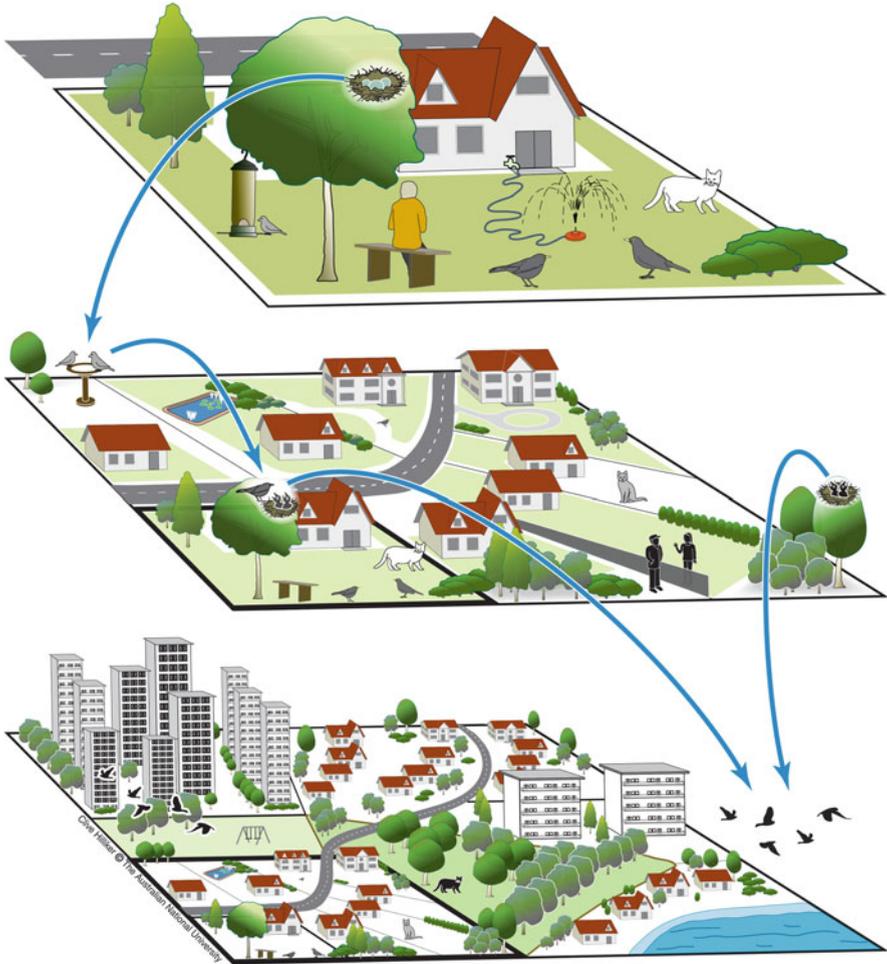
quality of rural habitats (Mason 2000). Urban-rural gradient studies tend to show that dense urban centres have a homogenising effect on bird communities (Clergeau et al. 2006), but intermediate levels of urban development, characterised by residential gardens, are often associated with peaks in bird richness or abundance, where so-called urban adapter species thrive and mix with a few ‘urban avoiders’ and ‘urban exploiters’ (Blair 1996; Sewell and Catterall 1998; Blair 2004; Tratalos et al. 2007). Although residential landscapes may not support many habitat specialists, they can be important for species of conservation concern, such as house sparrows (*Passer domesticus*) and starlings (*Sturnus vulgaris*) that are declining in the UK (Fuller et al. 2009). Moreover, gardens certainly contribute to the conservation of common birds which appear to be declining more rapidly than rarer species in Europe (Inger et al. 2015).

In parallel with their direct significance for birds, gardens also have indirect conservation value by increasing people’s engagement with birds and fostering a commitment to nature conservation more widely (Cannon 1999). When householders participate in citizen science bird monitoring programmes, it often leads to important data for fundamental scientific research, such as how climate change affects bird communities (Princé and Zuckerberg 2015). Participation also empowers the householder by providing tangible benefits for people, such as improved sense of place (Evans et al. 2005). Evidence is also growing that interacting with wildlife is beneficial to human health. For instance, when controlling for variation in demographic characteristics, the species richness and abundance of birds in southeastern Australian neighbourhoods was positively related to human well-being (Luck et al. 2011).

In the rest of this chapter, we develop the conceptual framework of Goddard et al. (2010) by viewing gardens as socioecological constructs that can influence birds at multiple spatial scales (Fig. 18.1). Our geographic scope is global, but we necessarily focus on the UK, the USA and Australasia as there is a research bias towards these countries. In Sect. 18.2, we discuss the ecological drivers of bird diversity at garden to landscape scales. We then explore the human dimension in Sect. 18.3, highlighting garden management decisions that impact birds (i.e. landscaping, bird feeding and cat ownership) and their underlying socio-economic drivers. In Sect. 18.4, we turn our attention to urban policy, planning and tools for incentivising and engaging householders and other stakeholders in bird conservation initiatives. We conclude in Sect. 18.5 by highlighting key knowledge gaps and research priorities. Addressing how private gardens can be better designed and managed to strengthen their potential for bird conservation and human well-being will greatly benefit urban planning and policy.

