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A comparison of estimates of relative abundance from a weakly structured mass-participation bird atlas survey and a robustly designed monitoring scheme

JUDIT K. SZABO,[1,2]* RICHARD A. FULLER[1,3] & HUGH P. POSSINGHAM[1]
[1] *The Ecology Centre, University of Queensland, St Lucia, Queensland 4072, Australia*
[2] *Research Institute for the Environment and Livelihoods, Charles Darwin University, Darwin, Northern Territory 0810, Australia*
[3] *CSIRO Climate Adaptation Flagship and CSIRO Ecosystem Sciences, 41 Boggo Road, Dutton Park, Queensland 4102, Australia*

Estimates of population size are frequently used in conservation. Volunteer-conducted surveys are often the only source of information available, but their reliability is unclear. We compare data from a weakly structured national bird atlas collected by volunteer surveyors free to choose where and when to visit with data from an independent suite of monitoring surveys that used a stratified sampling design. We focus on the Mount Lofty Ranges, South Australia, a region that has lost most of its native vegetation. Both datasets comprise several thousand 20-min 2-ha searches carried out between 1999 and 2007. The atlas dataset reported more species, and covered habitats more comprehensively, but showed greater variability in the temporal and spatial distribution of survey effort. However, after we restricted the atlas dataset to native eucalypt woodlands, reporting rates from the two schemes were very strongly correlated. The structured surveys tended to record more species that are normally detected by call and the unstructured surveys recorded more species using edges and open habitats. Minimum population estimates from the two datasets agreed very well. The strength of concordance depended on whether overflying birds were included, highlighting the importance of distinguishing such records in future surveys. We conclude that appropriate calibration using selected regional surveys, including surveys to estimate absolute densities, can enable volunteer-collected and weakly structured atlas data to be used to generate robust occupancy and minimum population estimates for many species at a regional scale.

Keywords: Australia, bird surveys, Mount Lofty Ranges, population estimates, volunteer-collected data.

Increasingly, large datasets are being assembled and analysed to answer questions in applied ecology and conservation across large geographical and taxonomic areas (Robertson 2008, Boakes *et al.* 2010). This has principally been made possible by the collation of records collected by volunteers or 'citizen scientists' (Cooper *et al.* 2007), be they independent field naturalists or volunteers contributing to organized research projects (Greenwood 2007). The reliability of inferences from data derived from undirected 'convenience' sampling (e.g. birdwatchers searching for rare species), or projects designed to be attractive to volunteers, has been questioned in the scientific literature (e.g. Anderson 2001). However, others have actively encouraged the growth of citizen science and analysis of the resulting data, reasoning that appropriate statistical and analytical tools and the large size of the datasets means that noise and bias can be minimized (e.g. Hochachka *et al.* 2007, Szabo *et al.* 2010).

*Corresponding author.
Email: judit.szabo@cdu.edu.au

Population size estimates and the direction and rate of change over time are fundamental products of applied ecology, being frequently used in conservation assessments, biodiversity reporting, to inform policy and to investigate the outcomes of management interventions (Newson *et al.* 2008). Estimates of bird population and distribution are often derived from volunteer-based surveys, with major schemes including atlas programmes in the UK and Europe (Lack 1986, Gibbons *et al.* 1993, Hagemeijer & Blair 1997, Baker *et al.* 2006), North America (Root 1988, Price *et al.* 1995), southern Africa (Harrison *et al.* 1997) and Australia (Barrett *et al.* 2003).

In many situations, volunteer-based surveys constitute the only available data, but making inferences based on noisy or biased data could be worse than making no inferences at all. When evaluating such concerns, it is necessary to distinguish between those problems that result in systematic bias and thus affect inferences drawn, and those that only generate noise, essentially effects that can generate Type I and Type II errors, respectively (Anderson 2001). One of the main potential sources of error relates to the non-random placement of survey effort in schemes in which observers are allowed to choose survey locations (Szabo *et al.* 2007). While such datasets could simply be disregarded (Anderson 2001), this risks losing potentially useful sources of information and could undermine public support for, and engagement in, citizen science programmes. More measured approaches might be to compare data affected by such problems with data from schemes in which probabilistic sampling has been used (e.g. Newson *et al.* 2008) or to identify important covariates of error and incorporate them into analysis (e.g. Pautasso & McKinney 2007, Phillips *et al.* 2009).

Three broad classes of variable affect detection probability: attributes of (1) the observer (training, education, experience, interest, hearing, vision, height, fatigue, etc.), (2) the environment (temperature, rainfall, wind, habitat type, season, time of day, vegetation density, human disturbance, cloud cover, etc.), and (3) the species (coloration, behaviour, size, gender, flock size, calling intensity, calling rate, matedness; Anderson 2001). Although these might only generate noise, which might be compensated for by the large sample sizes attained, there may be systematic biases in each of these elements. While this gives cause for concern, such datasets are often the most comprehensive (and sometimes the only) data available on the distribution and abundance of species across large areas, and the great benefit of comprehensiveness must be traded off against noise and/or bias in the datasets.

