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# A quantitative survey of avian population densities in a heterogeneous tropical urban landscape

David J. X. Tan

**Abstract.** Although birds are one of the most well studied taxa in Singapore, little remains known about absolute population estimates for various bird species owing to the prevalence of measures of relative abundance as the chief form of estimating bird populations. This pilot study is a first attempt at obtaining baseline absolute population density estimates for common bird species by applying distance sampling theory in a highly patchy and heterogeneous landscape—the Kent Ridge Campus of the National University of Singapore. The results suggest that distance sampling generally outperforms relative abundance estimates owing to its ability to account for variations in detectability across different species and habitat types. The results demonstrate that distance sampling is a viable technique for estimating absolute population densities in heterogeneous landscapes like Singapore so long as certain assumptions and caveats of survey design are taken into account. This study therefore proposes that distance sampling be considered in future bird population surveys to allow for more reliable density estimates to be obtained, which would go to no small length toward better informing and shaping conservation policy in the future.

**Key words.** distance sampling, urban ecology, avian population density estimation

## INTRODUCTION

Despite most of the earth's avian diversity being located in the tropics, the preponderance of avian population studies focusing on temperate regions has resulted in a relatively impoverished understanding of tropical avian community structures (Marzluff et al., 2001; Chace & Walsh, 2006; Ortega-Alvarez & MacGregor-Fors, 2011; Sodhi et al., 2011). While the number of comprehensive studies on tropical bird abundances has grown over the past few decades (Terborgh et al., 1990; Thiollay, 1994; Terborgh et al., 1997; Peh et al., 2006) only a few of these studies have focused on Southeast Asia (Yorke, 1984; Marsden, 1998; Sodhi et al., 2005), and even fewer studies have examined avian community structures in urban-dominated heterogeneous landscapes (Suarez-Rubio & Thomlinson, 2009; Zhou & Chu, 2011).

Among the Southeast Asian nations, Singapore ranks as one of the more extensively surveyed localities in the region, with ornithological records dating back to the 1840s (Wang & Hails, 2007) as well as possessing an active ornithological community comprised of academics, civil servants, and members of civil society (Sodhi et al., 1999; Castelletta et al., 2005; Gan & Ramakrishnan, 2005; Lim & Lim, 2009). In spite of this, there exists a relative paucity of estimates of absolute avian population densities in Singapore (Kang, 1989; Sodhi et al., 1997; Lim et al., 2003). Previous bird surveys have generally involved simple bird counts and therefore only allow for the calculation of relative abundance estimates of bird populations (Lim & Lim, 2009). While such measures of abundance are undoubtedly useful for analysing population trends, population indices based on relative abundance estimates provide little information about actual population sizes (Gregory et al., 2004) and are highly susceptible to variations in bird detectability owing to differences in observer skill, habitat type and behavioural traits (Buckland et al., 2008).

In contrast, distance sampling theory (Buckland et al., 1993) provides a method for determining absolute population densities, corresponding with the average number of birds per hectare within a particular area, that accounts for variations in detectability by modeling the detection function g(x) based on the statistical decrease in observations with increasing observer distance (x). This allows for the Effective Strip Width (ESW) of the transect to be estimated so long as (1) all objects on the transect line (x = 0) are detected (g(0) = 1), (2) objects are detected at their initial location and (3) distances are measured accurately (Buckland et al., 1993). As long as the three aforementioned assumptions are met, unbiased absolute density estimates can be calculated with the equation $\hat{D} = \frac{n}{2 \times L \times ESW}$ where '*n*' is the number of birds and 'L' is the transect length. Despite its utility, distance sampling remains relatively uncommon as a survey method in Singapore, with only one prior study having been conducted in 2010 for the purposes of estimating the population density of the Sunda colugo (*Galeopterus variegatus*) (Lim & Ng, 2010).

Department of Biological Sciences, National University of Singapore, 14 Science Drive 4, Singapore 117543; Email: david.tanjx@gmail.com


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Table 1. Major habitat types of the NUS Kent Ridge campus, adapted from Tan et al. (2010).

| Habitat Type | Estimated Area (ha) | Description |
|---|---|---|
| Secondary Forest | 26.9 | Dense *Adinandra belukar*-type heath forest dominated by *Adinandra dumosa* (Tan et al., 2010). |
| Parkland | 2.6 | Managed vegetation largely dominated by cultivated trees, especially *Samanea saman*, and turfed with *Axonopus compressus*. |
| Field | 6.0 | Large, flat and open grassy field mostly turfed with *Axonopus compressus* with little to no tree cover. |
| Low-density Built Areas | 4.7 | Low-rise urban buildings (generally < 3 storeys) with low human traffic/disturbance levels and relatively low levels of ground concretisation. |
| Built Areas | 113.0 | Medium to high-rise buildings with high levels of human traffic/disturbance. Most ground surfaces concretised into pavements with sparsely distributed patches of *Axonopus compressus* as well as habitat-softening features such as urban landscaping and roadside trees (e.g. *Albizia saman*). |

To compare the effectiveness of distance sampling vis-à-vis relative abundance in assessing wild bird populations, the first ever distance sampling-based avian population census in Singapore was conducted to estimate the absolute population densities and relative abundances of common birds within a highly patchy and heterogeneous landscape. This pilot study was conducted at the Kent Ridge campus of the National University of Singapore (NUS) owing to its heterogeneous mix of different habitat types within close proximity to each other, and represents a first attempt at obtaining unbiased baseline density estimates of common bird species for both the NUS campus and for Singapore in general. This study will therefore allow for the assessment of the effectiveness of distance sampling as a surveying technique in small but highly heterogeneous urban-dominated habitats, as well as provide important caveats and considerations for future distance sampling surveys.