## 18.2 Ecological Drivers of Bird Diversity

Ecological drivers of garden bird diversity range from the scale of an individual feature (e.g. a tree or a shrub) through to that of an entire city (Goddard et al. 2010). These scales match different events in the life histories of birds, from the



**Fig. 18.1** Ecological and social drivers of garden birds at nested spatial scales. At the household scale (*top panel*), individual habitat features (e.g. trees and shrubs) and householder decisions (e.g. to provide supplementary food, to irrigate the garden and have an outdoor cat) affect the ability of birds to find suitable forage and nest sites. At the neighbourhood scale (*middle panel*), the extent, composition and configuration of habitat in groups of adjacent gardens are dictated by variation in socio-economic status and social norms in gardening practices. At the city scale (*lower panel*), the clustering of residential habitat with other urban green spaces reflects urban planning and green infrastructure strategies. *Blue arrows* represent the different life stages of garden birds at different scales in the garden hierarchy, from eggs in an individual nest at the garden scale (*top panel*) to nestlings in multiple gardens (*middle panel*) and fledglings dispersing within and beyond the city (*lower panel*)

consumption of a berry, to the placement of a nest for the season and the interactions of birds within a population or with birds of other species (Fig. 18.1). It is important to note that whilst all birds require habitat to forage, nest, shelter and interact, what constitutes good habitat can vary substantially between bird species,

as does the scale in which an individual species uses that habitat (Hostetler and Holling 2000). For example, cactus wrens (*Campylorhynchus brunneicapillus*) use habitat features at the scale of the individual property, whilst Lesser Goldfinches (*Spinus psaltria*) respond to habitat features at the neighbourhood scale, and Gila woodpeckers (*Melanerpes uropygialis*) are influenced by features from a combination of scales (McCaffrey and Mannan 2012). The ecological drivers of garden bird diversity, furthermore, vary with changing climatic and geographical context. Cultivated gardens with a year-round supply of leafy plants will be perceived by birds in tropical regions much differently to their desert cousins. In this section, we address each scale of ecological drivers in turn and discuss their relevance for different birds and for different geographic contexts.

### 18.2.1 Garden-Scale Ecological Drivers

Gardens are landscaped with individual vegetation and structural features, such as trees, shrubs, grasses, rocks and water features. These garden-scale features directly link to the availability of habitat resources for birds and are important for shaping urban bird communities (Daniels and Kirkpatrick 2006; Evans et al. 2009; van Heezik et al. 2008; Chamberlain et al. 2004; Parsons et al. 2006). Gardens, however, exhibit individual variability, whereby different styles of gardens emphasise different habitat features with ensuing effects to the bird community. For example, van Heezik et al. (2008) found that gardens in New Zealand with at least 30% open-vegetated space supported more specialist native bird species, including species that were virtually absent in gardens with minimal vegetation.

Cultivated gardens can be ‘oases’ of otherwise scarce resources, for example, as sources of water in dry climates (Bock et al. 2008). However, polarisation between intensively managed exotic vegetation and more ‘natural’ plant communities, especially in urban areas with relatively short histories, can result in decreasing habitat resources for the native bird community (but see Gleditsch 2016). For example, studies from Phoenix, USA, have shown that gardens landscaped with native desert plants contain more abundant foraging resources and have lower foraging costs, as well as a bird community less dominated by aggressive urban birds and more desert bird species, than gardens landscaped with exotic plants (Lerman et al. 2012b; Lerman and Warren 2011). Similarly, in Hobart, Australia, gardens with native plants had significantly more native birds (Daniels and Kirkpatrick 2006). When comparing gardens landscaped with native plants with gardens landscaped with exotic vegetation, Burghardt et al. (2009) found that the native gardens supported a greater abundance and richness of caterpillars and, subsequently, birds, since many species rely on this food source for feeding nestlings. Native gardens were especially important for bird species of conservation concern, supporting eight times the abundance of these species compared with exotic gardens. Interestingly, there is much less evidence for a positive association between native plants and birds in the UK or Europe, where urbanisation has a longer history.

Many birds, including those sensitive to urban landscapes, benefit from vegetation structural complexity, which can be achieved by planting out garden beds with dense shrubs, allowing leaf litter to accumulate under trees, choosing tussock or meadow grasses over lawn or leaving areas of lawn unmown. For instance, in Canberra, Australia, urban-sensitive native species were more likely to be recorded in front gardens characterised by trees, shrubs and ground covers compared with front gardens dominated by lawns (Ikin et al. 2013b). The capacity of gardens to contain these diverse habitat features is positively related to garden size. Smith et al. (2005), for example, found that larger gardens in the UK were more likely to have large trees and a higher diversity and occurrence of vegetation than smaller gardens. Kaoma and Shackleton (2014) likewise found that larger gardens in South Africa had a higher density of trees. It is concerning, therefore, that the current trend in housing development is for smaller gardens that require low maintenance and lack diverse habitat resources (van Heezik et al. 2008; Gaston et al. 2005; van Heezik and Adams 2014).

It is worth noting that the design and management of gardens can result in perverse outcomes for the bird community by leading to the overabundance of some bird species that might have negative consequences for other species. For example, in Australia, nectar-rich cultivars of native flowering shrubs are promoted as being “bird friendly” by nurseries, and people plant them to entice birds to their gardens. However, studies have shown that these flowering shrubs attract aggressive Australian honeyeaters that exclude smaller birds, many of which are declining in urban landscapes (Parsons et al. 2006; French et al. 2005). Further, Davis et al. (2014) showed that these shrubs also provided a year-round source of nectar within suburban areas compared with native forests, supporting high densities of large nectar-feeding parrots whose dominance then enables them to outcompete other species for nesting resources (Davis et al. 2013).