Bird surveys and atlases (which are large-scale collections of standardized surveys conducted by a large number of people) are the most popular faunal surveys worldwide (Gibbons *et al.* 2007). Atlases are relatively cheap to produce and can convey important information for management and conservation (U.S. North American Bird Conservation Initiative Monitoring Subcommittee 2007), and may identify areas for more detailed ecological research (Donald & Fuller 1998). Generally, their main aim is to describe species' distributions and most atlas projects have not attempted structured population censuses of species in order to avoid complicated techniques (Gibbons *et al.* 2007). Nevertheless, reporting rates are frequently calculated and used as an index of relative abundance, and reporting rates from repeated atlases have also been used to track changes in geographical ranges (Gates & Donald 2000) and detection frequencies (Barrett *et al.* 2007). Relative indices have also been calculated from other monitoring schemes (Freeman *et al.* 2001, Zbinden *et al.* 2005). While some of these approaches have been criticized, correcting for known sources of bias can make results from even weakly structured datasets more robust (Bart *et al.* 2004).

We aim to compare data from a national volunteer-based bird atlas programme in Australia with data from an intensive regional survey in part of South Australia that used a robust stratified sampling design. In this comparison we are primarily concerned with the effects of differences in survey design, as we do not have any information suggesting differences in observers' skill levels. First, we determine the strengths and weaknesses of the two types of survey data for estimating regional bird population sizes, and then assess whether robust population estimates can be derived from the weakly structured survey.

METHODS

Our study area was the Mount Lofty Ranges (MLR), which have been extensively cleared of their native vegetation since European colonization. The MLR is an area of high conservation concern given the dependence of many species on its native woodland habitats (Ford & Howe 1980). Being an 'island' of woodland in South Australia that is surrounded by drier areas, its avifauna consists of mostly closed populations and it supports several endemic subspecies and many isolated populations of species typical of wetter woodlands and forests (Paton *et al.* 1994). After about 150 years of habitat clearance, patches of native vegetation are surrounded by mixed agricultural land, such as pasture, cropland, vineyards and orchards (Westphal *et al.* 2003). Of a total area of 500 000 ha, only about 10–16% is considered good habitat for woodland birds, primarily eucalypt woodland with an understorey (Westphal 2003). Between 35 and 50 bird species were predicted as early as the late 1970s to suffer local extinction (Ford & Howe 1980). This prognosis seems to be correct – some local populations are already extinct and others are declining at an alarming rate (Szabo *et al.* 2011).

We used data from two independent surveys of the MLR region. Woodland bird surveys have been carried out in the MLR by the Nature Conservation Society of South Australia/University of Queensland (hereafter referred to as NCCSA/UQ) since 1999/2000. The project aims to detect changes in bird populations resulting from clearance of native woodland and habitat restoration initiatives (Possingham *et al.* 2004) and thus surveys were limited to the two major native woodland types: (1) stringybark woodland, dominated by *Eucalyptus baxteri* and *Eucalyptus obliqua*, and (2) gum woodland dominated by *Eucalyptus leucoxylon* and *Eucalyptus fasciculosa*. Other woodland types, such as mangroves and *Melaleuca-* and *Callitris*-dominated communities are of limited extent in the MLR (Smith & Goodwins 2001).

The survey unit was a 20-min search by a single observer of a 2-ha area, and surveys were carried out by volunteers each spring between September and early January (Possingham *et al.* 2004). The location of the surveys was determined by stratified random sampling based on the type, quality and size of the habitat. The *c.* 20 volunteers who conducted the surveys were competent local birdwatchers, although they received no specific training for these surveys. They were partially compensated for their field costs (15–20 AUD/h) as they were asked to survey hard-to-reach places that were not particularly attractive for birdwatching, and had to organize permission from landholders to access some sites. We make no assumptions about the skill levels of the observers or the degree of variability between them.

Each of the 170 survey sites was visited three times each season, with some variation during the course of the research programme. During each survey, the observer separately recorded the number of birds of each species detected (1) within the 2-ha plot, (2) flying over the plot and (3) outside the plot at the time of detection. The first two categories were separated because both within-plot and overflying birds were recorded by the Birds Australia (BA) scheme (see below), and the third was included to check for any tendency for birds outside the plot to be included in survey data. When the same 2-ha plots were repeatedly surveyed on the same day, we only used the first survey of the day. We extracted data from 8 November 1999 to 25 January 2008, a total of 3877 surveys of which 2152 were in stringybark woodland and 1725 in gum woodland.

