Chapter 10
Counting Birds in Urban Areas: A Review of Methods for the Estimation of Abundance

Yolanda van Heezik and Philip J. Seddon

Abstract Counts of birds can inform studies with different goals, such as estimating population size, monitoring populations over time and in response to environmental change, and estimating vital rates to model population dynamics. Because estimates need to be reasonably accurate and precise, considerable thought has gone into developing counting techniques that enable robust estimation of abundance, taking into account probability of detection, which can vary between species, land cover types and over time. In recent years these have been applied to over 60 % of studies estimating bird abundance conducted in non-urban landscapes. However, robust estimation techniques are not being similarly applied to studies in urban areas. We reviewed 162 articles in which birds had been counted and abundance and/or occupancy reported in urban areas, spanning the years 1991 to 2015, and found that only 11 % attempted to account for variable detectability; few of these had modelled detectability satisfactorily. There was no indication of increasing methodological rigour over time. Counting birds in urban areas poses significant challenges; robust techniques are constrained by limitations imposed by built structures, social factors and a mosaic of many small private parcels of land. We present a framework for estimating bird abundance and discuss the strengths and weaknesses of the different approaches, relating each to the urban context. Citizen science initiatives are considered as a good fit in urban areas and are increasing in number: sampling designed for all landscapes might be inappropriate in urban areas, but counting protocols should allow the modelling of detection probability.

Keywords Bird monitoring • Detectability • Distance sampling • Abundance estimates • Population assessment • Urbanisation

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10.1 Birds in Urban Areas

Birds are visible, charismatic and widely distributed and have long attracted attention from researchers and enthusiastic amateurs alike. Birds are counted for a wide variety of reasons: to estimate population size and monitor changes over time, to evaluate habitat requirements of species, to record distributions and how these might change in response to environmental modification and to provide estimates of vital rates that can be used to model population dynamics (Bibby et al. 1992). Estimates of the size and spatial extent of a focal population are necessary to investigate size-dependent or density-dependent relationships and assess the impacts of competition and predation on populations of interest over spatial and temporal scales (Williams et al. 2001). Abundance estimates are also particularly useful for evaluating the performance of a population model, by indicating whether important biological factors influencing changes in population status have been incorporated (Williams et al. 2001). Most conservation management programmes involve some manipulation of abundance, whether it is enhancing populations of species of conservation concern or controlling pest species, and a measure of population abundance is the basic metric that indicates whether a management action has achieved its goal.

In the last few decades, there has been a huge increase in interest in urban areas by ecologists and conservation biologists. The realisation that the accelerating growth of cities is responsible for many environmental and social problems today has created an urgency to improve our understanding of the ecology of urban landscapes, in order that we are in a better position to protect and enhance the biodiversity in the spaces where most of us lead our daily lives (McDonnell et al. 2009). There is also rapidly mounting evidence that regular contact with nature is essential for our physical and psychological wellbeing (Keniger et al. 2013; Russell et al. 2013; Lovell et al. 2014). Birds are a high-profile, popular, visible and well-described taxon and have been used extensively as proxies for other biological components of ecosystems (Warren and Lepczyk 2012). Counts of birds have been carried out to explore patterns of community structure and the mechanisms driving species' distributions along urbanisation gradients (Blair 1996; Clergeau et al. 1998; Sandström et al. 2006; van Heezik et al. 2008; Menon et al. 2014) but also to investigate the impacts of specific land uses, such as gardens (Gaston et al. 2005; Daniels and Kirkpatrick 2006; Goddard et al. 2016), housing developments (Mason 2006; Tratalos et al. 2007), parks and cemeteries (Latta et al. 2012) and urban woodlands (Donnelly and Marzluff 2004; Hedblom and Söderstrom 2010; Heyman et al. 2016) and wastelands (Meffert 2016). Long-term data sets allow the evaluation of how bird assemblages change over space and time (Catterall et al. 2010; Shultz et al. 2012), and comparisons between urban and regional populations of some birds of conservation concern have provided insights into the causes of population declines of some species (Fuller et al. 2009). More recently, studies have identified the relationships between the socio-economic and cultural characteristics of human populations and the abundance and diversity of birds (Kirkpatrick et al. 2007; Loss et al. 2009; Luck et al. 2013; van Heezik et al. 2013). These factors drive bird assemblage structure and diversity and can be very important in urban areas. The use of birds as indicators of ecosystem health and change (see Herrando et al. 2016) opens up possibilities for engaging the public in data collection to inform understanding and management. Citizen science (i.e. the involvement of citizens from the non-scientific community in academic research (Tulloch et al. 2013), is a potentially powerful tool for counting birds in urban areas, with multiple benefits. On the one hand, it functions to engage and educate urban residents about the species with which they share their living space (McCaffrey 2005; Vargo et al. 2012), and it also enables the collection of wide-scale and long-term data on spatial distributions of birds in cities.