Compared with research focused on other urban green spaces, there have been relatively few studies of bird habitat use in gardens. These studies have had contrasting findings about the relative importance of local garden-scale variables compared with those measured in the surrounding landscape. For example, Lerman and Warren (2011) found that front garden habitat, particularly the presence of desert trees and shrubs, was twice as important as regional landscape features for explaining variation in the urban bird community of Phoenix, USA. In comparison, French et al. (2005) found that the abundance of nectivorous species in gardens within Sydney, Australia, was better predicted by the vegetation characteristics of adjoining and more distant gardens than the vegetation characteristics of a single garden. In the next section, we explore the influence of neighbourhood and city-scale ecological drivers on garden birds.

## 18.2.2 *Neighbourhood and City-Scale Ecological Drivers*

Birds are highly mobile and most species use the landscape at multiple scales, including scales larger than an individual garden (Hostetler and Holling 2000; Hostetler and Knowles-Yanez 2003, Litteral and Shochat 2016; but see Daniels and Kirkpatrick 2016). This leads to scale mismatch, whereby the capacity of gardens to provide habitat for birds is constrained or enhanced by the suitability of habitat in the surrounding landscape (Goddard et al. 2010; Warren et al. 2008; Catterall 2004). For example, Gambel's quail (*Callipepla gambelii*) needs areas of desert vegetation larger than the average garden, but when a garden adjoins parkland or a group of gardens are planted with desert vegetation, the cumulative effect could provide sufficient habitat (Hostetler and Knowles-Yanez 2003). Recent research has demonstrated that whilst local-scale factors may well be most important, acknowledging the contribution of landscape-scale attributes can improve our understanding of birds in urban environments (Melles et al. 2003; MacGregor-Fors and Ortega-Álvarez 2011; Suarez-Rubio and Thomlinson 2009; Donnelly and Marzluff 2004).

The collective attributes of neighbouring gardens represent important drivers of garden bird diversity. Belaire et al. (2014), for example, investigating birds within groups of gardens in the Chicago region, USA, found that the richness of native birds was strongly related to the aggregated wildlife value of the gardens (e.g. the presence of trees, plants with fruits or berries and mix of evergreen and deciduous trees). In contrast, the vegetated area and canopy cover within the neighbourhood and amount of green space within the wider landscape provided a weaker explanation of native bird richness within gardens. Chamberlain et al. (2004), in comparison, found that the occurrence of garden birds across the UK was strongly influenced by neighbourhood attributes, such as the presence of brownfield and green space areas.

Neighbourhood attributes influence local-scale garden characteristics and landscape-scale habitat suitability (Catterall 2004). For example, gardens in the inner city (i.e. located in areas of high housing density) support different bird assemblages to gardens in exurban developments (Kluza et al. 2000; Germaine et al. 1998). Several studies have found higher bird diversity when residential streets were planted with native trees (Ikin et al. 2013b; White et al. 2005; Young et al. 2007). For example, native forest bird species richness within residential landscapes of Seattle, USA, was positively related to tree density in the landscape (Donnelly and Marzluff 2006). These positive relationships suggest that tree cover could be used as a surrogate measure of available bird habitat. However, caution is needed when using tree cover as a proxy of habitat for all species. Urban-sensitive birds in Canberra, Australia, for example, had no relationship with tree cover but instead were related to habitat structural complexity, i.e. diversity of strata, including shrubs and ground layer (Ikin et al. 2013b). How cities are planned and developed can also strongly influence the habitat values of gardens for birds (see Sect. 18.4.2).

## 18.3 Social Drivers of Garden Bird Richness and Abundance

### 18.3.1 *Socio-economic Status*

In recent decades, urban ecology has emerged as an important discipline that integrates natural and social sciences in order to understand what drives ecological functioning in and around towns and cities (Tanner et al. 2014; Grimm et al. 2008). Perhaps more so than any other urban habitat, private gardens can be seen as ‘hybrid’ creations (Power 2005) or ‘socioecological constructs’ (Goddard et al. 2010) such that understanding residential landscapes demands an interdisciplinary approach (Cook et al. 2011). A central tenet of urban ecology is the association between human social stratification and urban biodiversity (Warren et al. 2010), and studies of urban vegetation have consistently found that wealthier neighbourhoods support greater levels of vegetation cover or higher plant diversity (e.g. Hope et al. 2003; Martin et al. 2004; Lubbe et al. 2010; but see: Meléndez-Ackerman et al. 2014). There are a number of putative mechanisms for this so-called luxury effect, but given the cost of planting trees and shrubs, economics can certainly hinder people’s choices for adding to or replacing existing vegetation to their property. Financial constraints can be particularly prohibitive for renters who are at the mercy of their landlord’s landscaping decisions (Lerman and Warren 2011). On the other hand, the luxury effect suggests an avenue of outreach to work with higher income areas to landscape more sustainably and with wildlife in mind, given they have the financial capacity to do so. Interestingly, Kirkpatrick et al. (2012) show that household income can also predict attitudes towards vegetation, with wealthier people more likely to value trees (see also Sect. 18.3.2.1).

The human-dictated plant community can be seen to form a ‘template’, and it has been suggested that birds and other higher taxa are indirectly influenced by social factors via this plant template (Faeth et al. 2011; Luck et al. 2012). As predicted, a positive correlation between bird diversity and neighbourhood income (or related measure of socio-economic status) has been demonstrated in North America (Lerman and Warren 2011; Kinzig et al. 2005; Melles 2005), Australia (Luck et al. 2012) and Europe (Strohbach et al. 2009). These findings have led to concerns about environmental injustice, with lower-income neighbourhoods exposed to lower native bird diversity (Lerman and Warren 2011). However, in the gardens of Dunedin, New Zealand, van Heezik et al. (2013) found only a weak positive association between house value and bird richness, whilst in Chicago, USA, Loss et al. (2009) document an inverse relationship between neighbourhood income and native bird richness. These counter examples suggest that more work is needed before we can generalise findings to cities in different regions and with different development histories (Warren et al. 2010). Other measures of socio-economic status, such as education level or occupation, as well as demographic factors, such as householder age, can also predict the provision of bird-friendly garden habitat

(e.g. Goddard et al. 2013; Lepczyk et al. 2004b). These patterns can be complicated by the legacies of past management due to the maturation of vegetation over time (Luck et al. 2009). For instance, Boone et al. (2010) found that tree coverage in Baltimore, USA, in 1999 was better explained by the 1960 demographics than by 2000 demographics.