More extensive surveying was carried out in the MLR by volunteers as part of a national atlas project run by Birds Australia (Barrett *et al.* 2003). The site selection protocol is the only major difference between the two datasets: BA volunteers were free to choose their sites and the time of their visit, while those surveying under the NCCSA/UQ protocol were instructed to go to particular sites at particular times. The preference of BA volunteers in general is for 2-ha sites in homogeneous habitat. Some sites were frequently visited, others were visited only once and there was no attempt to achieve stratification in time, space or environment. As the MLR covers a large area (65 000 ha of vegetation) and the number of birdwatchers is low, there was no close spatial and temporal overlap between surveys of the two datasets. To make the two datasets comparable in survey method, area covered and time of year, we restricted the BA dataset to those surveys that matched the NCCSA/UQ dataset. We extracted all 20-min searches of 2-ha plots from the BA database within the same date range as the NCCSA/UQ dataset. We removed all surveys outside the MLR, and those with a positional accuracy of > 5 km, including those where positional accuracy was unknown, resulting in a total of 3700 BA surveys. Observers listed the species within or flying over the 2-ha plot without distinguishing between the two categories. Because very few surveys contained count information, we treated all BA data as presence/absence. In a country-wide study most BA atlas recorders (95.7%) reported that they could identify all or most of their local birds (Weston *et al.* 2006).

Because count data were generally unavailable in the BA surveys, we used reporting rates to compare the two datasets, i.e. the proportion of all surveys in the same habitat type in which a given species was detected. To ensure sufficient sample sizes, we considered only those species with > 20 records in either dataset. We also excluded nocturnal species and waterbirds, which are poorly recorded using the 20-min 2-ha survey method. This resulted in a set of 61 species (Table 1). Species in the MLR use native eucalypt woodland to varying degrees, and data from surveys conducted only in this vegetation type would underestimate population sizes to varying degrees. Therefore, we classified species into two categories: dependent or not dependent on stringybark and/or gum woodlands, using the volumes of the *Handbook of Australian, New Zealand and Antarctic Birds* (Higgins & Davies 1996, Higgins 1999, Higgins & Peter 2002, Higgins *et al.* 2001, 2006a,b).

We calculated minimum population densities from the NCCSA/UQ dataset by dividing the number of individuals detected per species within the 2-ha plots (excluding overhead transients) by the total area surveyed in each of the two woodland types. Because the BA data recorded presence/absence rather than abundance, we used the average number of individuals detected per species per survey from the NCCSA/UQ dataset (including only those surveys where at least one individual was seen) to estimate abundances for each presence in the BA dataset. Also, because some BA presence results from birds only recorded flying over the survey area, we reduced each species' reporting rate by the same proportion as the decline in reporting rate when excluding fly-over records from the NCCSA/UQ dataset. We extrapolated these minimum population densities to minimum population estimates using the areal extent of each habitat derived from the floristic mapping (Smith & Goodwins 2001) to produce population estimates for all eucalypt woodland in the MLR.

RESULTS

The 3700 BA surveys contained records for 204 species, and the 3877 NCCSA/UQ surveys contained records from 141 species in the MLR between 1998 and 2007. The NCCSA/UQ surveys were restricted to native vegetation remnants and covered far fewer locations (170 sites) than the BA surveys (940 sites; Fig. 1). However, of the 940 BA sites, there was great variation in the number of surveys per site; 653 (69%) were visited only once, whereas two were visited more than 100 times. Overlaying the positions of the BA surveys onto detailed floristic vegetation mapping revealed that 358 of the surveys were in stringybark woodland, 830 in gum woodland, 73 in other native vegetation, and the vast majority (2439 surveys) in other land cover types such as non-native vegetation, urban areas or wetlands.

NCCSA/UQ sites were surveyed relatively evenly across years, whereas the BA dataset peaked markedly in 2000 and 2001, corresponding to a concerted survey effort for a project to map the distribution of birds across the whole country (Barrett *et al.* 2003, Fig. 2a). There was a strong seasonal bias in NCCSA/UQ dataset, with surveys being conducted only in the spring and early summer period, whereas the BA surveys were spread more evenly throughout the year (Fig. 2b). To ensure comparability among surveys, particularly where migratory species are concerned, we removed all records from the BA dataset between 1 February and 31 August for all further analyses, resulting in 554 surveys in eucalypt woodland, 167 in stringybark and 387 in gum.

