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Avian biodiversity monitoring in Australian rangelands

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Abstract Birds have been widely regarded as a key element in monitoring biodiversity both in Australia and elsewhere. We believe that, although birds are unlikely to be an umbrella or indicator taxon for other biota (other vertebrates, invertebrates, plants, microorganisms), they do represent a taxon that can be monitored more easily and with less effort per datum than other biotic components. It has been shown by the great participation rate of lay observers (whom we call monitors) in several schemes (notably the Birds Australia Atlas programs) that there is a capacity to mobilize the public to undertake bird surveying. Although there are many limitations to acquiring high-quality information (scale, dynamism, mobility, irruptiveness, paucity of monitors over much of the rangelands), we think that these can be dealt with to allow the use of birds as a key component of biodiversity monitoring. We outline some of the possible options for statistically characterizing monitoring data for rangeland birds.

Key words: decision-making, environmental pressures, impacts, reporting rates, statistical evaluation, time series.

INTRODUCTION

Birds are used extensively as a monitoring component for biodiversity status both in Australia and overseas (e.g. Australian Bird Count, Ambrose 1989; Christmas Bird Counts, Root 1988). Birds are also widely studied for their responses to changes in agricultural landscapes, particularly in terms of habitat loss, habitat fragmentation (Hobbs 1993; Brock & Jarman 2000; Mac Nally *et al.* 2000; Watson *et al.* 2000; Woinarski *et al.* 2000; Major *et al.* 2001) and habitat change (e.g. altered fire regimes; Franklin 1999). In the present study, we consider sources of data and statistical evaluations of change and status, but we do not intend this brief review to advise on survey techniques (there are numerous existing reviews; e.g. Recher 1989) or on specific survey designs. We do, however, consider some of the major issues involved in undertaking monitoring in sparsely settled rangeland areas and offer some advice on the way in which the monitoring might be undertaken and assessed.

BIRDS AND BIODIVERSITY MONITORING

Birds as general biodiversity indicators?

It seems increasingly unlikely that birds (or indeed any one taxon) will reflect overall biota-wide status (Howard *et al.* 1998; Ricketts *et al.* 1999; Andelman & Fagan 2000; Fleishman *et al.* 2001; Mac Nally *et al.* 2002). This is hardly surprising given that the mobility of birds is orders-of-magnitude greater than the mobility of other taxa (except, perhaps, bats and macropods; Bennett *et al.* 1998). This mobility can vary seasonally or with life stage. Many bird species make more use of resources in 3-D space than other animals. Spatial patterning of invertebrates is generally at a finer scale than that of vertebrates (Ferrier *et al.* 1999), including birds, and it is unlikely that birds will adequately reflect the status of invertebrates and many of the smaller, sedentary vertebrates (reptiles, small mammals; Mac Nally *et al.* 2002).

Certain taxa, such as birds, butterflies and large mammalian carnivores have sufficient charisma to attract the interest and involvement of many lay persons (see Andelman & Fagan 2000). The Birds Australia Atlas programs (I and II) (Blakers *et al.* 1984; Barrett *et al.* 2004) and the Birds on Farms (Barett & Davidson 1999; Barrett 2000) surveys show that lay persons can be engaged to collect monitoring data. Birds are generally regionally more species-rich than other vertebrates. This means that in many regions, vertebrate species richness will be dominated by birds. This is not to say that birds are ecologically or managerially more important than other vertebrates, but that overall trends in vertebrate species richness will be more likely to be affected strongly by trends in avian richness than by trends in other vertebrate classes. We believe that birds are substantially cheaper to survey to a given level of data accuracy than mammals or reptiles (see Mac Nally & Brown 2001 for an explicit example

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for reptiles). These points favour the usefulness and utility of monitoring birds in rangelands.

Sources of data: Scale and dynamism

Sources of data are contingent on the spatial scale at which the data are collected, which ties the data to locations in a more diffuse way as that scale increases, and on the dynamism of avian assemblages. The latter has a marked effect on the representativeness of information collected at a point in time. For example, species moving widely within short time intervals probably are unlikely to be picked up in small-scale surveys, irrespective of a species' abundance. More extensive spatial surveys probably will be more likely to detect such species. However, larger-scale surveys become less well-linked to specific point locations and, therefore, provide less information on the impacts of processes peculiar to those point locations.