Fig. 1. Map of the NUS Kent Ridge campus showing the transect line in red and the habitat distribution. Green = Secondary forest; Light green = Field; Purple = Built Areas; Light blue = Low-density Built Areas; Yellow = Parkland.

## MATERIAL AND METHODS

**Study design.** The bird density census was conducted using a modified line transect survey method; wherein a single transect line (4.08 km) (Fig. 1) passing through the five major habitats of NUS (Table 1) was drawn. Effort was made to ensure minimal overlap in survey areas to avoid double counting. Each transect survey was conducted by walking at a slow and constant pace from one end of the transect line to the other, with each survey ranging from two to three hours at an average of 2.68 hours per survey. Birds were detected mainly through visual confirmation with a pair of Vanguard binoculars (10 × 42) and detections based on bird vocalisation were not recorded unless visual detection of the bird was made within five to ten seconds of vocalisation.

To fulfill the criterion that all birds on the transect line are detected, looking ahead was practiced to detect birds ahead of the observer to ensure that observer disturbance did not bias distance measurements away from the centerline. Looking ahead also ensured that birds were detected at or near to their initial positions to minimise the effect of observer disturbance. Distance measurements based on bird detections were made using a Nikon LASER 550AS Rangefinder (6 × 21, 6°), with a measurement range of 10 m to 500 m precise to 0.2 m. For distances less than 10 m, a combination of visual estimation and transect tape measurements (Yamayo Million 100 m) was used to measure the horizontal perpendicular distance of the detected bird to the transect line. Additionally, marked arc of fire sticks were placed at various points along the transect in 10 m intervals up to the nearest treeline perpendicular to the transect to aid in visual estimation of detection distances. Effort was made to ensure that distance measurements were made as a result of chance detections rather than point scanning or flushing. Additionally, position relative to the transect line of detected birds (right/left hand side) was also noted to account for areas where habitats differ on either side of the line. For birds detected during temporary observer deviations from the transect, direct distance measurements were recorded. Surveys were conducted at three main periods of the day in the morning (before 1200h), afternoon (1200–1600h) and evening (after 1600h) to account for diurnal variation in bird density. In all, a total of 51 replicate surveys were conducted between the months of January 2012 and March 2012 in weather conditions ranging from fair weather to light rain.

Although bird detections were recorded regardless of residential status, density estimates were only calculated for resident species and migratory species excluded given the limited time scope of the study as well as the transient nature of many migratory species. In addition, birds detected in midflight were not included in density estimates due to the difficulty in establishing initial position and measuring aerial detection distances, thereby excluding most raptors, swallows (Hirundinidae) and swifts (Apodidae) from the distance data collected. Species observed but not used for calculating density estimates were nonetheless noted down and compiled into an overall bird list for NUS (see Table S1 in Supplementary Data).

**Distance-sampling analysis.** Distance data was analysed with Program DISTANCE (version 6.0 Release 2) to obtain population density estimates for each bird species (Thomas et al., 2010). To account for the patchy nature of the surveyed area, the transect line was post-stratified by habitat type into 26 segments of varying lengths, with each segment functioning as a single sampling instance. Habitat-specific density estimates were calculated for each species where sample sizes permitted (at least 60–80 detections). For rare species with comparatively smaller sample sizes, measurements were pooled across related (e.g., Columbidae) or similar (e.g., Nectarinidae and Dicaedae) species to derive a common detection function from which densities could be estimated. As recommended by Buckland et al. (1993), a baseline truncation of 5% of the largest observations was applied to exclude outliers that provide little information for density function estimation. Best-fit models were chosen based on the minimum Akaike's Information Criterion (AIC) score, which selects for models that fit the data with a minimal number of parameters (Buckland et al., 1993).

**Relative abundance analyses.** To compare the effectiveness of ongoing bird survey methods with that of distance sampling, relative abundance values were calculated for each species for each habitat stratum. To determine the level of concordance between the two methods for habitats where avian detectabilities are likely to differ, simple linear regressions were calculated in R v3.2.0 (2015-04-16) to compare the relative abundance values with the distance-based density estimates for the secondary forest and built area habitat strata. In addition, the Shannon Diversity Index score was calculated for each habitat type using the formula, $H = -\Sigma p_i LNp_i$, where $p_i$ is the relative abundance for the *i*th species in the habitat.