Reporting rates among the 61 study species were strongly positively related between the two datasets in both the stringybark plots (standardized major axis regression: $r^2 = 0.95$, $n = 61$, $P < 0.001$; Fig. 3a) and the gum woodland plots ($r^2 = 0.92$, $n = 61$, $P < 0.001$; Fig. 3b). These relationships were much stronger than those without temporal trimming of the BA data ($r^2 = 0.68$, $n = 61$, $P < 0.001$ and $r^2 = 0.56$, $n = 61$, $P < 0.001$ for stringybark and gum woodland, respectively). The slopes of both relationships were not significantly different from unity (stringybark: $\beta = 1.005$, 95% CI = 0.917–1.093; gum woodland: $\beta = 1.001$, 95% CI = 0.878–1.125) and the residuals were approximately normally distributed, indicating that both datasets under- and over-estimated species' reporting rates fairly evenly in relation to each other. The intercept of the relationship did not differ significantly from zero for the stringybark plots (intercept = 0.003, 95% CI = −0.019 to 0.025), although it exceeded zero in the case of gum woodland plots (intercept = 0.041, 95% CI = 0.005–0.078, $P = 0.027$), indicating a slight general

Table 1. The ratio of reporting rates between the two surveys in the two woodland habitats for the 61 study species. Values are reporting rates from the Birds Australia dataset divided by those from the NCCSA/UQ dataset, so each value indicates how many presences there are in the BA survey for each presence in the NCCSA/UQ surveys. Six species, all with very low reporting rates, were detected in NCCSA/UQ but not BA surveys in stringybark (Brown Treecreeper, Crested Pigeon, Eastern Rosella, Peaceful Dove, Spotted Dove, White-winged Chough). The Chestnut-rumped Heathwren was detected in three BA surveys in gum woodland, but not by NCCSA/UQ surveys in that habitat. Within the categories, species are in taxonomic order following Christidis and Boles (2008).

Species	Stringybark	Gum woodland
Dependent on native vegetation		
Common Bronzewing *Phaps chalcoptera*	0.73	0.58
Horsfield's Bronze-cuckoo *Chalcites basalis*	1.45	0.42
Fan-tailed Cuckoo *Cacomantis flabelliformis*	1.45	0.69
Sacred Kingfisher *Todiramphus sanctus*	0.39	1.89
White-throated Treecreeper *Cormobates leucophaeus*	0.72	0.76
Brown Treecreeper *Climacteris picumnus*	NA	2.7
White-browed Scrubwren *Sericornis frontalis*	1.35	1.11
Chestnut-rumped Heathwren *Hylacola pyrrhopygia*	1.07	NA
Weebill *Smicrornis brevirostris*	1.78	0.84
Striated Thornbill *Acanthiza lineata*	0.73	0.5
Yellow Thornbill *Acanthiza nana*	0.64	0.56
Brown Thornbill *Acanthiza pusilla*	1.15	0.53
Spotted Pardalote *Pardalotus punctatus*	1.93	0.75
Striated Pardalote *Pardalotus striatus*	0.83	0.92
Eastern Spinebill *Acanthorhynchus tenuirostris*	1.03	0.4
Yellow-faced Honeyeater *Lichenostomus chrysops*	0.76	0.65
Crescent Honeyeater *Phylidonyris pyrrhopterus*	1.17	0.5
Brown-headed Honeyeater *Melithreptus brevirostris*	0.84	0.66
Eastern White-naped Honeyeater *Melithreptus lunatus*	1.02	1.25
White-browed Babbler *Pomatostomus superciliosus*	12.89	1.04
Golden Whistler *Pachycephala pectoralis*	0.88	0.57
Rufous Whistler *Pachycephala rufiventris*	0.83	0.79
Grey Shrike-thrush *Colluricincla harmonica*	0.63	0.79
Dusky Woodswallow *Artamus cyanopterus*	3.44	2.62
Grey Fantail *Rhipidura fuliginosa*	0.91	0.59
Scarlet Robin *Petroica multicolor*	0.58	0.52
Mistletoebird *Dicaeum hirundinaceum*	1.72	0.56
Not dependent on native vegetation		
Spotted Dove *Streptopelia chinensis*	NA	3.99
Crested Pigeon *Ocyphaps lophotes*	NA	0.78
Peaceful Dove *Geopelia striata*	NA	1.24
Collared Sparrowhawk *Accipiter cirrhocephalus*	1.84	1.19
Yellow-tailed Black-cockatoo *Calyptorhynchus funereus*	0.73	1.42
Galah *Eolophus roseicapillus*	0.65	0.92
Sulphur-crested Cockatoo *Cacatua galerita*	0.48	1.03
Rainbow Lorikeet *Trichoglossus haematodus*	1.28	1.3
Musk Lorikeet *Glossopsitta concinna*	1.49	1.2
Purple-crowned Lorikeet *Glossopsitta porphyrocephala*	0.71	0.72
Crimson Rosella *Platycercus elegans*	0.93	1.12
Eastern Rosella *Platycercus eximius*	NA	2.6
Red-rumped Parrot *Psephotus haematonotus*	1.61	0.95
Laughing Kookaburra *Dacelo novaeguineae*	1.03	2.18
Superb Fairy-wren *Malurus cyaneus*	1.23	1.04
Yellow-rumped Thornbill *Acanthiza chrysorrhoa*	4.3	1.02
White-plumed Honeyeater *Lichenostomus penicillatus*	1.61	1.53
Noisy Miner *Manorina melanocephala*	38.66	3.47
Red Wattlebird *Anthochaera carunculata*	1.23	1.19
New Holland Honeyeater *Phylidonyris novaehollandiae*	2.37	1.11
Black-faced Cuckoo-shrike *Coracina novaehollandiae*	0.43	1.1
Australian Magpie *Cracticus tibicen*	1.31	1.67