Urbanised landscapes are unique in terms of the extent of modification and degradation and in the heterogeneity and variety of different land uses (McDonnell and Pickett 1990). Direct ecological impacts include the replacement of native vegetation by buildings, roads and other structures; indirect impacts on vegetation composition and structure, which reduce habitat quality, are brought about through fragmentation and habitat degradation (Pennington and Blair 2012). Urban bird communities are also distinctive: as the degree of urbanisation increases, assemblages are composed of higher proportions of urban exploiters, species that form commensal relationships with humans, and in some countries, species which are non-native. Species that do not tolerate the transformed landscape (urban avoiders) drop out of the community, whereas urban adapters are often at their densest at intermediate levels of urbanisation (Blair 1996, 2004; Clergeau et al. 1998; McKinney 2006; Pennington and Blair 2012; Menon et al. 2014). The mechanisms behind these patterns are not well known, but local vegetation structure, availability of supplemental food, winter microclimate, proximity of remnants of native vegetation and human socio-economic factors all play a role in explaining the relative abundance of urban adapter species (van Heezik et al. 2008; MacGregor-Fors and Schondube 2011; Rodewald 2012; reviewed in Marzluff 2001).

Behaviour of birds also changes in response to urbanisation (Donaldson et al. 2007; Evans et al. 2010; Kitchen et al. 2010). Those species that thrive in urban environments, i.e. the urban adapter and exploiters, appear to possess behavioural traits that allow a more flexible response to high levels of disturbance and novel challenges (Møller 2010; Lowry et al. 2012 and refs within; Miranda 2016). Individuals of some urban species use human-subsidised resources and artificial structures, are less wary or more bold in temperament than their rural conspecifics (Vines and Lill 2014) and respond to increased year-round food resources by breeding earlier than their rural counterparts and by altering their foraging patterns and the food they eat (reviewed in Lowry et al. 2012). Some urban birds have also modified their behaviour in response to urban noise pollution, by shifting the frequencies and timing of vocalisations to improve communication (Lowry et al. 2012; Potvin and Mulder 2013; Potvin et al. 2014; reviewed in Macías-García et al. 2016). Finally, even the size and shape of birds that have adopted an urban lifestyle may differ from that of their rural counterparts: this might arise if bird populations in urban areas are established by a small number of individuals, and stochastic morphological divergence has arisen due to founder effects (Evans et al. 2009).

10.2 Objectives and Scope of the Chapter

Although the scope of research and the number of studies into the ecology and behaviour of urban birds has expanded hugely, the relative newness of the discipline raises the question of whether those studies relying on some estimate of the abundance of bird populations are applying appropriately rigorous methodology. Approaches for the estimation of bird abundance in non-urban areas might not be readily applied in urban regions. A key issue is detectability, the probability of counting a bird when it is present in the survey area. It cannot be assumed that detectability is perfect, that one species will be detected with the same certainty as other species or that a given species will be detected with the same probability in different habitats. In almost all situations, some individuals will be present but remain undetected, biasing metrics based on simple counts. The differences in behaviour that have been identified between urban and rural populations of the same species, such as flight distances (Møller 2008), have implications for detectability if comparisons are being made between populations. Moreover, the fine-scale heterogeneity of land uses typical of urban landscapes could cause detectability to vary within the same species across habitats. Counting birds in towns and cities is challenging: traditional robust methods are often constrained by limitations imposed by built structures, social factors and ownership of land. Here we present a framework for estimating the abundance of urban birds. We review the methods commonly used to count birds in urban areas and discuss the strengths and weaknesses of different approaches.

10.3 A Framework for Estimating Abundance

Population size is an appealingly transparent metric of population status, but its reliable estimation is fraught with difficulty. Ideally an estimate of abundance will be precise (low sampling variance) and accurate (unbiased). Precision will be improved through sampling intensity, recognising that it is virtually always impractical to conduct a true census (complete population count), and instead estimates of population size are based on some form of sampling. Accuracy may never be known since true population size is what is being estimated, but obvious sources of bias can be eliminated in any careful survey design, such as pre-count training of observers, and standardisation of survey conditions such as time of day and weather that take into account the behaviour of the target species. At the heart of any attempt to estimate animal abundance is the issue of detectability. Lancia et al. (1996) provide a concise categorisation of abundance estimation methods based on whether individual animals might not be detected during surveys. Figure 10.1 provides a simplified framework adapted from Lancia et al. (1996) for abundance estimation which first makes the distinction between methods to derive estimates of absolute abundance (population size (N) or density (D)) and methods that would

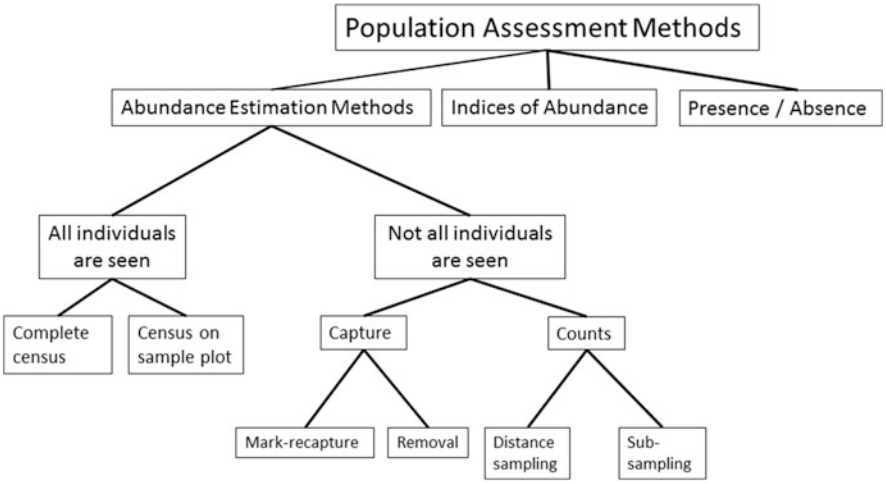

Fig. 10.1 A framework to assist in the estimation of abundance of urban birds, which distinguishes between methods that derive absolute abundance, indices of relative abundance, and presence/absence. Adapted from Lancia et al. (1996)

yield an index of relative abundance, or simple presence/absence. Indices and presence data, while apparently simpler than absolute abundance data, are not assumption-free.