## RESULTS

**Relative abundances.** In all, a total of 3846 individual bird detections were made, comprising 57 species detected over the course of the survey period. Of these 57 species, 6 species were migrants, including one breeding visitor, the Blue-throated bee-eater (*Merops viridis*). As for resident breeding species, the surveyed area was overwhelmingly dominated by Javan mynas (*Acridotheres javanicus*), with a relative abundance of 46.5% compared to the 10.7% relative abundance of the second-most abundant species, the Yellow-vented bulbul (*Pycnonotus goiavier*). Overall, the 10 most abundant resident species comprised 84.2% of the total bird detections. A complete list of bird species observed, including species detected but not recorded for density estimation, can be found in the supplementary data. In all, including species not used for density estimates and past species records, a total of 58 resident species were detected on the NUS campus, which comprises 35.4% of the entire extant resident bird fauna of Singapore.

Across the various habitat types surveyed, secondary forest habitats were the richest and most diverse, with 42 species recorded and a Shannon diversity index score of 2.82, while field habitats were the least diverse and most species depauperate with 10 species having been observed and with a Shannon diversity index score of 0.59. Between these two extremes, built areas, parkland and low-density built areas constituted fairly intermediate habitats with similar Shannon diversity index values ranging from 1.96 (built areas) to 2.13 (parkland) and with richness counts ranging from 23 (parkland) to 36 (built areas).

**Distance-based density estimates.** Density estimates for all species were derived through unclustered analysis save for the Asian glossy starling (*Aplonis panayensis*), with 91.4% of the detected starlings occurring in clusters ranging from 2 to 31 individuals. Due to a relatively small number of clusters (number of clusters = 20), however, the calculated density estimates for the Asian Glossy Starling may not be reliable. The absolute density estimates as well as coefficient of variation (%CV) values are summarised in Table 2. The estimated total bird numbers for each species within specific stratum were computed by multiplying the density estimates with the overall habitat area (Table 1) and the results are summarised in Table 3.

Overall, built areas represented the habitat stratum with the most extreme variation in bird density, containing the bird species with the highest and lowest density, the Javan myna ($D = 2.40\ ha^{-1}$) and the Scarlet-backed flowerpecker (*Dicaeum cruentatum*) ($D = 0.028\ ha^{-1}$) respectively. Across habitat types, variation in density differed from species to species, with some species such as the Scarlet-backed flowerpecker exhibiting large differences in density across different habitats (95% decrease in density from secondary forest to built habitats) while densities remained largely invariant for species such as the Black-naped Oriole (*Oriolus chinensis*).

Comparing the results of the distance sampling and relative abundance methods, the results of the linear regressions (Fig. 2) show that for the built area habitat stratum, the distance sampling-based density estimates are strongly positively correlated with the relative abundance values, with an adjusted $R^2$ of 0.9684 and a relationship of Density = 4.5471*(Relative Abundance). In contrast, the linear correlation for the secondary forest habitat stratum indicates a much weaker, but nonetheless positive correlation between distance-based density estimates and relative abundance, with an adjusted $R^2$ of 0.7508 and a relationship of Density = 16.0744*(Relative Abundance).