Table 1. (Continued)

Species	Stringybark	Gum woodland
Grey Currawong *Strepera versicolor*	0.96	0.89
Willie Wagtail *Rhipidura leucophrys*	6.44	1.65
Little Raven *Corvus mellori*	1.01	0.94
Magpie-lark *Grallina cyanoleuca*	4.96	2.43
White-winged Chough *Corcorax melanorhamphos*	NA	1.91
Silvereye *Zosterops lateralis*	1.17	0.79
Welcome Swallow *Hirundo neoxena*	6.44	2.23
Tree Martin *Petrochelidon nigricans*	6.1	1.27
Common Blackbird *Turdus merula*	0.65	0.9
Common Starling *Sturnus vulgaris*	1.45	1.55
Red-browed Finch *Neochmia temporalis*	1.69	1.88
European Goldfinch *Carduelis carduelis*	0.76	0.56

Figure 1. Spatial distribution of 2-ha, 20-min counts conducted by (a) BA and (b) NCSSA/UQ. (c) Location of the survey area within Australia.

overestimation in NCCSA/UQ data relative to BA data for that habitat type. Including birds detected outside the survey plot by NCCSA/UQ surveyors slightly weakened the relationship between the two datasets for both stringybark ($r^2 = 0.93$, $n = 61$, $P < 0.001$) and gum woodland plots ($r^2 = 0.91$, $n = 61$, $P < 0.001$). Moreover, the slope of the relationship for gum woodland became significantly greater than unity ($\beta = 1.173$, 95% CI = 1.021–1.325), although the relationship

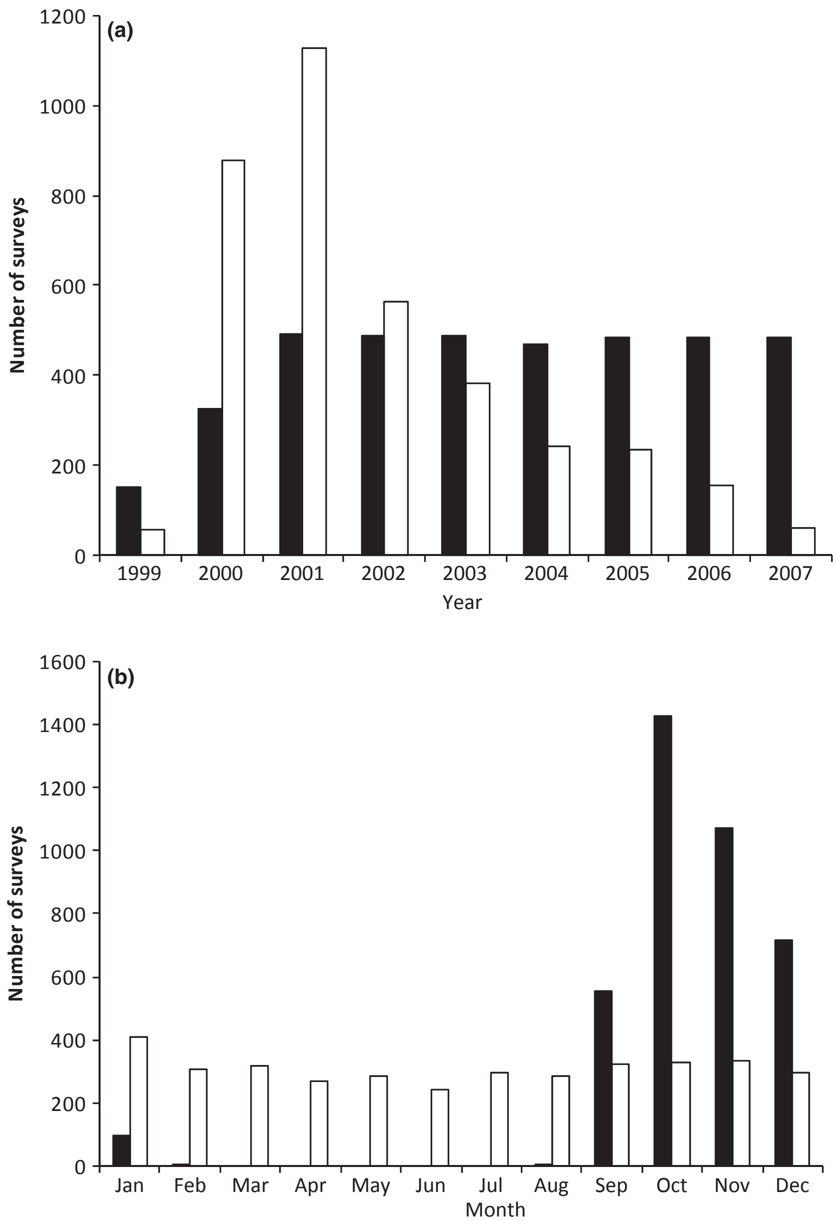

Figure 2. Temporal distribution of 2-ha surveys in the NCCSA/UQ (filled bars) and BA (open bars) datasets by (a) year and (b) month.