10.3.1 Indices of Relative Abundance

An index of relative abundance is any measure that is correlative of absolute abundance. A typical bird count index would be the number of birds of focal species seen or heard during a defined survey period, from points or along transects, within an area of interest. A suitable index will have some positive relationship with absolute abundance. This relationship need not be linear but must be monotonic over all reasonable values of N (Williams et al. 2001). The utility of relative abundance indices is therefore dependent on the assumption of a constant probability of detection, although the method itself does not allow for the testing of this assumption (Norvell et al. 2003). Johnson (2008) defends indices by arguing that quantitative methods that account for variable detectability are limited in their practical application, particularly when extensive multispecies surveys are being carried out, and have their own shortcomings. However, indices are less likely to be adequate if comparisons are being made across habitats or between species when detectability rates are probably not similar, but they may be useful for monitoring populations if the variation in detectability is considerably less than the variation in population size sought to be detected, and is independent of population size (Johnson 2008). However, even when researchers are able to reduce the variability in detectability through study design, such as standardising count times, durations, observer skills, weather conditions and habitat features, the assumption of constant detectability is consistently violated, making comparisons of relative abundance between years within a single species and a single habitat tenuous (Norvell et al. 2003). Although widely applied, indices are seldom validated against some robust estimate of abundance of the target population.

10.3.2 Presence/Absence Data and Occupancy Modelling

Presence-absence surveys seek to confirm the presence of a focal species within the survey area, so that the recording of even one individual would be sufficient to confirm the presence. Species absence, however, is much more challenging to confirm and becomes even more problematic where the intention is to quantify the occupancy of habitat patches by a focal species, i.e. the proportion of patches occupied within a landscape, otherwise expressed as the likelihood that the focal species is present within a given habitat patch. Traditionally presence/absence surveys have made the implicit and untested assumption that there is complete detection of the target species, i.e. if the species is present at a given site, it will be seen and recorded. But for many species, the probability of detection under virtually all survey regimes will be imperfect (Gu and Swihart 2004). The failure to record a species as present when it is actually there will result in overestimation of absences and underestimation of the proportion of the patches occupied. Reliance on simple presence/absence survey data can bias estimates of changes in even relative abundance, since it is impossible to exclude the possibility that recorded colonisations arise through the misclassification of a patch as vacant in earlier surveys (Hanski 2002; Moilanen 2002).

Models have recently been developed to estimate the proportion of sites occupied by a species when the detection probability is less than one (MacKenzie et al. 2002, 2003, 2006; Royle and Nichols 2003). The basis for these modelling approaches is the repeated survey of a sample of sites within a relatively short time frame, during which it is assumed there have been no systematic changes in the occupancy state of sites. These models can be applied to data collected over a single time period, e.g. 1 year, to assess the status of the population (MacKenzie et al. 2002; Royle and Nichols 2003) or to data collected over longer time frames, such as multiple years, to assess trends in occupancy and to estimate localised extinction and colonisation rates (MacKenzie et al. 2003). The model consists of N sites being visited on T sampling occasions. The presence or absence of the species is recorded at each visit, and the detection histories for each site are then constructed and site occupancy rates estimated (MacKenzie et al. 2002, 2003, 2006).

10.3.3 Measures of Estimating Absolute Abundance

For count-based evaluation and modelling of a population, indices and presence/absence data will not suffice and some estimate of *N* (abundance) or *D* (density) is necessary (Krebs 1999).These absolute abundance estimation methods may be divided into those where the probability of detecting an animal is one, and those where incomplete detectability is likely, i.e. some proportion of the population will be missed during surveys. In the unlikely case that the entire target population can be detected and counted, this would constitute a census. A more likely scenario would involve the complete count of all individuals within a sample plot, in which case the usual sampling considerations of sample unit placement and number will apply. In most cases however, it is reasonable to expect that not all animals will be seen in any given survey, thus most of the development of abundance estimation theory has concentrated on estimating detection probability and using this to account for the missing (undetected) proportion of a population and to adjust survey data.

10.3.3.1 Capture Methods

The robust estimation of detection probability can be approached in two main ways: capture-based methods and count-based methods. Capture methods may entail the systematic capture and removal (often killing) of individuals to derive an estimate of *N*, not surprisingly most often used on common, harvested and pest species (Pierce et al. 2012). For most other situations, estimates are based on the capture, marking and recapture (or resighting) of individuals over short time periods. The simplest case would be the capture, marking and release of some unknown proportion of a target population and the subsequent capture of a second sample comprising a mix of unmarked animals and those captured previously (Greenwood 1996). This two-sample ($k = 2$) mark-recapture estimator is known as the Lincoln-Petersen estimator and is the basis for other mark-recapture methods; more precise estimates are possible where $k > 2$ (Krebs 1999). Mark-recapture methods of abundance estimation make some important assumptions relating to capture probability; a detailed discussion is given in Williams et al. (2001). Spatially explicit capture-recapture models, which incorporate information about the likelihood of animals being captured in “traps”, are a relatively new addition to the literature on abundance estimation (overviewed in Borchers 2012), with “traps” including detection devices that do not actually catch the animal.