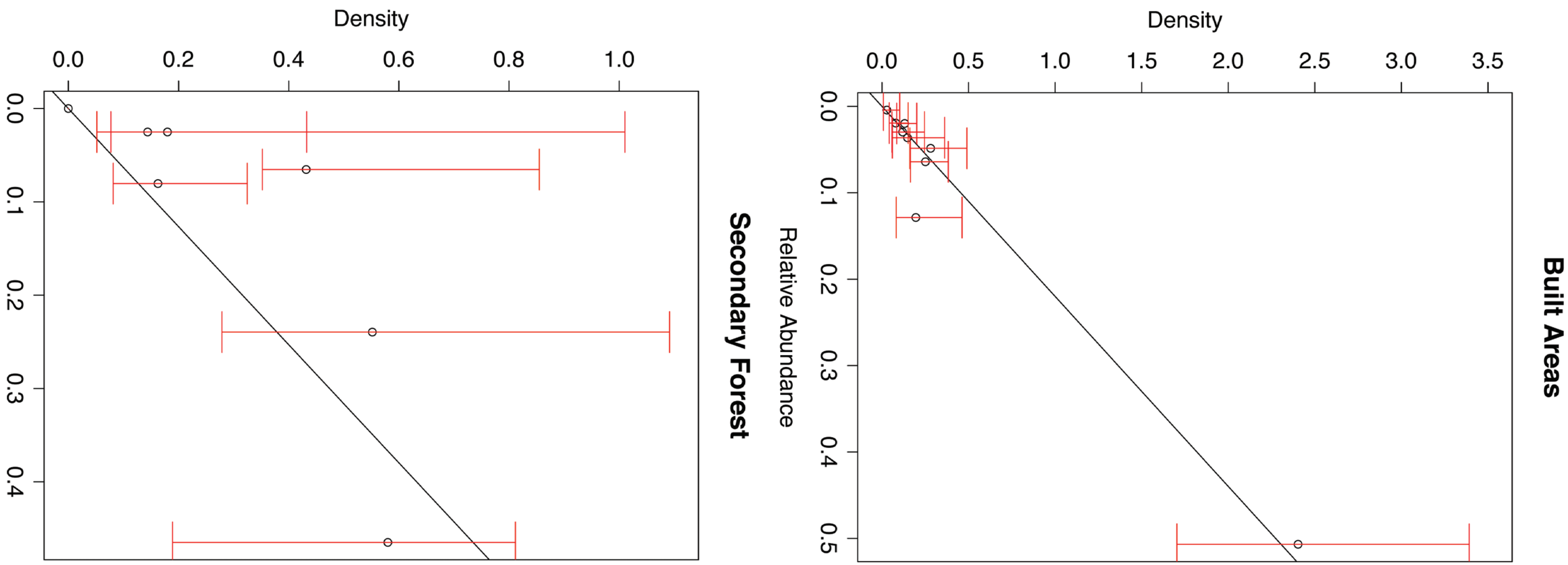


Fig. 2. Scatterplot showing the correlation between distance-based density estimates and relative abundances for both built areas and secondary forests, with the 95% confidence intervals plotted in red. For built areas, the adjusted $R^2$ is 0.9684 with a relationship of Density = 4.5471*(Relative Abundance). For secondary forests, the adjusted $R^2$ is 0.7508, with a relationship of Density = 16.0744*(Relative Abundance).

Table 2. Estimated relative abundances (RA) and population densities (D) ($ha^{-1}$) of the 10 most abundant species stratified by habitat type. Species not detected within a particular habitat stratum are assigned a density of 0.00 for that particular stratum, while species with too few detections for density estimates to be calculated are assigned a value of '–'. Species for which the %CV is close to or below 20%, indicative of low levels of uncertainty, are highlighted in bold.

| Species | Secondary Forest | | | Parkland | | | Field | | | Low-density Built Areas | | | Built Areas | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | RA | D | %CV | RA | D | %CV | RA | D | %CV | RA | D | %CV | RA | D | %CV |
| Javan myna | 0.46490 | 0.58004 | 38.20 | 0.38889 | 4.2404 | 7.95 | 0.88382 | 2.8329 | 40.39 | 0.48693 | 1.5951 | 8.65 | 0.50683 | 2.4035 | 16.34 |
| Yellow-vented bulbul | 0.23953 | 0.55186 | 31.82 | 0.20000 | 1.4843 | 7.05 | 0.00 | 0.00 | – | 0.13072 | 0.50888 | 23.41 | 0.06406 | 0.25092 | 20.48 |
| Rock pigeon | 0.00 | 0.00 | – | 0.00 | 0.00 | – | 0.00 | 0.00 | – | 0.01471 | – | – | 0.12859 | 0.19546 | 40.71 |
| Black-naped oriole | 0.08040 | 0.16268 | 31.56 | 0.01481 | – | – | 0.00 | 0.00 | – | 0.06046 | 0.21428 | 19.08 | 0.02967 | 0.11939 | 33.83 |
| Olive-backed sunbird | 0.02513 | 0.17992 | 68.33 | 0.01111 | – | – | 0.00415 | – | – | 0.02941 | – | – | 0.04852 | 0.28091 | 25.67 |
| Asian glossy starling[a] | 0.03250 | – | – | 0.00741 | – | – | 0.00 | 0.00 | – | 0.01634 | – | – | 0.04333 | 0.03867 | 60.35 |
| Collared kingfisher | 0.02834 | – | – | 0.04444 | – | – | 0.02490 | – | – | 0.00490 | – | – | 0.03627 | 0.14755 | 42.83 |
| Pink-necked green pigeon | 0.02574 | – | – | 0.02963 | – | – | 0.00415 | – | – | 0.02288 | – | – | 0.01931 | 0.07989 | 30.57 |
| Brown-throated sunbird | 0.02513 | 0.14394 | 44.79 | 0.01481 | – | – | 0.00 | 0.00 | – | 0.01144 | – | – | 0.01978 | 0.13109 | 19.69 |
| Scarlet-backed flowerpecker | 0.06533 | 0.43182 | 29.51 | 0.01111 | – | – | 0.00 | 0.00 | – | 0.01961 | – | – | 0.00424 | 0.02809 | 63.44 |

(a) Density estimates for the Asian glossy starling were calculated through clustered analysis and may not be reliable due to small sample size (number of clusters = 20).

Table 3. Estimates of total bird numbers for the 10 most abundant species stratified by habitat type. Values in parentheses are 95% confidence intervals. Species not detected within a particular habitat stratum are assigned a density of 0.00 for that particular stratum, while species with too few detections for density estimates to be calculated are assigned a value of '–'.

| Species | Secondary Forest | Parkland | Field | Low-density Built Areas | Built Areas | Total |
|---|---|---|---|---|---|---|
| Javan myna | 16 (7, 34) | 11 (9, 13) | 17 (5, 65) | 8 (6, 10) | 247 (175, 348) | 298 (223, 399) |
| Yellow-vented bulbul | 15 (8, 29) | 4 (3, 4) | 0 | 2 (1, 5) | 26 (17, 39) | 47 (34, 64) |
| Rock pigeon | 0 | 0 | 0 | – | 20 (9, 48) | 20 (9, 48) |
| Black-naped oriole | 4 (2, 9) | – | 0 | 1 (1, 2) | 12 (6, 25) | 18 (11, 30) |
| Olive-backed sunbird | 5 (1, 20) | – | – | – | 29 (16, 50) | 35 |
| Asian glossy starling[a] | – | – | 0 | – | 4 (1, 13) | 4 (1, 13) |
| Collared kingfisher | – | – | – | – | 15 (6, 37) | 15 (6, 37) |
| Pink-necked green pigeon | – | – | – | – | 8 (4, 16) | 8 |
| Brown-throated dunbird | 4 (1, 10) | – | 0 | – | 13 (9, 21) | 21 |
| Scarlet-backed flowerpecker | 12 (6, 22) | – | 0 | – | 3 (1, 11) | 15 |

(a) Density estimates for the Asian glossy starling were calculated through clustered analysis and may not be reliable due to small sample size (number of clusters = 20).