There are a number of other confounding spatio-temporal factors that obscure the association between socio-economic status and bird diversity. For example, neighbourhood house prices are positively associated with proximity to urban green spaces and natural habitats (e.g. Morancho 2003; Bolitzer and Netusil 2000; Gibbons et al. 2014); avian diversity also tends to increase with greater green space provision in the wider landscape (Sect. 18.2.2). A number of studies have found that bird species richness increase with neighbourhood age, a pattern that is typically explained by the maturation of vegetation (Munyenembe et al. 1989; Palomino and Carrascal 2006). In contrast, Mason (2006) found no relationship between avian richness or abundance and housing development age in a small English town, whilst Loss et al. (2009) document higher species richness in newer developments in Chicago because they tend to support *more* natural habitat than older suburbs. These findings underline the fact that the provision of vegetation in residential landscapes is key to supporting avian diversity, even though the factors predicting the extent of vegetation may vary with urban context. Individual bird species may also respond differently to variation in habitat structure that correlates with socio-economic status. For example, Shaw et al. (2008) show that house sparrow (*Passer domesticus*) decline in UK cities has been more prevalent in affluent neighbourhoods that offer reduced nesting and feeding opportunities.

### **18.3.2 Individual Behaviour, Attitudes and Beliefs**

Householders undertake a variety of actions to encourage birds in their gardens, including providing food and water, installing bird boxes and planting or maintaining vegetation (Lepczyk et al. 2004b; Davies et al. 2009). A primary motivation for these gardening choices may be to observe or protect nature (Clayton 2007; Goddard et al. 2013). However, for many households, their decisions are driven by other underlying factors, including social norms and financial constraints, which might pose as barriers for widespread participation in wildlife gardening (Goddard et al. 2013). This section highlights how individual behaviour, attitudes and beliefs interact with gardening decisions, the extent of these choices and the consequences of these decisions on people and garden birds. We focus initially on planting vegetation and bird feeding since the majority of research addresses these two wildlife garden activities, but we also highlight the considerable threat provided by domestic cats in residential landscapes.

### 18.3.2.1 Vegetation and Landscaping

Gardening decisions are underpinned by a series of value judgements that influence the habitat quality of gardens for birds. Gardens are often viewed as an extension of the home, blurring the line between ‘wild’ and ‘domestic’ (Clayton 2007). Therefore, householders have the urge to tidy-up the garden space, which might be at odds with supporting wildlife (Lerman et al. 2012a). An ‘ecology of prestige’ explains this phenomena whereby a householder places importance on maintaining gardens to a standard that adheres to neighbourhood expectations, as well as reflecting aspirational social goals (Larsen and Harlan 2006; Grove et al. 2006; Kurz and Baudains 2012). Implications for this include that the homeowner landscapes with their neighbours in mind rather than for wildlife, even though they may have pro-environmental intentions (Nassauer et al. 2009; Goddard et al. 2013). Householders can also be constrained by the legacy of previous owners; most homeowners inherit a garden and thus what exists does not necessarily reflect their taste or choices for landscaping (Larsen and Harlan 2006; van Heezik et al. 2013). This can detract from the garden as “an expression of me” (Freeman et al. 2012) and the intrinsic value of gardens in providing a connection with the natural world.

The influence of social norms on gardening decisions highlights the need to encourage wildlife gardening at the neighbourhood scale (Goddard et al. 2010; Cooper et al. 2007; Warren et al. 2008). Neighbourhood associations, incentive programmes and greater communication by conservationists can help achieve these goals (Belaire et al. 2014). For example, one key message is that specific plant choices (e.g. native vs. non-native) have cascading impacts for native biodiversity (Daniels and Kirkpatrick 2006; Burghardt et al. 2009; Lerman and Warren 2011). Expressing this message as positive and regionally relevant, i.e. that native birds are what makes the particular city unique (Aronson et al. 2014), can help improve communication effectiveness and thus uptake (van Heezik et al. 2012). Individual homeowners may also find that they share common values. In an extensive survey of households in New Zealand, Freeman et al. (2012) found that over 80% of respondents saw value in planting native plants for birds. Neighbour mimicry is another way that gardening practices spread. This can be advantageous when a “keystone neighbour” champions landscapes with wildlife in mind (Goddard et al. 2013; Warren et al. 2008; Hunter and Brown 2012).

### 18.3.2.2 Bird Feeding

Feeding garden birds has become the most widespread and popular wildlife gardening activity in the Western world (Gaston et al. 2007; Jones and Reynolds 2008) and can influence bird populations at multiple spatial scales (Fuller et al. 2012; Fuller et al. 2008; Galbraith et al. 2015). In the UK, nearly 50% of householders feed garden birds (Davies et al. 2009), spending £200 million per year (British Trust

for Ornithology 2006). This equates to approximately one bird feeder for every nine birds in the UK (Davies et al. 2009). A similar percentage of householders feed birds in the USA (approximately 50 million people; USA Government 2011), Australia (Rollinson et al. 2003; Jones and Reynolds 2008) and New Zealand (Galbraith et al. 2014).