10.3.3.2 Counting Methods

Count-based methods either directly estimate the detection probability or collect data that enables the modelling of detection probability. Direct estimation methods require an appropriate subsample of the focal population and take the form of either double sampling or the use of a radio-tagged subpopulation. In double sampling a large number of survey units are counted using some rapid low intensity effort, such as direct counts during an aerial survey, and a random subsample of the same units are counted intensively, equivalent to a census on a sample plot. The counts obtained from the subsample can then be used to estimate the proportion of animals seen during the wider survey, and this relative probability of detection can be used to correct the abundance estimates for the whole survey region (Pierce et al. 2012). Double sampling assumes that the subsample units have been truly censused and that the two sets of counts are sufficiently close in time as to sample the same population. With a radio-tagged subsample of animals, it is known precisely how many animals are available to be counted and how many of these are missed using any rapid survey method. As for double sampling, the ratio of the counts from the rapid method to the counts from the subsample provides an estimate of the proportion of animals seen.

Strip transects and fixed radius point counts apply the implicit assumption that all objects of interest are detected within a predefined strip each side of a transect line or within a fixed distance from a point. In this way the area of interest is readily calculated, and estimates of density can be derived. However, the critical assumption of perfect detectability within the defined area is seldom tested explicitly. Failure to meet this assumption will result in overestimation of abundance where fewer detections are made at greater distances from the line or point. The problem of decreasing likelihood of detection with distance from the observer led to the development of distance sampling, now one of the most widely used methods for abundance estimation (Buckland et al. 2008). Distance sampling involves the modelling of a detection function using information on the distance at which animals are detected, by sight or sound, from a point or perpendicular to a transect line. The limits of detection do not need to be defined or constrained during surveys. There are four assumptions of distance sampling: that objects directly on the point or on the transect line are never missed, that objects do not move before detection, that detections are independent of each other, and that distances are measured accurately.

The major advantage of distance sampling is that it takes into account the decreasing ability of the observer to detect objects with increasing distance. As objects are detected, their distance from the point of observation or perpendicular distance from the transect line is recorded, and through the fitting of a detection function to the distance data, an estimate of density can be made (Buckland et al. 1993). If the size of the sample area is known, density estimates can be converted into estimates of sample population size. Examination of the detection functions for even highly visible species indicates that detection probability declines rapidly with distance, further casting into doubt the validity of estimates from strip transects and fixed radius points; an accessible introduction to distance sampling is provided by Buckland et al. (2001).

10.4 Counting Techniques in Urban Areas: Current Practice

10.4.1 Methodology

To obtain an overview of the ways in which researchers have sought to estimate the abundance of birds in urban areas, we combined the results of searches on the Web of Science (https://webofknowledge.com), cross-checked with searches on Google Scholar (https://scholar.google.co) and Wildlife and Ecology Studies Worldwide (http://web.b.ebscohost.com) using the search terms *Urban* + *Bird* + *Abundance*. We included only peer-reviewed papers, and did not restrict the search to specific journals, but focussed on the last ~24 years of research as there are relatively few papers on urban birds prior to 1991, and earlier studies would not have been able to apply techniques developed in recent decades. The resulting list is therefore not an exhaustive summary of all urban bird counting studies but is indicative of the range of approaches applied.

10.4.2 Results

We found 162 articles published in 68 journals in which birds had been counted, and abundance and/or occupancy reported in urban areas, spanning the years 1991 to 2015 (details available upon request to the authors). The context of the studies was very variable, including urban/rural gradients, altitudinal gradients, urban farmland, forest, riparian areas, gardens, golf courses, green walls, housing developments, parks, cemeteries, prairie fragments, railways, streetscapes, suburbs and grasslands. Abundance was reported using a wide variety of terms: only one study reported that a census had been made, some reported occupancy, others proportional abundance, relative abundance or an index of abundance. Density was also reported on one occasion as relative density.

Of the five studies that reported occupancy, three accounted for detectability in the calculation of the estimate; however, only 17 of the 160 studies that reported abundance (11 %) made any attempt to account for detectability. We separated the studies into those published between 1991 and 1999 ($n = 13$), between 2000 and 2010 ($n = 87$) and between 2011 and 2015 ($n = 61$), but there was no evidence of a real increase in the proportion of studies accounting for variable detectability over the 24-year period (Fig. 10.2).

10.4.2.1 Measuring Detectability

A number of studies acknowledged that variable detection might be an issue, but justified in a variety of ways not having modelled detectability, e.g. asserting that