Scale has become somewhat of a *bête noire* in ecological research and monitoring (Wiens *et al.* 1987; Mac Nally 2002, 2004; Wiens 2002). In rangelands, unlike in many other circumstances, there will be few circumscribed sampling units (e.g. lakes, Carpenter 1996; remnant woodlands, Mac Nally *et al.* 2000; Watson *et al.* 2000). Most often, what we choose to sample will be an arbitrary cut in a continuous spatial 'cloth'. For example, there were three fairly arbitrary survey 'scales' used in the Birds Australia Atlas program (II) (Barrett *et al.* 2004): (i) 2-ha counts; (ii) within-500-m radius counts; and (iii) within-5-km radius counts. These options implicitly point to a need to collect information at multiple scales because no single scale is sufficient.

Dynamism characterizes avian assemblages throughout Australia, becoming more pronounced as one moves from the coasts towards the centre. For example, a series of rangeland sites studied by the New South Wales National Parks and Wildlife Service (M. Ellis, unpubl. data) has been sampled on two occasions in different seasons, and the results indicate large variations in species occurrence on sites, even on consecutive days. Seasonal variations also are pronounced – generally less than two-thirds of the species found were detected in both seasons. These levels of variation are not peculiar to Australia. Studies of breeding-bird assemblages in 84 rangeland sites in Nevada, USA, showed that the average Jaccard similarity from 2001 to 2002 was just 41%, ranging from 14% (i.e. only two species of a total of 14 species recorded over both years were recorded in both years) to 69% (E. Fleishman, C. Betrus & L. Penfield, pers. comm.). Rangeland ecologists will be familiar with these levels of short-, medium- and longer-term variations in bird assemblages (although, for a contrary view, see Cody 1994).

DATA QUALITY

We always face limited funding for monitoring (Mac Nally 2002), which partly explains the attractiveness of engaging the lay community to conduct surveys such as the Atlases. However, data quality varies and is contingent on: (i) spatial extent of coverage; (ii) temporal intensity of surveys (number of surveys per season or per annum); (iii) observer bias and experience; (iv) species' idiosyncratic characteristics, affecting detectabilities; and (v) rarity and irruptiveness.

The Australian rangelands cover a vast area (approximately 70% of Australia), have poorly developed road networks (perhaps thankfully; Forman *et al.* 2002) and are sparsely populated. These are obviously important issues, and rangeland areas are often so sparsely populated by monitors that standard survey regimes are unlikely to provide adequate information for analysis. It is relatively easy to show how influential temporal intensity is in shaping the view of the local fauna, especially for animals as mobile as birds (Colwell & Coddington 1994; Mac Nally & Horrocks 2002). Rare species are always problematic, often being the species of most conservation concern and also those requiring the most intensive survey effort and methods to be detected adequately (Rabinowitz *et al.* 1986; Gaston 1994; Garnett *et al.* 2003). Assessment of the spatial and temporal patterns of irruptive species (e.g. Watson 1995) is almost as difficult as monitoring rare species.

It is important to consider the framing of an 'objective function' that summarizes the relative importance of different components of the fauna and the accuracies that one is willing to accept, given that survey funding is limited and survey expertise sometimes is hard to hire. There is always a trade-off between spatial coverage (i.e. number of point locations) and temporal intensity of monitoring (i.e. number of repeated surveys at individual points). However, unlike in many temperate parts of Australia, distances travelled to reach individual point locations in rangelands are often very great, so it makes sense to conduct longer surveys (i.e. for a greater period of time) at fewer points because the costs (both temporal and monetary) of getting to locations in much of the rangelands are relatively great. We also may have to accept that the possibilities of obtaining adequate information for analysing trends for sparsely distributed and rare species are virtually nil and we may be forced to admit that such species will not figure in a monitoring appraisal of the region in question.

Tiers of data quality from lay-community sources

Involvement of a wide cross-section of the human community is likely to be required if large areas of the rangelands are to be monitored. This was proven to be possible by the Birds Australia Atlas programs, but would require long-term dedication to the process so that coordination and effective use of data occurs. Given the need to expand the involvement of lay persons (including land-holders) in avian monitoring in rangelands, it would be useful to use several tiers of information depending on the investment that a monitor makes. For example, a participating land-holder might choose one of three possible tiers, according to his or her commitment and/or perceived returns. Three possibilities may be as follows:

1. Asystematically collected annual list of birds observed on the land-holder's property.
2. Asystematically collected accumulated records of birds seen over the year, possibly stratified according to habitat type.
3. Seasonal 2 ha/20 min counts conducted by contractors or accredited monitors (e.g. persons from field-naturalist clubs) of habitat-stratified surveys.

Variations of such tiers are possible, and in the future 'green accreditation' for the produce of the property might be linked to the conduct of such surveys, with greater credit being attached to the tiers generating highest-quality data.