The social drivers of garden bird feeding have received recent interest. Fuller et al. (2008) show that feeding was more prevalent in wealthier neighbourhoods of Sheffield, UK. In contrast, other studies have found neighbourhood income to be unrelated to bird feeding (Davies et al. 2009, Goddard et al. 2013) or unrelated to the total prevalence of bird feeding but related to the type of food provided (Lepczyk et al. 2012). People who feed birds are more likely older and less mobile, living in single-dwelling houses and have a greater interest in nature (Fuller et al. 2008; Lepczyk et al. 2012; Galbraith et al. 2014; Goddard et al. 2013). People feed birds for a wide variety of reasons, ranging from the personal (e.g. increased happiness: Jones and Reynolds 2008; Galbraith et al. 2014), to the moral (e.g. to “give something back” or to “help the birds”: Jones and Reynolds 2008; Goddard et al. 2013; Galbraith et al. 2014) and the environmental (e.g. to increase over-winter survival or to provide habitat resources for threatened species: Jones and Reynolds 2008; Mason 2000). In Australia, where bird feeding is discouraged (Jones and Reynolds 2008), people’s decision *not* to feed may also be environmentally motivated, i.e. through the belief they are helping garden birds by not feeding (Rollinson et al. 2003).

In addition to the number and types of people feeding birds, the kinds of food provided also strongly influences garden bird populations. Typical items include sugar water, commercial seed and meat (Lepczyk et al. 2012; Rollinson et al. 2003); bread is also provided, particularly in lower-income neighbourhoods (Lepczyk et al. 2012). Galbraith et al. (2014) estimated that householders in New Zealand were putting out a staggering 5.1 million loaves per year, causing a shift in the bird community towards species tolerant of this food source.

The implications of feeding on garden birds are varied, and both benefits and risks to bird populations have been identified (Jones and Reynolds 2008; Fuller et al. 2008; Galbraith et al. 2014). However, the majority of supplemental food studies are not conducted at home feeders and thus might not mimic garden conditions nor do they control for human factors (Robb et al. 2008). Nonetheless, these data provide an insight as to the ecological consequences of bird feeding. Positive effects on the bird community include higher garden bird richness and abundance (Fuller et al. 2008), increased survival over winter or when food is scarce (Jones and Reynolds 2008) and increased breeding success (Schoech and Bowman 2001). For example, Orros and Fellowes (2015) found that supplementary feeding by over four thousand households in Reading, UK, provided food for up to 320 red kites (*Milvus milvus*), explaining their high abundance in the urban area. However, the benefits of feeding birds are more often articulated in terms of their benefits to humans, through increased well-being (Fuller et al. 2008; Goddard et al. 2013), opportunities to interact with nature (Lepczyk et al. 2012) and ecosystem services (Orros and Fellowes 2012).

Despite these positive aspects of bird feeding, this practice can also have negative implications for garden birds. Putting out food too frequently, or too unreliably, can cause dependence on supplementary feeding, and using food with low nutritional value might decrease individual fitness (Rollinson et al. 2003). High-fat and high-protein supplemental foods might lead to earlier laying dates, which might be detrimental when insect foods required for nestlings are not in synchrony (Schoech and Bowman 2001). Garden bird feeding also increases competition from other species attracted to gardens/feeders, causing novel interspecific associations. For instance, Bonnington et al. (2014) found that bird visitation to feeders in Sheffield, and the amount of food removed by birds, declined by more than 90% due to competition with squirrels. Other risks associated with the increased aggregations of birds around feeding stations include predation (although this may be counteracted by increased vigilance and the dilution effect; Dunn and Tessaglia 1994) and disease transmission (Robb et al. 2008). Providing supplementary food might also alter garden bird communities. For example, in Australia and New Zealand, garden bird feeding has favoured exotic species, especially when seeds are the primary supplemented food item (Galbraith et al. 2014; Daniels and Kirkpatrick 2006; Galbraith et al. 2015). Fuller et al. (2008) found that feeders in UK gardens do not entice new bird species to urban areas but are associated with an increase in the abundance of species that use feeders frequently. If the negative implications of garden bird feeding outweigh positive aspects, householders might be creating an ecological trap (Schlaepfer et al. 2002). For example, this may occur when supplementary food creates a false cue that gardens are high-quality habitat, but there are insufficient resources for breeding or increased stresses, leading to nest failure (Balogh et al. 2011).

Many organisations provide guidelines on when and whether to feed birds (Cannon 1998). In the UK, the British Trust for Ornithology encourages bird feeding and promotes the practice as a conservation action, e.g. through supporting populations of song thrushes (*Turdus philomelos*) (Mason 2000). In the USA, householders in the northeast are encouraged to remove bird feeders between April and November to reduce conflicts with American black bears (*Ursus americanus*) and to encourage birds to take advantage of the abundant natural food sources, e.g. insects for nestlings and berries (MassWildlife 2014). However, there appears to be a mismatch between public and private spaces, with feeding discouraged in National Parks yet a proliferation of bird feeding supply stores, which leads to mixed messages to the public (Lepczyk et al. 2012). In Australia, on the other hand, bird feeding is discouraged in both public and private places, yet it is still a very popular activity so the message does not seem to be working (Jones and Reynolds 2008).