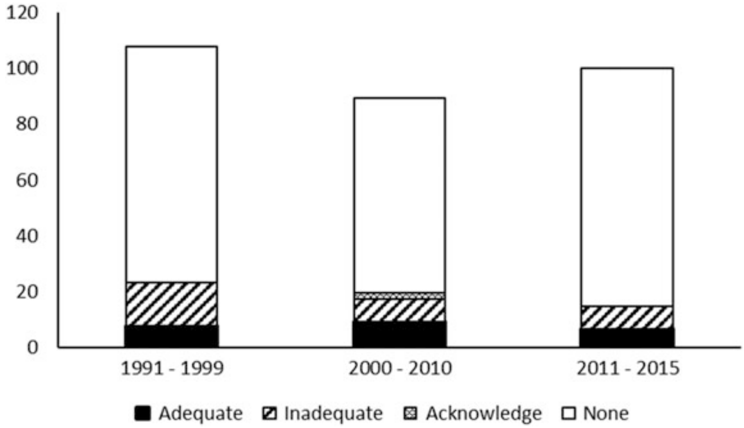

Fig. 10.2 Review of 162 articles published between 1991 and 2015 in which birds had been counted and abundance and/or occupancy reported in urban areas, with regard to whether variable detection was accounted for: adequate = variable detectability accounted for in study design and analytical methods; inadequate = methods adopted inadequate to account for detectability; acknowledge = variable detection acknowledged as a potential problem but not addressed; or none = neither acknowledged nor addressed

their methodology ensured that variation in detection probabilities was less than the variation in population size, that long sampling periods (e.g. 20 min) maximised the probability of detection of birds or that although the detectability of the species counted could differ among the habitats, a comprehensive study plot survey method meant that it was safe to assume that the observers were able to observe all individuals present during the survey period, and therefore habitat-related differences in the detectability did not significantly influence results. However, even when counts are made over longer periods of time, it is still possible to miss individuals, and it is more likely that individuals are counted more than once. A short count duration reduces the potential influence of evasive movements by the animals counted in response to the observer (Scott and Ramsey 1981). For these reasons standardised point counts recommended by different institutions are usually 5 min in length, and if they extend to 10 min, the data should be separated into time intervals (Ralph et al. 1993, 1995).

Some authors claimed that modelling detectability was not an issue because their study focused on within-species differences across habitats. In fact detectability of the same species can vary across habitats, and counts that do not account for detectability might arrive at erroneous conclusions through underestimating abundance in some habitats relative to others. The authors of one study compared two counting techniques (area search and strip transects) and concluded that both failed to provide 100 % detectability. One study adopted the approach of carrying out some pilot studies, and from these concluded that there was no significant difference in detection of bird species, so modelling of detectability was not warranted. Some authors acknowledge detectability, but made no attempt to model it, whereas others measured distances to detections or counted birds within detection bands, but then did not appear to use this information to model detection probabilities.

10.4.2.2 Use of Relative Abundance and Indices of Abundance

A number of studies in our review ($n = 14$, 9 %) reported they had measured relative abundance, relative density in one case or an index of abundance. By doing so they acknowledge that their counts were not designed to estimate absolute abundance. In fact the 89 % of studies reviewed that did not estimate detection probability were effectively presenting an index of abundance, but without acknowledging they were doing so. Standardised point count surveys have been recommended to provide data resulting in indices of abundance that are comparable across years, habitats and studies and that can be used for monitoring populations (Ralph et al. 1993, 1995). Recently Matsuoka et al. (2014) called for a revival of common standards in point count surveys, after a review by them of 125 studies across Northern America revealed a large variability in point count technique—only 3 % of the counts carried out over the period 1992–2011 followed recommended standards for count duration and radius. We also found considerable variability in duration and radius of point counts. Durations ranged between 3 and 30 min, and radii between 25 and 100 m and in some cases were unlimited. Longer count periods may be necessary to enhance detectability of songbird species if the gap between songs exceeds 5 min, but for species that move during the duration of the count, longer durations result in birds being detected more than once and birds absent from the count area initially, can enter it during the count period resulting in an overestimation of density (Buckland 2006; Johnson 2008). For example, density estimates of birds were 22–56 % higher for a 10-min count than for a 5-min count (Granholme 1983). Buckland (2006) recommend the adoption of the snapshot approach to address the problem of bird movement, which involves the observer detecting and following movements of birds at the point, and then defining a moment when the distances from the point are recorded.

There are a number of variables that can influence bird counts, such as the observer's ability to detect and correctly identify birds, environmental conditions that affect bird behaviour and observer efficiency and the physical and behavioural attributes of the birds that make them conspicuous, all of which can vary over time (reviewed in Rosenstock et al. 2002). Some studies in our review justified the absence of detection modelling by claiming that because counts were made by only one observer in similar weather conditions and because they were only interested in within-species differences in abundance across habitats, detection modelling was not necessary. To be reliable, index counts must demonstrate a positive correlation with actual bird density that is consistent across habitats and in different conditions (Rosenstock et al. 2002). Nichols (2014) argues that there are good reasons to expect variable detection probabilities when making comparisons across species, locations and times; these non-random differences are likely to preclude any consistent correlations and therefore argue against the use of count-based indices. While standardisation might reduce the influence of these factors, it is unlikely that detectability is constant (Nichols et al. 2000). In a review of studies testing for constancy of detection, Kellner and Swihart (2014) found that 86 % of studies reported significant variation and suggested that it is prudent to assume that detection probabilities differ, and therefore investigators should provide evidence of their equivalence before using indices. Indices of abundance also lack any measure of precision, without which comparisons might yield spurious results (Rosenstock et al. 2002). Point counts can be designed to account for imperfect detection and yield abundance estimates with measured precision that are comparable across time and space. Despite this only a very small proportion of the studies we reviewed used this methodology.