for stringybark woodland remained indistinguishable from unity ($\beta = 1.121$, 95% CI = 0.983–1.259). Thus, there was no evidence that BA surveyors were including large numbers of birds detected outside the survey plot. However, BA estimates were generally lower than NCCSA/UQ estimates when birds outside the plot were excluded from the latter, but higher when birds outside the plot were included, suggesting that a small number of individuals outside the plot may have been recorded in the BA data.

Whereas the overall concordance between the two datasets was very high, reporting rates from BA and NCCSA/UQ surveys differed at least two-fold in 22 cases involving 18 species (Table 1). Species with relatively high reporting rates in the BA dataset were mostly those not dependent on eucalypt woodland, and those using edge or open habitats such as Magpie-lark (see Table 1 for scientific names and taxonomy, which follows Christidis & Boles 2008), Noisy Miner and Welcome Swallow, suggesting a residual habitat effect in the BA

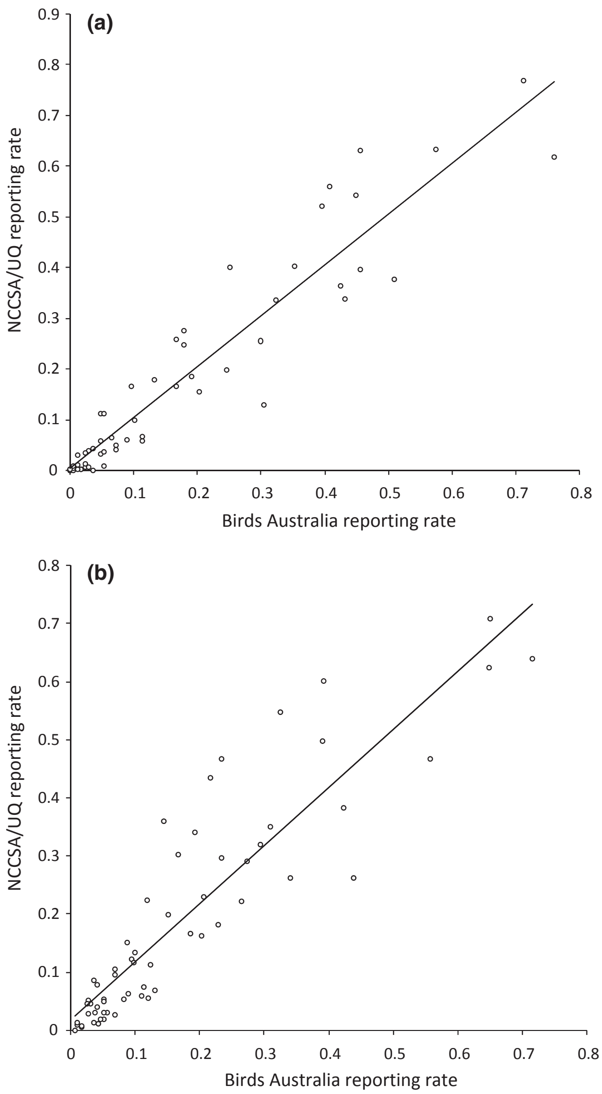

Figure 3. Comparison of BA and NCSSA/UQ reporting rates for 61 bird species in the Mount Lofty Ranges in (a) stringybark woodland ($y = 1.0051x + 0.0027$; $r^2 = 0.899$) and (b) gum woodland ($y = 1.0015x + 0.0166$; $r^2 = 0.817$). Values are the proportion of surveys in which each species was recorded, including birds physically using the 2-ha survey plot and those flying over the survey area.

data. Although data were restricted to surveys in eucalypt woodland using floristic mapping, NCSSA/UQ surveys were conducted > 50 m from the edge of habitat patches, where edge species were rarely encountered. Differences in reporting rates were limited and spread rather evenly between the two datasets for the woodland-dependent species (Table 1), with two-fold discrepancies in one or other of the habitats occurring for Sacred Kingfisher, Eastern Spinebill, Crescent Honeyeater, Horsfield's Bronze-cuckoo (relatively high reporting rates in NCCSA/UQ surveys), Brown Treecreeper, Dusky Woodswallow and White-browed Babbler (relatively high reporting rates in BA surveys).

The ratio between reporting rates in the two habitats provides a measure of habitat preference. These ratios were strongly related between BA and NCCSA/UQ datasets ($r^2 = 0.79$, $n = 61$, $P < 0.001$), with the two surveys agreeing which habitats had the higher reporting rate for 47 of the 61 species (Fig. 4).