### 18.3.2.3 Cats

Cats represent the most severe anthropogenic threat to bird populations in residential landscapes. In the USA, domestic outdoor cats, both owned pets and unowned,

kill roughly 1.5–3.7 billion birds per year (Loss et al. 2013). In southeastern Michigan, an estimated one bird is killed per km each day, including species of conservation concern (Lepczyk et al. 2004a). In a study documenting nest success and post-fledgling survival in Washington, DC, almost half of gray catbird (*Dumetella carolinensis*) predation was attributed to domestic cats (Balogh et al. 2011). Cats also induce fear and alter behaviour of suburban birds causing sublethal impacts. For example, the fear instilled by domestic cats can reduce fecundity by one offspring per year which results in up to a 95 % reduction in bird abundances (Beckerman et al. 2007; see also Bonnington et al. 2013). In Chicago neighbourhoods, native birds (predominantly migratory species) were less abundant with the presence of owned outdoor cats (Belaire et al. 2014). Cats are known to roam, and their impact is not restricted to their owner's garden. For example, Thomas et al. (2012) found that cats moved within a daily range of 1.94 ha, van Heezik et al. (2010) found that mean home range size was 3.2 ha, and Eyles and Mulvaney (2014) found that cats may travel up to 900 m into nearby protected habitats. Susceptibility to cat predation can vary with natural history traits. For example, Cooper et al. (2012) found that ground-foraging and cavity-nesting birds, including individuals using nest boxes, were as much as three times more susceptible to cat predation.

The majority of information on cat impact relies on owner surveys documenting the number and type of prey brought home. However, many cat owners are in a state of denial regarding their pet's negative impact on wildlife. This primarily stems from the fact that their pet does not necessarily bring home all their prey items, and, hence, the owner does not perceive a conservation problem (van Heezik et al. 2010). To quantify this mismatch, Loyd et al. (2013) fitted "KittyCam" video cameras on 55 owned, free-roaming cats in Athens, USA. They found that cats returned only 23 % of prey items to their household, 28 % of prey items were consumed and an additional 49 % remained at the capture site (Loyd et al. 2013). Similarly, in Reading, UK, Thomas et al. (2012) found that although cats were killing on average 18.3 birds per year, only 20 % of cats returned four or more dead prey to their households. Thus, many studies relying on surveys grossly underestimate the sheer volume of wildlife falling victim to domestic cats.

The devastating impact caused by owned cats has led to a number of mitigation strategies. One obvious strategy is for pet owners to prevent cats from going outside. The American Bird Conservancy "Cats Indoors" campaign has spearheaded this conservation issue through promoting partnerships between wildlife biologists, veterinarians, humane societies, wildlife rehabilitators and nature centres. The key message being that keeping your cat indoors is better for the cat, better for birds and better for people (abcbirds.org). A modified version of this strategy is to keep cats indoors at night. In a study from Florida, USA, video-monitored northern mockingbird (*Mimus polyglottos*) nests had the majority of cat predation events occurring at night. Thus, recommending cats indoors at night time might lessen the impact during the vulnerable nestling stage (Tracey 2011). However, in Dunedin, New Zealand, cats brought more prey home during the day compared with night time, whereas evening prey items consisted largely of

introduced mice and rats (van Heezik et al. 2010). Another strategy is to prohibit cats within defined buffer zones surrounding important wildlife sites (Thomas et al. 2012). Belled collars have also been shown to be effective in reducing predation, with one study demonstrating a halving in prey items returned home after belled collars were fitted and no evidence that cats adjusted their hunting behaviour to become stealthier (Ruxton et al. 2002). Thomas et al. (2012), however, found that only a quarter of cat owners use collars due to perceived risks to their pets.

## 18.4 Policy, Planning and Tools for Incentivising and Engaging Householders

### 18.4.1 *Engaging the Householder*

Sympathetic management for birds in residential ecosystems can arise through a combination of ‘bottom-up’ and ‘top-down’ mechanisms. Given that the management of private gardens lies largely outside direct government control, the diversity of plants and birds in gardens is mainly a result of bottom-up processes, i.e. the collective outcome of individual landscaping decisions (Kinzig et al. 2005). Therefore, initiatives that incentivise householders and local communities to increase the diversity of vegetation in their gardens should be a priority for avian conservation in residential settings. There is a long history of ‘wildlife-friendly’ gardening in the UK (Ryall and Hatherell 2003; Knight 1954) and the USA (Thomas et al. 1973), and conservation NGOs, government agencies and local agricultural extension programmes have adopted various initiatives for engaging householders in bird-friendly gardening activities. Examples include the Royal Society for the Protection of Birds’ (RSPB) ‘Homes for Wildlife’ scheme in the UK (<http://www.rspb.org.uk/hfw/>) and the National Audubon Society’s ‘Audubon at Home’ project in the USA ([http://www.audubon.org/bird/at\\_home/](http://www.audubon.org/bird/at_home/)). A recent review of the National Wildlife Federation’s (NWF) Certified Wildlife Habitat™ programme shows that householders who certify their gardens have greater quality and quantity of habitat than random non-certified gardens, but the effect on birds is unknown (Widows and Drake 2014). The NWF and other organisations are also scaling up their efforts and developing tools for certifying neighbourhoods and communities (e.g. the National Audubon Society’s Bird-Friendly Communities programme) to better capture the spatial-scale birds use (Warren et al. 2008). Homeowner and similar neighbourhood associations also have the potential to influence garden designs at a larger scale but from a top-down approach. The institutional framework of these private entities, combined with their prescriptive landscape guidelines, could provide a vehicle for delivering wildlife-friendly gardening features. In fact, neighbourhoods governed by homeowner associations in Phoenix, AZ, supported greater native bird diversity compared with ungoverned neighbourhoods (Lerman et al. 2012a). Hostetler

et al. (2011) also recommend the introduction of policies that provide incentives for housing developers to engage residents through environmental education programmes.