10.4.2.3 Use of Mark-Recapture Estimates

Abundance estimation by mark recapture was not used in any of the urban studies we reviewed. While capturing birds at locations across the urban landscape is possible, recoveries of marked birds that have dispersed across the city can be very difficult due to problems regarding access to private parcels of land, which make up most of a city's surface. Radio-tracking birds in urban areas would also be challenging for the same reason. However, spatially explicit mark-recapture methods are certainly an option to estimate the abundance of localised populations of birds in parks, reserves and other green spaces. The "captures" can be actual captures in traps or mist nests, but birds can also be captured acoustically or on camera (Borchers 2012), and spatially explicit mark-recapture analysis can be applied to incorporate information on the location of traps relative to animals to address the question of what area the traps cover (Efford 2004; Borchers 2012) and hence to estimate bird density.

In many cases the majority of detections recorded when counting land birds are based on auditory cues; however, the ability of observers to detect bird vocalisations varies significantly according to the amount of vegetation and background noise (Pacifici et al. 2008). Localisation of singing birds can be imprecise (Alldredge et al. 2007, 2008), and accurate measurement of distances to birds is one of the assumptions underlying distance sampling (Buckland et al. 2001). Dawson and Efford (2009) explore the use of an array of microphones to enable a spatially explicit capture-recapture analysis (SECR) of bird calls to produce density estimates. This approach requires that cues of individual birds are able to be distinguished and that all individuals vocalise during the sampling period. This methodology has been further developed to address some of the assumptions of Dawson and Efford (2009) that are unlikely to always hold and has been generalised for use in many situations (Stevenson et al. 2014). The various methodological approaches using passive acoustic data to estimate density are reviewed in Marques et al. (2013). None of the studies reviewed here used acoustic surveys and SECR, and careful consideration is necessary when applying this technique in urban areas. Problems associated with background noise and impacts of vegetation volume and built structures are particularly pertinent. Traffic and other urban noise could overlap with parts of the acoustic frequencies of some bird calls (Potvin et al. 2014). As the signal-to-noise ratio decreases, the signal becomes less detectable, and a threshold should be selected that is high enough to ensure detection irrespective of noise (Dawson and Efford 2009). Given that as few as two microphones can be used to collect necessary data, it could be feasible to carry out a study in an urban landscape, but the method remains to be tested.

10.4.2.4 Use of Distance Sampling

Distance sampling was the approach most commonly used in the 11 % of studies ($n = 17$) in our survey that accounted for detectability when estimating density or population size. In these studies practitioners typically counted birds from a point and either measured distance to each detection or measured them into a number of bands. While point counts are less efficient and less accurate than transects at counting birds, and errors in estimating distances or violations of assumptions generate more bias (Buckland 2006; Johnson 2008), point counts are the only feasible option across large parts of the urban landscape, because they are more likely to be able to be placed randomly with regard to the animals' distribution, which is one of the preconditions behind distance sampling (Buckland et al. 2001). If transects were placed randomly with respect to the landscape, they are unlikely to be able to be traversed as they would inevitably cross many parcels of private land and built structures. Studies reviewed here using transects sought to circumvent this problem by placing transects parallel with roads; however, any data collected in this way are likely to be unrepresentative of the surrounding area (Thompson et al. 1998; Buckland et al. 2008).

One of the limitations of distance sampling in multispecies studies is that detectability can be modelled only in species for which there are sufficient numbers of detections, perhaps as few as 30, but guidelines suggest 60–100 (Buckland et al. 2001; Rosenstock et al. 2002). Avian communities are typically composed of a relatively small number of common species and a much larger number of rare species. The strategy adopted in ten of the reviewed studies was to model detectability on species pooled according to similar morphology and behaviour, assuming that these species had similar detection characteristics. The use of surrogate species is not well studied. Surrogates should be sympatric with the uncommon species of interest, and be similar with respect to all factors influencing detectability, i.e. microhabitat use, behaviour, size, vocalisation type and pattern (Rosenstock et al. 2002). In two of the studies reviewed, surrogates were matched to rare species for habitat type and ease of detectability and comprised only a small proportion of the total. However in two other studies, surrogate detection functions were used on the majority of species, while in one study the reporting of methods was not sufficiently detailed to determine the extent of use of surrogates. Abundance estimated in this way should be treated with caution because detectability patterns may differ between the pairs of species (Buckland et al. 2008). Assumptions about detectability can be tested in distance by including the species as a covariate and conducting a multiple covariate distance sampling analysis (Buckland et al. 2008). We found only two multispecies studies which were sufficiently rigorous to the extent that they limited their density estimations to species for which they could model detectability.

Given the small proportion of urban bird studies that addressed variable detection, it was not surprising that none adopted any of the strategies proposed for difficult species (Buckland et al. 2008). For example, distance sampling can be combined with mark recapture in double-observer methods for both point and line transect sampling in situations where it is likely that not all animals at the point or on the transect are detected (Borchers et al. 2012), an assumption underlying distance sampling (Buckland et al. 2008). By using two or more observers, a combination of mark recapture and distance sampling can be used: both observers record overlapping detections independently of each other, or alternatively one of the observers is unaware of the detections made by the other, and the birds detected by both observers are considered as recaptures with the distance from the animal to the observer recorded as a covariate (Borchers et al. 2012). This spatially explicit capture-recapture model can then allow inferences about animal abundance and density.

10.4.3 Summary and Recommendations

10.4.3.1 Use of Presence/Absence

Simple presence/absence surveys can provide the basis for quantitative resource selection analyses, without any associated estimation of abundance, but there are formal methods available to consider incomplete detectability to derive estimates of the occupancy of discrete patches in an urban matrix. These could also be used to evaluate extinction and colonisation probabilities.