Given the strong agreement in reporting rates between the two datasets, population estimates for native woodland-dependent species in the MLR derived from them were very similar (Table 2). The large discrepancy in reporting rate for White-browed Babbler in stringybark woodland did not cause a large difference in the minimum population estimates because the absolute reporting rate was very low in that habitat in comparison with gum woodland.

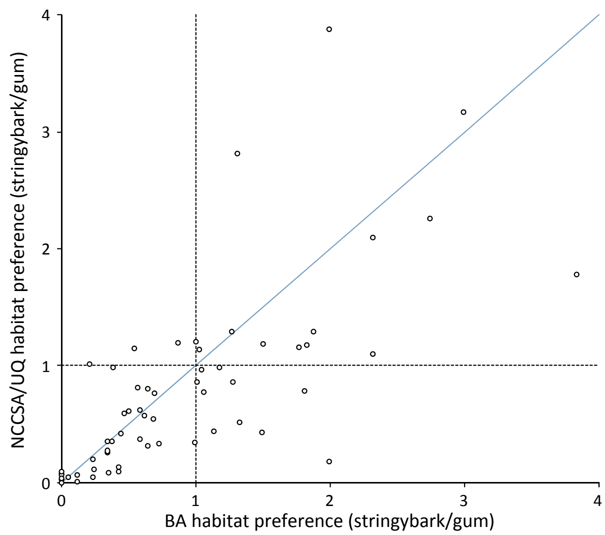

Figure 4. Ratios of reporting rates in the two habitat types. Species with a ratio below 1 (dotted line) had a higher reporting rate in gum woodland. Four species had higher reporting rates in gum woodland in the BA dataset but not in the NCCSA/UQ dataset (Sacred Kingfisher, Laughing Kookaburra, Common Blackbird and Crimson Rosella), whereas 10 had a higher reporting rate in stringybark in the BA dataset but not in the NCCSA/UQ dataset (Noisy Miner, Horsfield's Bronze-cuckoo, Spotted Pardalote, Eastern Spinebill, European Goldfinch, Crescent Honeyeater, Silvereye, Yellow-faced Honeyeater, Grey Currawong, Superb Fairy-wren).

Table 2. Population estimates for eucalypt woodland-dependent species in the Mount Lofty Ranges. Species are in taxonomic order following Christidis and Boles (2008).

Species	BA	NCCSA/UQ
Common Bronzewing	4982	7633
Horsfield's Bronze-cuckoo	1980	3046
Fan-tailed Cuckoo	2008	2012
Sacred Kingfisher	1911	1527
White-throated Treecreeper	16 190	22 074
Brown Treecreeper	2651	1010
White-browed Scrubwren	31 448	24 667
Chestnut-rumped Heathwren	294	273
Weebill	6078	6765
Striated Thornbill	58 065	94 873
Yellow Thornbill	2847	5029
Brown Thornbill	28 391	31 999
Spotted Pardalote	6784	6712
Striated Pardalote	67 395	75 273
Eastern Spinebill	11 205	20 926
Yellow-faced Honeyeater	42 825	62 310
Crescent Honeyeater	30 161	43 938
Brown-headed Honeyeater	6145	8717
Eastern White-naped Honeyeater	15 620	13 113
White-browed Babbler	4532	3961
Golden Whistler	13 647	19 152
Rufous Whistler	3602	4537
Grey Shrike-thrush	16 417	22 521
Dusky Woodswallow	3647	1362
Grey Fantail	29 484	39 765
Scarlet Robin	3383	6070
Mistletoebird	7860	12 549

DISCUSSION

We will first comment on the extent of the broad similarities between the two datasets and we discuss the fact that there is little systematic bias in reporting rates, then go on to highlight some of the species that depart from the general correlation between the two datasets and explore potential reasons for this. Unfortunately, we do not know true population densities of the species concerned and therefore we cannot determine the extent to which either survey reflects reality.

There was a good fit between species reporting rates from a regional survey based on a stratified random design (NCCSA/UQ) and a subset of national data collected by volunteers for a weakly structured national atlas programme (BA) in spite of significant differences in the original spatial and temporal patterns of sampling between the two datasets. We achieved a high level of concordance only by carefully trimming the BA data to match those obtained by using the more robust sampling strategy. Because the NCCSA/UQ surveys distinguished birds present in the survey plot, overflying birds and those outside the plot, we were able to assess how this influenced reporting rates. The concordance between the two datasets was strongest when overflying birds were included, and there was no evidence that birds outside survey plots were being reported in the BA data.

The two surveys broadly agreed about habitat preferences, although for some species there was strong disagreement (Table 1). In such cases, more targeted surveys will be necessary to allow confident assessment of habitat preference. Other species showing contradictory habitat preferences were not woodland specialists and so the disagreement was probably a result of the BA surveys including more edge habitat than the NCCSA/UQ surveys.