## DISCUSSION

**Effectiveness of distance sampling in estimating population abundances.** With this being the first distance sampling-based avian population study for Singapore, there presently exist few density estimates against which results from this study can be compared. Although Castelletta (2001) reported comprehensive multi-species density estimates for various forest patches in Singapore, the use of 50 m fixed-width transects and morning-only surveys in that study indicates that variation in detectability was not accounted for within a habitat where detectability biases are likely to be significant, and comparisons with this dataset would therefore be of limited utility. One notable exception was the study by Lim et al. (2003) on urban Javan myna populations in Singapore, which derived an island-wide density estimate of 2.35 mynas per hectare based on fixed-width transects in relatively detectability-unbiased urban habitats, which is fairly similar to the mean density value of 2.08 and the urban density value of 2.40 obtained in this study.

It should be noted however, that most of the density estimates calculated, especially for densities estimated through pooling across similar species, indicate fairly high levels of uncertainty with %CV values higher than the 20% coefficient of variance recommended by Buckland et al. (1993). In addition, the %CV values for the secondary forest stratum are high relative to that of the built area stratum, likely due to the fact that the denser and deeper vegetation cover of the secondary forest habitat limits the bird detection rate whereas for urban areas the concentration of most birds around the shallower and sparser streetscape vegetation likely renders it easier to detect birds during surveys. Consequently, for the same survey effort, detection rates and therefore the sample size of secondary forests is likely to be lower compared to built areas, and this smaller sample size is likely to play a role in the high %CV values obtained.

**Comparing distance sampling and relative abundance estimates.** While the results of both the distance sampling and relative abundance approaches show a general concordance between the two metrics, the difference in correlation coefficients between the built area and secondary forest habitat strata suggests that habitat characteristics are likely to bias relative abundance estimates.

For the built area habitat stratum, the high $R^2$ correlation coefficient between the relative abundance and distance-based density values suggests that the ratio of relative abundance to distance-based density is largely constant, indicating that the detectability of avian species within built areas is high and that the relative abundance estimates are likely to be representative of the true population density. In contrast, the lower linear correlation observed for the secondary forest stratum indicates that relative abundance metrics are unable to adequately predict the true density value, suggesting that aspects of secondary forest habitats (such as dense tree cover) are likely to hamper the detectability of avian species. In particular, it is also interesting to note that the for the points that lie above the trendline – indicative of an underestimated relative abundance and low detectability – the point that deviates the most from the trendline belongs to the Scarlet-backed flowerpecker, the smallest species among the birds surveyed and one of the smallest bird species in Singapore (Robson, 2010). Conversely, the point that lies below the trendline—indicative of a species whose relative abundance has been overestimated—belongs to the Javan myna, by far the most conspicuous species among the birds surveyed owing to its propensity for foraging in the open on the ground. In addition, despite the high %CV values observed for the secondary forest stratum, much of the spread of the 95% confidence intervals for species with underestimated relative abundances occurs above the trendline, implying that the observed discrepancy between relative abundance and density values are unlikely to be affected by the high levels of uncertainty.

The results therefore suggest that the applicability of relative abundance metrics for surveying bird populations is likely to vary depending on the nature of the target species as well as the characteristics of the habitat being surveyed. For relatively open habitats such as urban areas where species are often conspicuous (by nature of being synanthropic), relative abundance can function as a meaningful metric for assessing population levels. However, for denser habitats such as forests and woodlands, the effects of detectability are likely to reduce the effectiveness and reliability of relative abundance estimates in general. In addition, the results also suggest that for habitats where relative abundance is known to possess a strong linear correlation with absolute density and the exact relationship between the two is known, it may be possible to apply a mixture of distance sampling and relative abundance methods to interpolate the densities of rare but similarly detectable species based on the relative abundance values calculated, although this likely requires further investigation and verification.

**Caveats and considerations for future survey design.** Although distance sampling appears to present a marked improvement over relative abundance estimates for surveying avian populations, there remain several caveats associated with this pilot study that need to be accounted for. Many of the considerations influencing the survey design were aimed at taking the heterogeneity and patchiness of the survey area into account, and these may in turn have affected the quality of the calculated results. For instance, although Buckland et al. (1993) recommend using point transects for patchy environments, the topography of the Kent Ridge campus, with much of the secondary forest habitats occurring on steep ridge slopes, renders point transect sampling unfeasible due to problems of accessibility, giving rise to the predominance of edge effects in the collected data. Furthermore, the small area of the secondary forest fragment favoured the selection of a single continuous line transect along the boundaries of various habitat types followed by post-stratification rather than multiple randomly selected transect lines for each habitat type. Consequently, this survey design limits the number of possible independent replicate samples obtainable.