10.4.3.2 Use of Indices of Relative Abundance

It is important to recognise that any index of relative abundance is not assumption-free, in that it assumes that the metric being quantified varies positively and monotonically with actual abundance. Any index needs to be validated against some species-specific estimate of abundance, perhaps derived from a subset of the survey region.

10.4.3.3 Use of Censuses

Total counts of all individuals of interest over the entire survey area are probably justified only on very small plots. Extrapolation from plot-based census counts can be used to derive an estimate of total population size or mean density, taking into account inter-plot variability. However, extreme heterogeneity and issues of restricted access in urban areas make the placement of random or fully representative plots problematic, and plot size is likely to have to be challengingly small in order to have confidence that all birds were detected.

10.4.3.4 Estimation of Actual Abundance with Incomplete Detectability

In spite of the range of methods for estimating the probability of detection, surprisingly few of the studies we reviewed accounted for imperfect detection. In a survey published in 2002 on methods used to count land birds across all landscapes, in 224 papers from nine major journals, 95 % of studies relied on index counts (area counts, points, strip transects, mapping techniques), and only 13 % of studies (total proportions were >100 % because many studies used more than one method) used empirical modelling approaches (variable distance transects, variable circular plots or distance sampling; Rosenstock et al. 2002). Only 4 % of studies used distance sampling (Rosenstock et al. 2002). More recently, a literature review of 537 articles from 10 journals, published between 1970 and 2011, that estimated abundance of a range of taxa across various scales and landscapes, reported that just 23 % accounted for imperfect detection (Kellner and Swihart 2014). The proportion of studies addressing imperfect detection increased over time, from <25 % in 1971, 1981 and 1991, to 29 % and 35 % in 2001 and 2011, respectively, but for birds was over 40 % in 2001 and over 60 % in 2011 (Kellner and Swihart 2014). Our figure of 11 % of studies accounting for imperfect detection in urban landscapes is significantly lower than that for studies in non-urban landscapes. Urban ecology is a relatively recent discipline, and it is possible that its newness has engendered a lack of rigour that should be addressed in future studies.

10.5 Citizen Science and National Bird-Monitoring Programmes

The popularity of citizen science, whereby volunteers are involved in the collection of data for research and monitoring, has increased hugely in recent years, aided by the integration of the internet into daily lives and the use of new phone technologies (Tulloch et al. 2013; Dickinson et al. 2010; Bonney et al. 2014). Benefits derived from citizen science are broad: the data collected can facilitate the investigation of ecological processes over broad geographical scales, on private land and over long time scales (Howe 2006 in Tulloch et al. 2013; Dickinson et al. 2010), resulting in information that would otherwise be unaffordable (Tulloch et al. 2013). The participation in citizen science programmes can also deliver significant social outcomes, such as educating the public about science (Brosshard et al. 2005), but also documenting information to inform sustainable management of harvests, protected area establishment and environmental air quality (Bonney et al. 2014). The oldest and most common citizen science projects are bird-monitoring schemes, for example, the National Audubon's Christmas Bird Count, running since 1900 in the USA (Greenwood 2007). Bird monitoring can be categorised as cross-sectional surveying, e.g. atlases (for a review of bird atlases in urban areas, see Luniak 2016), and longitudinal surveying, e.g. breeding bird surveys (reviewed in Tulloch et al. 2013). Citizen science initiatives often span many landscapes, including urban environments (see Goddard et al. 2016; Herrando et al. 2016), but most frequently provide information on the presence of birds rather than abundance. Citizen science has been described as a "good match" for the field of urban ecology (Dickinson et al. 2010): large numbers of potential volunteers live in urban areas and are able to access the private land which comprises the greatest proportion of the urban landscape. However the design and analysis of data from citizen science projects can be challenging, and designs that have been implemented to improve the reliability of the data do not always work well in urban areas.

Many citizen science-based bird-monitoring programmes do not take species' detectability into account. Murgui Pérez (2011) compared four independent estimates of bird population sizes in Spain obtained through citizen science and found large differences between the estimates for most species, sometimes up to 30-fold and particularly in urban areas. He attributed these to a lesser extent to differences in observer skills (professionals versus amateurs), a possible effect of field methods (transects versus point counts) and differences in study design (bias in the sites selected to be counted) and to a greater extent on whether detectability was taken into account. In one programme, which estimates national population sizes for common birds (SACRE, Seguimiento de Aves Comunes Reproductoras en España), an effective census radius was calculated for each species, with the assumption that all records of each species would fall within that effective sampling area (Carrascal and Palomino 2008). Not surprisingly population estimates using the SACRE data were higher than from data where detectability was not modelled, as has been observed elsewhere (van Heezik and Seddon 2012). Murgui Pérez (2011) also speculated that extrapolations of data collected from limited habitat types to non-surveyed areas could have resulted in overly conservative estimates for some species, whereas the SACRE data were based on 22 habitat types and not subject to the same degree of cautious extrapolation. However some of the SACRE estimates appear too large to be likely, compared to total European bird populations, casting doubt on the reliability of applying the effective census radius to model detectability (Murgui Pérez 2011). The effective census radius approach might have been species-specific but also needed to be habitat-specific. Regional population estimates of jackdaw *Corvus monedula* in Spain using the technique of Carrascal and Palomino (2008) resulted in large discrepancies when compared with figures obtained through careful censuses, in this case depending on the time of year counts were made (Blanco et al. 2014). Urban environments are typically comprised of high heterogeneity of habitat types, and so researchers should make sure that detectability for each species does not vary between habitats and seasons before applying a general effective radius width.