Putting these programmes into practice raises more challenges, but this implementation gap has the potential to be bridged through citizen science. Private gardens are ripe for the public to participate in scientific research, and researchers are increasingly partnering with households for assistance with data collection. Often, citizen science projects have multiple goals that benefit both the citizen and the scientist (Bonney et al. 2009). For example, the Cornell Lab of Ornithology based in New York, USA, and the British Trust for Ornithology have a number of citizen science projects focused on documenting garden birds, breeding success and winter bird populations (see Table 18.1). These programmes have the potential to provide important data for fundamental scientific research. For example, using data from the Cornell Lab of Ornithology's Project FeederWatch, scientists were able to link a warming climate to a shift in garden bird communities (Princé and Zuckerberg 2015). Further, these programmes provide an opportunity for the public to learn about their local birds with the hope of a return investment in conservation initiatives, such as providing wildlife habitat in private gardens, which will ultimately benefit these birds (Cooper et al. 2007; van Heezik et al. 2012).

### ***18.4.2 Garden Birds in Urban Planning and Policy***

The value of residential habitats for birds is also influenced by top-down processes such as urban planning. There is an urgent need to understand how best to plan and design cities to minimise their impact on biodiversity, and the extent of residential green space is a key consideration in the urban 'land sparing' vs. 'land sharing' debate (Lin and Fuller 2013; Soga et al. 2014). For instance, under a land-sharing scenario, increasing the size of individual gardens in new developments will increase the likelihood that householders will plant or retain trees and other vegetation (Loram et al. 2008; Smith et al. 2005) with positive knock-on effects on garden-scale bird diversity (van Heezik et al. 2013). However, compact development under a land-sparing scenario, comprising high residential density and small backyards, could support greater bird diversity at the city scale, due to a trade-off between maximising population viability of urban birds and people's interactions with birds in their own gardens (Sushinsky et al. 2013).

In addition to the amount of habitat present in gardens and the surrounding neighbourhood, how that habitat is spatially configured also affects birds (Ikin et al. 2013a; Pellissier et al. 2012; Huste and Boulinier 2011; Huste et al. 2006). Habitat connectivity throughout the city is especially important to allow birds to disperse through the urban matrix (Tremblay and St. Clair 2011; Fernandez-Juricic 2000; Shanahan et al. 2011; Donnelly and Marzluff 2006). Rudd et al. (2002) demonstrated that gardens are essential for providing connectivity; urban green spaces were found to be unviable for a hypothetical indicator species unless

**Table 18.1** Selected garden bird citizen science initiatives in the UK, the USA, Australia and New Zealand

Programme name	Geog. scope	Year launched	Design	Institution	Website	Example academic references
Big Garden Birdwatch	UK	1979	One-hour survey within 2-day winter period each year	Royal Society for the Protection of Birds	<a href="http://www.rspb.org.uk/birdwatchbbc">www.rspb.org.uk/birdwatchbbc</a>	–
Garden BirdWatch	UK	1995	Weekly survey	British Trust for Ornithology	<a href="http://www.bto.org/volunteer-surveys/gbw">http://www.bto.org/volunteer-surveys/gbw</a>	Cannon et al. (2005), Morrison et al. (2014)
Neighborhood Nestwatch	USA	2000	Mentored programme of individual visits by scientists to private gardens. Nest monitoring during the breeding season and tracking annual survival of colour-ringed backyard birds in six US cities	Smithsonian Migratory Bird Center	<a href="http://nationalzoo.si.edu/scbi/migratorybirds/research/neighborhood_nestwatch/newsletter/about_nm.cfm">http://nationalzoo.si.edu/scbi/migratorybirds/research/neighborhood_nestwatch/newsletter/about_nm.cfm</a>	Ryder et al. (2010)
Great Backyard Bird Count	USA	1998	Minimum 15-minute survey within 4-day winter period each year	Cornell Lab of Ornithology, National Audubon Society	<a href="http://gbcc.birdcount.org/">http://gbcc.birdcount.org/</a>	Guralnick and Van Cleave (2005)
Project FeederWatch	USA	1976	Weekly survey over two consecutive days between November and April	Cornell Lab of Ornithology, Bird Studies Canada	<a href="http://feederwatch.org">http://feederwatch.org</a>	Princé and Zuckerberg (2015)
Birds in Backyards	AUS	2000	Various survey methods (single day, 20-minute survey; 1-week list; 1-year list)	BirdLife Australia	<a href="http://www.birdsinbackyards.net/">http://www.birdsinbackyards.net/</a>	Parsons et al. (2006)
Garden Bird Survey	Canberra, AUS	1981	Weekly tally over whole year	Canberra Ornithologists Group	<a href="http://canberrabirds.org.au/observing-birds/frequently-asked-questions/surveys/">http://canberrabirds.org.au/observing-birds/frequently-asked-questions/surveys/</a>	Grarock et al. (2013)
New Zealand Garden Bird Survey	NZ	2007	One-hour survey within 10-day winter period each year	Landcare Research	<a href="http://www.landcareresearch.co.nz/science/plants-animals-fungi/animals/birds/garden-bird-surveys">http://www.landcareresearch.co.nz/science/plants-animals-fungi/animals/birds/garden-bird-surveys</a>	Spurr (2012)

networks of garden habitat were enhanced to allow movement through the urban matrix. It is therefore critical that gardens are planned in relation to the larger network of green infrastructure, drawing on Colding's (2007) theory of 'ecological land-use complementation' that advocates the clustering of private and public green spaces to maximise ecological functioning in urban ecosystems. This approach is exemplified by 'conservation development' (of which Conservation Subdivisions are a type), an alternative form of residential development in which homes are built on smaller lots and clustered together (Reed et al. 2014). To better inform the design of such conservation development, it is imperative to collect data on existing bird-habitat relationships in natural lands designated for urbanisation (Stagoll et al. 2010).