Additionally, post-stratification by habitat also raises the issue of detection correlations between successive line segments (Thomas et al., 2010). The lack of true replicates and the small number of post-stratified 'pseudo-replicates' resulting from the survey design are likely to have contributed toward the relatively high %CV values (>20%) observed in the density estimates and render the 95% confidence intervals highly unreliable in this particular instance. In spite of this, the limitations inherent in the survey design point not to any systemic errors but are instead functions of time and resource limitations since imprecision stemming from too few true replicates can be ameliorated by defining more replicate lines, even if replicate lines overlap, so long as replicate sampling intervals are sufficiently large to avoid unnecessary bias (Buckland, 2006; Thomas et al., 2010).

Owing to the nature of the single line transect, it is also likely that population densities in field and built habitats may be underestimated owing to short-term hourly variations in bird detectability. To ameliorate this bias, an equal number of repeat surveys were conducted across the morning, afternoon and evening to account for diurnal variations in abundance and detectabilities. The fact that the sequence of habitat sampling was nonrandom does however result in built and field habitats being consistently surveyed at the midpoint of each sampling instance. This can be further mitigated by varying transect routes such that the sequence in which habitats are sampled are randomised to some degree. Moreover, the use of repeat surveys across three different survey times likely biases the results in favour morning surveys, which are generally longer in duration due to the higher levels of bird activity (see Fig. S3 in Supplementary Data). In addition, the pooling of survey times into a single analysis assumes that sufficient sampling effort has been invested toward accounting for the effect of temporal variation in detectability. Instead, future surveys ought to record the time at which individual detections were made to allow for number of detections to be plotted against time to determine if any correlation exists between detectability and time, and each of the three survey times should be analysed as separate strata or as covariates.

**Distance sampling in patchy human-dominated landscapes.** Despite the numerous caveats associated with this study and the relatively high %CV values of the density estimates, the fact that the key assumptions of distance sampling—that all objects on the transect line are detected, objects are detected at their initial location, and that distance measurements are accurate—are upheld points toward the conceptual validity of distance sampling as a method for surveying avian population abundances. In the context of patchy human-dominated landscapes such as the NUS campus, the ability of distance sampling to account for variations in detectability across different species and habitat types adds to its utility in providing relatively unbiased estimates of absolute abundance that are comparable between different habitats, although the lower detection rates associated with denser habitats such as secondary forests may require higher sampling efforts to accurately survey.

One major concern with implementing distance sampling, however, is the level of skill required for observers to collect reliable distance data since extensive training is required to visually estimate distances. This has been cited as one of the reasons why local groups of bird observers prefer to apply relative abundance measures instead of deriving absolute measures of density (D. L. Yong, pers. comm.). With the increasing availability and affordability of precise measurement equipment such as measuring tape, transect tape and electronic rangefinders, it is likely that constraints of observer training can easily be overcome. Alternatively, simplified distance intervals such as the one employed by the Breeding Bird Survey in the United Kingdom (Newson et al., 2005) may also be applied to mitigate the problem of the level of observer expertise. In this light, distance sampling is therefore a highly feasible and useful method for estimating the abundance of bird populations in patchy, human-dominated landscapes and it is hoped that the adoption of robust census methods will lead to more reliable density estimates, which would go to no small length toward better informing and shaping conservation policy in the future.

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## SUPPLEMENTARY DATA

Table S1. A checklist of birds observed during the survey period between the months of January and March 2012. Bird names and status classifications follow Wang & Hails (2007), where C = Common; U = Uncommon; R = Rare; RB = Resident Breeding; I RB = Introduced Resident Breeding; R(B) = Resident Unconfirmed Breeding; WV = Winter Visitor; PM = Passage Migrant.