The British Breeding Bird Survey (BBS), which was introduced in 1994 and covers a range of landscapes including urban, serves as an example of how large-scale citizen science data collection can still be carried out in a fairly rigorous manner to monitor population trends of a broad range of breeding birds in the UK (Newson et al. 2005). The BBS generates large numbers of detections of many species, by citizens who are usually experienced bird watchers. By recording birds in distance categories, the BBS allows the evaluation of detectability, and from this habitat-specific estimates of density and population size can be derived. However because birds are counted into only three distance intervals and the data from the third interval are often not used, there may be too little information on the shape of the detection function to allow goodness-of-fit testing (Buckland 2006). Newson et al. (2005) validated the estimates from the BBS by comparing them with those generated by other studies and found good agreement for most species. However the BBS design used transects to count birds, and while these may work well in most landscapes, in urban landscapes they invariably follow roads, and as such do not allow the robust estimation of density.

There are a few examples of specifically urban bird-monitoring programmes using citizen science: these include the Smithsonian Institute's Neighborhood Nestwatch Program, five studies on urban birds coordinated by the Cornell Lab of Ornithology, and the Tucson Bird Count (McCaffrey 2005). The Tucson Bird Count is a volunteer-based project using skilled observers to survey breeding birds at hundreds of sites across Tucson, using 5-min unlimited-radius point counts, with no assessment of detection probability, and primarily producing distribution maps rather than abundance indices (Turner 2003). Herrando et al. (2012) used bird-monitoring data collected in two cities, Barcelona and Brussels, to develop a multispecies indicator for each city to be used to evaluate responses of birds to environmental changes in urban habitats in other European cities. The data from Brussels were collected using point counts at 98 sites with no estimate of detection probability, and in Barcelona birds were surveyed using eight 3 km transects. The study concluded that values provided by urban indicators can differ depending on the conceptual approach (Herrando et al. 2012); however, other factors relating to study design most likely also contributed to the variation; a reliable index should be based on similar study design and should evaluate detection probability.

10.6 Final Comments

Despite significant advances in the theory of animal abundance estimation, the development of accessible quantitative tools for abundance estimation and the robust application of these tools to estimate bird abundance in natural areas, a majority of the studies reporting on bird counts in urban areas apply methodology that most likely results in biased estimates. Virtually no method is free from some assumptions around the probability of detection of individuals of the target species. Indices of relative abundance should be validated against some estimate of actual abundance, complete detection in any census should be confirmed and occupancy estimation, or capture- or count-based estimates of actual abundance (whether expressed as density or number), should apply the appropriate tools that incorporate explicit modelling of detection probability.

While distance sampling has most commonly been applied to address problems of variable detectability, surveys based primarily on auditory cues often violate the basic assumptions of this approach, as do double-observer approaches (reviewed in Schmidt et al. 2013). Moreover, in recent years researchers have drawn attention to the existence of two detection probabilities (Newson et al. 2005; Schmidt et al. 2013). Distance sampling allows the estimation of the number of animals available for detection during the survey; however, it is possible that some animals in the area being surveyed are not available for detection, e.g. in an urban area, birds might be situated behind a built structure, or the point counts might be made of vocalising birds that do not call at all during the period of the count (Nichols 2014). Two detection probabilities can therefore be estimated: the probability that an individual bird is potentially detectable (availability), and the probability that it is detected, given that it is available at some time during the count (Nichols 2014). Schmidt et al. (2013) found that variation in detection due to the presence and availability was large and differed between species of birds counted in Denali National Park, Alaska. A number of methods can be employed to estimate the probability of availability for detection: Schmidt et al. (2013) suggest that repeated count surveys and mixture models for analysis would improve the sensitivity and effectiveness of many passerine-monitoring programmes. Most importantly, the investigator needs to be aware of the different approaches and choose the one that best suits the questions being addressed. For rare species, low numbers of detections might prohibit robust estimates using distance sampling methods, and the use of surrogate species might seem appealing; however, the appropriateness of surrogates should be explicitly tested.

While a number of studies address the issue of how to deal with error and bias in citizen science data sets (Bird et al. 2013; Tulloch et al. 2013; Isaac et al. 2014), trade-offs between data quality and quantity, quantification and standardisation of sampling effort and methods and mismatches in skills and expectations of data collectors and users (Robertson et al. 2010), as well as the study design itself, are all fundamental to how reliable the data collected will be. Design needs to be such that some evaluation of detection probability is possible. Urban citizen science bird-monitoring programmes that wish to evaluate abundance should use point counts and count either into distance bands (preferably greater than three) or record distances to all detections. When sufficient data are collected, habitat- and species-specific effective radii could be modelled and validated across a range of cities.

Mirroring the accelerating growth of urban areas and their human populations has been a rapid proliferation of studies conducted on urban bird populations as well as on other urban taxa, including those based on citizen science data. At present the majority of investigators are not applying sufficiently rigorous techniques when estimating urban bird abundance. While some recent studies have accounted for imperfect detection in a rigorous manner, there is still considerable scope for an improvement in abundance estimation techniques and also for trying approaches other than conventional distance sampling when estimating urban bird population size.

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