Management of existing gardens can also be 'scaled up' to maximise habitat heterogeneity at the city scale. For instance, Goddard et al. (2010) recommend the creation of 'habitat zones', whereby groups of gardens and adjacent habitats are managed under a common theme depending on landscape context (e.g. woodland, wetland). The creation and maintenance of bird-friendly residential landscaping will also require the engagement of housing developers. For example, in the USA, the Texas Parks and Wildlife Department has worked with developers to certify 'Texas Wildscapes' neighbourhoods, wherein the removal of trees and shrubs is minimised during construction. The Wildscapes neighbourhood was found to have higher bird diversity than a traditionally developed neighbourhood and a local natural area (Aurora et al. 2009). The academic community can also have an important role in encouraging developers, planners and policymakers to conserve urban avian biodiversity using university extension programmes (Hostetler 2012). Through a combination of sympathetic and collective management of existing gardens coupled with pre-emptive and more holistic urban planning, bird diversity could be retained in residential landscapes, despite smaller lot sizes, such that the urban land sharing vs. land sparing debate becomes somewhat of a false dichotomy.

## 18.5 Future Directions

McDonnell and Hahs (2013) call for renewed strategies in the discipline of urban ecology to better inform urban biodiversity planning and management. In addition to better temporal, spatial and cultural contextualisation, they recommend that urban ecological research (1) moves beyond patterns of distribution to the mechanisms and drivers of urban biodiversity and (2) ensures place-based studies can be generalisable by expanding to more cities, regions and countries. Although their call to action is at the city-wide scale, their points can also apply to the more specific urban environment of gardens.

When research questions incorporate an experimental approach, we can better identify some of the drivers of garden bird diversity. In 2006, Shochat and colleagues (Shochat et al. 2006) called for a more mechanistic approach to studying urban systems, but very few researchers have risen to the challenge of undertaking

such studies in gardens, and this is a barrier to the implementation of effective conservation strategies (but see Lerman et al. 2012b on how birds perceive garden quality comparing foraging decisions). In addition to the mechanistic approach, conducting these studies at multiple scales, from the individual garden through to the design of neighbourhoods and cities (Hostetler 1999), will further assess how garden birds interact with the urban matrix. And finally, designing additional studies that document vital rates such as fecundity and annual survival can further our understanding of source-sink dynamics, genetic flow and dispersal (Balogh et al. 2011).

Urban bird studies are often conducted within one city, and although this has advantages for local policy, it makes generalisations difficult. Establishing multi-city networks can be challenging, but introducing standardised methods not only provides rich opportunities for comparative studies but also has the potential to further frameworks and general principles of garden bird ecology (Table 18.1; Magle et al. 2012). Further, there is a research bias in garden bird ecology towards northern, temperate and developed cities, despite the fact that developing nations, which are often in tropical regions, are experiencing rapid urban growth (Seto et al. 2012).

Research questions and tools that have direct management and policy implications will have a stronger likelihood of implementation when we also address the decision-making process. There is a need for more socioecological interdisciplinary studies that examine the motivations behind, and barriers to, the creation of bird-friendly habitat from household to city scales and the subsequent ecological outcomes of management decisions (Cook et al. 2011). Social science can also inform how best to address environmental injustices with regards to unequal access to species-rich bird communities (Lerman and Warren 2011). Further, providing detailed information about specific habitat requirements also helps bridge the implementation gap. For example, identifying the composition, configuration and coverage of vegetation in gardens and neighbourhoods that is required for supporting native birds moves beyond the many recommendations of 'plant more trees' (Lerman et al. 2014). The scientific information should be transferable to urban planners and practitioners. In return, testing the efficacy of the policies and initiatives will further ensure conservation goals are being met.

In conclusion, understanding the drivers of bird diversity in gardens and residential neighbourhoods is crucial for the conservation of urban biodiversity and for maximising the encounters between people and wildlife that are beneficial to human well-being. The suburban mosaic of gardens supports a diverse avifauna, including species of conservation concern. However, individual gardens and neighbourhoods differ widely in their ability to sustain birds. These differences are driven by a complex and interacting range of ecological and social factors operating across scales. Vegetation structure is the most consistent predictor of garden avifauna, and management by householders and community groups should focus on the provision of habitat complexity by retaining tall and mature trees and planting new woody species and berry- and fruit-producing shrubs. Householders can also increase the abundance and richness of birds within their gardens through

the provision of supplementary food and the containment of cats. In addition, the spatial arrangement of gardens and other green spaces is an important predictor of garden birds, suggesting that gardens should be given a higher priority in urban green infrastructure plans. However, the provision and management of habitat in cities is inextricably associated with human social processes, and understanding the underlying socio-economic drivers of garden landscaping decisions is critical for making management recommendations.

Encounters between birds and people in gardens are beneficial to quality of life in an increasingly urban world. Community initiatives exist for incentivising sympathetic landscaping across gardens, and these can be promoted through engagement in citizen science. The scientific community has an important role in guiding city-scale development of green infrastructure to improve habitat connectivity for garden birds. Key areas for future research include experimental studies to understand drivers and mechanisms of garden bird diversity, comparative studies to test generalisations across regions and climates and interdisciplinary studies to better understand the link between the human and avian inhabitants of residential ecosystems.

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