| Species | | Status | Total Count |
|---|---|---|---|
| Sunda pygmy woodpecker | *Dendrocopos moluccensis* | C/RB | 12 |
| Banded woodpecker | *Picus miniaceus* | C/RB | 2 |
| Laced woodpecker | *Picus vittatus* | C/RB | 7 |
| Common flameback | *Dinopium javanense* | C/RB | 5 |
| Lineated barbet | *Megalaima lineata* | U/I RB | 3 |
| Coppersmith barbet | *Megalaima haemacephala* | C/RB | 6 |
| Dollarbird | *Eurystomus orientalis* | C/RB WV | 30 |
| White-throated kingfisher | *Halcyon smyrnensis* | C/RB | 58 |
| Collared kingfisher | *Todiramphus chloris* | C/RB | 109 |
| Blue-throated bee-eater | *Merops viridis* | C/RB | 5 |
| Blue-tailed bee-eater | *Merops philippinus* | C/WV PM | 3 |
| Asian koel | *Eudynamys scolopacia* | C/RB WV | 15 |
| Greater coucal | *Centropus sinensis* | U/RB | 1 |
| Yellow-crested cockatoo | *Cacatua sulphurea* | U/I RB | 1 |
| Tanimbar cockatoo | *Cacatua goffini* | C/I RB | 14 |
| Blue-crowned hanging parrot | *Loriculus galgulus* | U/R(B) | 10 |
| Red-breasted parakeet | *Psittacula alexandri* | C/I RB | 1 |
| Long-tailed parakeet | *Psittacula longicauda* | C/RB | 5 |
| Rock pigeon | *Columba livia* | C/I RB | 282 |
| Spotted dove | *Streptopelia chinensis* | C/RB | 44 |
| Peaceful dove | *Geopelia striata* | C/RB | 9 |
| Pink-necked green pigeon | *Treron vernans* | C/RB | 99 |
| White-breasted waterhen | *Amaurornis phoenicurus* | C/RB, U/WV PM | 29 |
| Cattle egret | *Bubulcus ibis* | C/WV PM | 5 |
| Black baza | *Aviceda leuphotes* | C/WV PM | 7 |
| Oriental honey buzzard | *Pernis ptilorhyncus* | C/WV PM | Confirmed sighting |
| Brahminy kite | *Haliastur Indus* | C/RB | 4 |
| White-bellied fish eagle | *Haliaeetus leucogaster* | C/RB | Confirmed sighting |
| Grey-headed fish eagle | *Ichthyophaga ichthyaetus* | R/RB | 2 |
| 'Dark swiftlet' | *Aerodramus sp.* | C/RB | |
| House crow | *Corvus splendens* | C/I RB | 1 |
| Large-billed crow | *Corvus macrorhynchos* | C/RB | 11 |
| Black-naped oriole | *Oriolus chinensis* | C/RB, R/WV | 152 |
| Pied triller | *Lalage nigra* | C/RB | 14 |
| Ashy minivet | *Pericrocotus divaricatus* | C/WV PM | 3 |
| Greater racket-tailed drongo | *Dicrurus paradiseus* | C/RB | 26 |
| Common iora | *Aegithina tiphia* | C/RB | 23 |
| Asian brown flycatcher | *Muscicapa dauurica* | C/WV PM | 44 |
| Oriental magpie robin | *Copsychus saularis* | U/RB | 7 |
| Asian glossy starling | *Aplonis panayensis* | C/RB | 125 |
| Common myna | *Acridotheres tristis* | C/RB | 11 |
| Javan myna | *Acridotheres javanicus* | C/I RB | 1788 |
| Hill myna | *Gracula religiosa* | U/RB | 3 |
| Pacific swallow | *Hirundo tahitica* | C/RB | 7 |
| Yellow-vented bulbul | *Pycnonotus goiavier* | C/RB | 413 |
| Olive-winged bulbul | *Pycnonotus plumosus* | C/RB | 11 |

| Species | | Status | Total Count |
|---|---|---|---|
| Oriental white-eye | *Zosterops palpebrosus* | U/I RB | 33 |
| Common tailorbird | *Orthotomus sutorius* | C/RB | 10 |
| Dark-necked tailorbird | *Orthotomus atrogularis* | C/RB | 6 |
| Ashy tailorbird | *Orthotomus ruficeps* | C/RB | 2 |
| Arctic warbler | *Phylloscopus borealis* | C/WV PM | 3 |
| White-crested laughingthrush | *Garrulax leucolophus* | C/I RB | 42 |
| Striped Tit babbler | *Macronous gularis* | C/RB | 5 |
| Scarlet-backed flowerpecker | *Dicaeum cruentatum* | C/RB | 63 |
| Brown-throated sunbird | *Anthreptes malacensis* | C/RB | 68 |
| Olive-backed sunbird | *Nectarinia jugularis* | C/RB | 140 |
| Crimson sunbird | *Aethopyga siparaja* | C/RB | 6 |
| Eurasian tree sparrow | *Passer montanus* | C/RB | 51 |
| Paddyfield pipit | *Anthus rufulus* | C/RB | 4 |
| Scaly-breasted munia | *Lonchura punctulata* | U/RB | 6 |

Table S2. Additional species records not observed during the study period. Bird names and status classifications follow Wang & Hails (2007), where C = Common; U = Uncommon; R = Rare; RB = Resident Breeding; I RB = Introduced Resident Breeding; R(B) = Resident Unconfirmed Breeding; Es = Escapee; WV = Winter Visitor; PM = Passage Migrant; NBV = Non-breeding Visitor.