We derived population estimates from the two sources that are in good concordance. Even though the two datasets are confounded by using the same abundance multipliers based on the NCCSA/UQ data, we wanted to provide the first regional estimates for a group of birds that are in severe decline. As BA volunteers seldom collect abundance data, we treated their data as presence/absence, and used abundance information from the NCCSA/UQ. Although detectability of bird species in open eucalypt woodlands is probably relatively high compared with other forest types, we did not account for possible variation in detectability (Tyre *et al.* 2003). Hence our population estimates should be considered minimum estimates.

Survey effort among sites was standardized in the NCCSA/UQ study design (Possingham *et al.* 2004), whereas two of the BA sites were visited more than 100 times in the survey period, suggesting substantial redundancy given that species accumulation curves reach an asymptote after only a few visits (Possingham *et al.* 2004). Volunteers free to choose their study sites appear to oversample 'good' birdwatching localities and undersample the less productive but perhaps more representative places, a phenomenon known as convenience sampling (Anderson 2001). Oversampling of habitats favoured by birdwatchers can lead to misleading conclusions (Surmacki 2005), for example if BA volunteers preferred easily accessible sites, or places with unusual vegetation types, such as riparian vegetation, swamp and heath. Ease of accessibility could be associated with high detection rates for edge species such as New Holland Honeyeaters and White-plumed Honeyeaters. Rare species are often over-recorded in convenience sampling as a result of surveyors actively searching for them or visiting known sites (Robertson *et al.* 1995, Gibbons & Gregory 2006), such as surveys clumped around well-known sites for the Chestnut-rumped Heathwren. Observers also can be reluctant to report data when they see only common birds or no birds at all (Bonney *et al.* 2009). Temporal bias can also lead to erroneous conclusions about population sizes (Surmacki 2005). For example, reporting rates can be diluted if too many surveys conducted at times of the year when the target species are not present are included.

The BA dataset achieved a level of coverage of this large continent that would not be possible with standardized scientific surveys, and could potentially contribute to national biodiversity indicators. With unlimited resources, conservationists would seek to monitor everything, using standardized protocols with high temporal resolution, and therefore could minimize bias. However, well-designed and well-conducted long-term surveys and data analyses are expensive (Field *et al.* 2004). As a consequence of this trade-off, most studies that use appropriate survey methods are either short-term and of limited scale, or they focus on removing particular biases. If surveys are of larger scale and longer duration they usually have a less rigorous design. In countries with a sufficient density of birdwatchers, atlases can play a pivotal role in (1) mapping the distributions of species, (2) generating national population estimates and (3) tracking the distribution and abundance of species over time, especially as inputs into biodiversity accounting metrics for political use (Dunn & Weston 2008).

Regional and national bird atlases are costly in terms of effort and the number of people involved, yet they are underutilized (Dunn & Weston 2008). We emphasize that volunteer-based surveys should be designed as robustly as possible, but should be used to their full capacity, as for many areas they are often the only historical data available. For ease of analysis and interpretation, it would be desirable to make volunteer-collected datasets more rigorous in the future. However, taking away the freedom of choice from volunteers as to when and where to survey might diminish the attractiveness of the activity, especially if they are using their own resources without receiving compensation. Nevertheless, more representative sampling should be an aim not to be compromised. An alternative is to make the surveys more spatially and temporally dense. This has been done successfully in the USA (Kelling *et al.* 2009), but it is much more difficult for volunteers to conduct surveys in Australia for two reasons. First, inland Australia has a very low population density and, in this sense, is more similar to Africa or central Asia than to European or North American countries. Secondly, breeding times differ across the continent, from seasonal breeders to nomadic or wet-season-prompted breeding events (Keast 1959). In the arid zone, birds are more nomadic with less regular patterns of migration (e.g. Maron *et al.* 2005, Roshier *et al.* 2006, Ziembicki & Woinarski 2007). This makes planning the temporal pattern of sampling much more difficult than in temperate-zone countries with predictable annual cycles.

Using volunteer-collected data can be misleading if people who interpret them and use them for research or decision-making do not understand or consider the limitations and biases inherent in these datasets. It is crucial to check the quality of the data before using them for management decisions (Resnik 1998). The main difference between the two datasets was that one is sampled by a stratified design and the other one in an ad-hoc manner. However, there are other differences. As the NCCSA/UQ surveys were conducted by about 20 people, there were 150–200 BA volunteers, so we would expect more heterogeneity in the bigger sample of volunteer capabilities. We assume the average birdwatcher in both groups has about the same ability to detect birds, but in the bigger group we would expect both better and worse observers than the average. As we had no information about the abilities of the birdwatchers, we could not analyse this factor in this study. We would encourage a meta-analysis of similar datasets around the world to determine whether the results reported here stand for other ecosystems.

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