STATISTICAL ANALYSES: CAUSALITY AND CHANGE-DETECTION

Many monitoring programs are designed to estimate the ecological effects of known, well-specified impacts (Downes *et al.* 2002). Many of these also are well-circumscribed in space and time. Many analysts would see the strongest inferences about the effects of human impacts as being derived from datasets in which there are replicated measurements for locations in which there have been demonstrable (and similar) impacts and locations in which those impacts have clearly not occurred (Downes *et al.* 2002; §5.2.2). For the strongest inference, these two sets of data ideally would be randomly allocated to these populations (Underwood 1996). In rangelands, most of these conditions are not likely to be met, so we must consider options in which one or more of the three (control or reference, replication, randomization) are not satisfied.

Rangeland pressures are likely to be spatially diffuse and to vary in intensity (Landsberg *et al.* 1999; Ludwig *et al.* 1999; Hoffmann 2000). This, combined with the dynamism of birds, adds complications because the possible effects of impacts might not be seen quickly or close to where those impacts are most critical (i.e. referred effects; Robinson & Traill 1996). Thus, many statistical analyses for monitoring are designed to attribute causality to impact effects, whereas rangeland monitoring probably will be more suited to detecting change per se. Tukey (1960) dealt long ago with the logical difference between deriving a strong inference or conclusion in a scientific context and the greater flexibility needed to trigger actions in monitoring and management. For example, there was some confusion between attributing cause and detecting change when environmental scientists inculcated on management institutions the reasoning behind assessing statistical power in environmental monitoring (Fairweather 1991; Hoenig & Heisey 2001). These arguments are less compelling when one is interested more in detecting change (direct inference on model parameters) than on the statistical significance of a change and attribution of cause.

TREND ANALYSES

Kinds of data determine pertinent statistical models

The nature of the data dictates the kinds of analytical models that are most appropriate. For example, presence data might be treated as a logit-transformed variable typically used in logistic regression. Presence data may be aggregated for use in a binomially distributed analysis that may be appropriate for reporting rates (RR; McCullagh & Nelder 1989). Abundance or relative abundance data usually fall into conventional normal or Gaussian linear modelling. Consistently small integral numerical values might be better fitted by using Poisson-distributed models.

We now consider several of the main candidates for assessing temporal sequences of data. Although we speak of impacts and their statistical identification, we more generally think of an impact as a point in time that separates two sets of information and about which we wish to compare the pre- and post-impact datasets. In this sense, the exact nature of the impact(s) may be unspecified and may need to be inferred secondarily. The two Birds Australia Atlases constitute such sets.

Before–after control-impact-like designs

A codified approach to human-impact monitoring involves variants of the 'before–after control-impact' or BACI designs (Underwood 1993). In this design, site randomization is not usually possible but the comparison of reference and impact locations following the application of known perturbations is central to the method. Sometimes, multiple reference locations are used but multiple impact sites might not be available.

Time-series analyses

Given its importance to economic, environmental and climatic trend analyses, there is now an immense amount of work published on time-series analysis (Box & Jenkins 1976; Hirsch *et al.* 1982; Box & Kramer 1992). Statistical inference on autoregressive–integrated moving-average (ARIMA) models is sophisticated (Harvey 1981), but seemingly not sufficiently useful for ecologists to use extensively. Conventional time-series analysis is so assumption-ridden (e.g. series stationarity) that detection of the kinds of changes (e.g. sudden density declines) we might be looking for requires substantial analyst intervention when building models. ARIMA-like models also have long-lasting memories of abnormal or aberrant values, which compromise their effectiveness.

Generalized additive models

The generalized additive model (GAM) is a flexible method in which trends are handled by using smoothing splines. GAM are being employed increasingly for trend analyses of environmental data, especially of physical and chemical quantities. For example, Jolly *et al.* (2001) used GAM to analyse time-series trends of water-quality variables. Output from the public-domain program 'gaim' (T. Hastie and R. Tibshirani, unpubl. data) includes measurements of a linear trend within the smoothing spline and an estimate of the statistical significance of this trend and also of the non-linear component of the spline. It is possible to include a dummy variable indicative of pre- and post-impact sets.

Control charts

The manufacturing industries have used control charts extensively since World War II. There are several methods by which industrial process monitoring has been approached (Rowlands & Wetherill 1991). A method devised in the 1950s – and continually refined since – is the use of cumulative sum (CUSUM) charts. Although regularly cited in handbooks and textbooks of water-quality monitoring (US Environmental Protection Agency 1972; Hipel & McLeod 1994; Newman 1995), there seems to have been little use of CUSUM in biological monitoring, but Kshatriya *et al.* (2001) have described a modelling framework with similar objectives.