| Species | | Status | Source |
|---|---|---|---|
| Red junglefowl | *Gallus gallus* | U/RB | March 2012, Unconfirmed sighting by Deborah Ho |
| Spotted wood owl | *Strix seloputo* | R/RB | August 2011, Confirmed sighting by David Tan |
| Red-whiskered bulbul | *Pycnonotus jocosus* | U/I RB | March 2012, Confirmed sighting by Richard T. Corlett |
| Oriental pied hornbill | *Anthracoceros albirostris* | U/RB | April 2013, Confirmed sighting by Chui Wai Kit |
| Large-tailed nightjar | *Caprimulgus macrurus* | C/RB | April 2013, Confirmed sighting by David Tan |
| Sunda scops owl | *Otus lempiji* | C/RB | August 2013, Confirmed sighting by Valentia Tan |
| Eastern crowned warbler | *Phylloscopus coronatus* | U/WV | October 2013, Carcass collected by TMSI staff |
| Hodgson's hawk cuckoo | *Cuculus fugax* | U/WV PM | November 2013, Confirmed sighting by David Tan |
| Siberian thrush | *Geokichla sibirica* | R/PM | November 2013, Carcass collected by Marcus Chua |
| Blue-winged pitta | *Pitta moluccensis* | U/WV | November 2013, Carcass collected by Tommy Tan |
| Crested goshawk | *Accipiter trivirgatus* | R/RB | January 2014, Confirmed sighting by Frank E. Rheindt |
| Blue-eared kingfisher | *Alcedo meninting* | R/RB | March 2014, Carcass collected by John S. Ascher |
| Schrenck's bittern | *Ixobrychus eurhythymus* | R/WV | March 2014, Carcass collected by John S. Ascher |
| Thick-billed green pigeon | *Treron curvirostra* | U/RB | December 2014, Carcass collected by John S. Ascher |
| Cinnamon bittern | *Ixobrychus cinnamomeus* | U/RB WV | December 2014, Confirmed sighting by Gabriel Low |
| Black-backed kingfisher | *Ceyx erithaca* | R/WV | October 2014, Carcass collected by David Tan |
| Jambu fruit dove | *Ptilinopus jambu* | U/NBV | January 2015, Carcass collected by David Tan |
| Savanna nightjar | *Caprimulgus affinis* | U/RB | February 2015, Confirmed sighting by David Tan |
| Rufous woodpecker | *Micropternus brachyurus* | U/RB | August 2015, Confirmed sighting by David Tan |

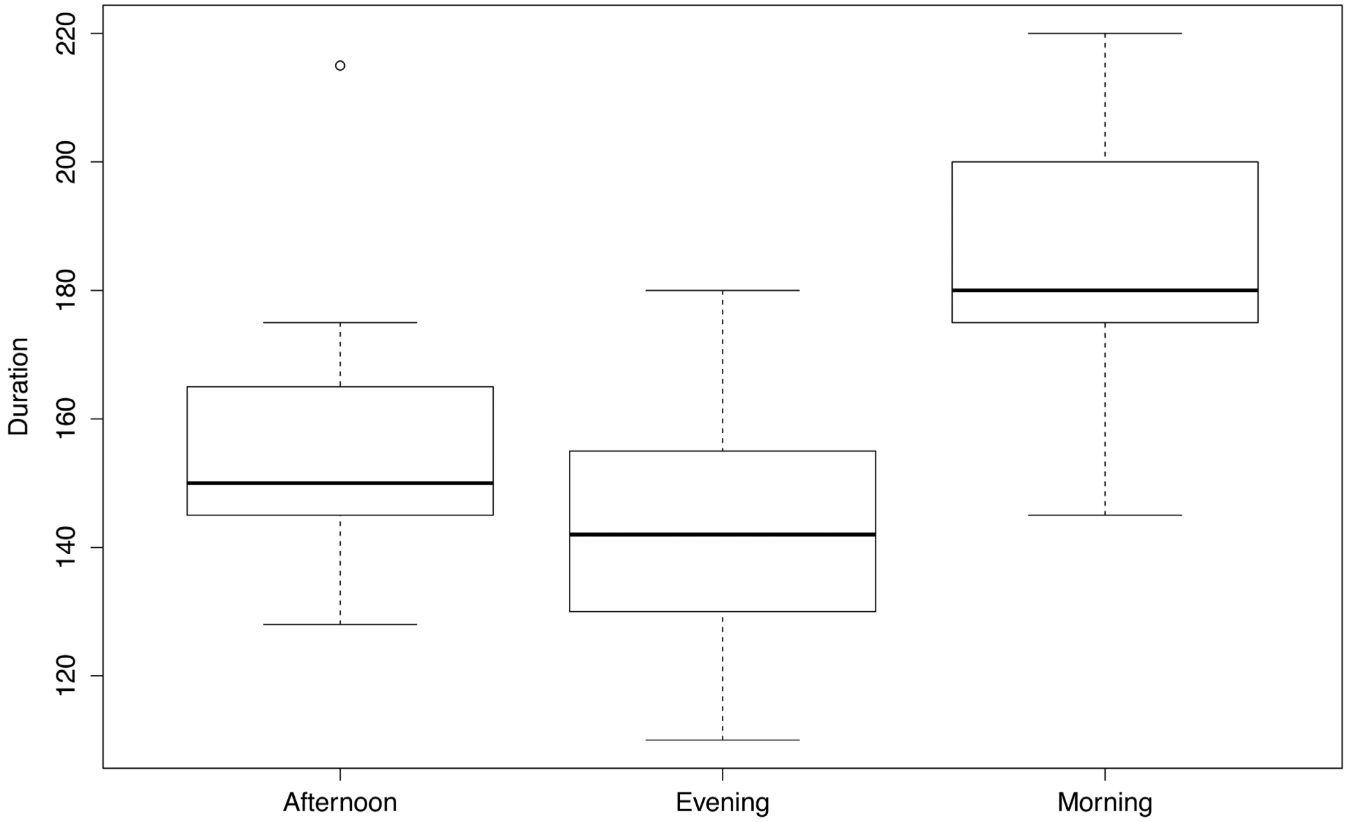


Fig. S3. Boxplot showing the spread of survey durations across the three survey timing strata, with the morning stratum exhibiting significantly longer survey durations relative to the other time strata.