Intervention analyses

Another approach that capitalizes on situations in which the precise timing of impacts is known is intervention analysis (Carpenter 1993). The idea here is simple: have a long sequence of pre-impact values and a long sequence of post-impact values for both impact and reference locations; in both, construct a statistic that encapsulates the difference between the pre- and post-impact datasets (perhaps the difference in means); then, randomize the dataset sequences and compute the same statistic for the randomized sequences. This is done many times, producing a distribution for the statistic against which the probability of the observed difference is assessed. If the impact is ecologically important, then the value of the observed statistic in the impact dataset should be improbably small (e.g. most of the larger values in the pre-impact series and most of the small ones in the post-impact series), and the value for the reference location should be close to the middle of the generated distribution for those data.

Reporting-rate models

All of the preceding approaches assume that the data are abundance data. There are many situations in which abundances are not recorded, or there are doubts about the accuracy of those abundance estimates. We might be satisfied that monitors can adequately identify species, but not that their measurements of numbers are reliable (i.e. monitors can identify most species well but might find it difficult to estimate numbers of individuals in flocking species accurately, such as the budgerigar *Melopsittacus undulatus*). However, if there are many separate surveys per location, it may be possible to treat the presence data as a binomially distributed variable with changes in RR taken as evidence of increases or decreases in abundance. Say one had B_i surveys in a spatial domain i before a known impact and A_i surveys afterwards. Given certain assumptions (e.g. a fixed mean occurrence rate μ before impact at all locations and a fixed post-impact mean rate common to all locations), if the species is reported in b_i pre-impact surveys and in a_i post-impact surveys in site i, then the impact (λ say) might be estimated by looking at the mean difference in μ_b = average (b/B) and μ_a = average (a/A) across sites. If reference locations are available, one hypothesizes that $(\lambda_{impact} - \lambda_{reference}) \neq 0$.

Birds Australia has conducted two major surveys of the distribution and abundance of birds in Australia, as well as more specific studies. An example of the use of Atlas data is where the RR in geographical units (e.g. Interim Biogeographic Regionalization for Australia (IBRA) regions, Thackway & Cresswell 1995; degree-square) are regarded as components of a meta-analysis of occurrences by geographical unit. For example, there may be 350 sheets submitted to the first Atlas in a certain IBRA region, with one species (e.g. the Australian pipit *Anthus novaeseelandiae*) being recorded on, say, 115 of those sheets. In the second Atlas, the figures may be 460 sheets and 65 occurrences. Analogous figures can be compiled across all IBRA regions. If we assume a common general change across all units, and that the sheets within a region are effectively sets of binomial trials, then the data can be analysed by either frequentist or Bayesian (Carlin 1992) approaches. A Bayesian analysis for the pipit (R. Mac Nally, unpubl. data) suggests that across all IBRA regions, the multiplicative change in the odds-ratio for RR between Atlases is 0.47 with a 95% credible interval of 0.40–0.55. There were 45 (of 76) IBRA regions in which there were substantial decreases in RR, defined as the difference, $\lambda_{\text{Atlas I}} - \lambda_{\text{Atlas II}}$, in which the upper 97.5-percentile < 0. Six IBRA regions (the Flinders and Olary Ranges, South Australia; Riverina, Victoria–New South Wales; the Avon Wheatbelt, Western Australia; Gascoyne, Western Australia; the Simpson–Strzelecki Dunefields, South Australia–Northern Territory; and the Eyre and Yorke Blocks, South Australia) had declines RR of >0.20. In only one IBRA region, the central Mackay Coast, Queensland, was RR substantially greater (+0.095). Analysis of degree-square data produced a similar picture, with the multiplicative change in in the odds-ratio for RR between Atlases being 0.39, with a 95% credible interval of 0.35–0.44.

MODUS OPERANDI

The key considerations for avian monitoring in rangeland areas are as follows. First, there generally will be few monitors, and the spatial coverage will be sparse. Reaching many parts of the rangelands takes a long time, so optimizing effort when monitors reach a location is desirable. Engagement of land-holders is a prudent way of increasing spatial (and temporal) coverage, albeit with qualifications on data quality. Second, many areas are very inhospitable for much of the year, with hot, arid summer conditions and, in some areas of northern Australia, long monsoonal rain periods, both of which make surveying difficult and odious. Third, habitats generally are open, so that identification of birds is relatively straightforward compared with densely forested habitats, such as in south-eastern Australia. Use of all information, visual, aural and incidental (e.g. vacant, recently used nests, shed feathers) is warranted even at the expense of the usual strictly defined consistency of sampling. Fourth, assemblages are often highly variable at local scales (e.g. several ha), featuring irruptions and wide-ranging flocking behaviour. Thus, the relatively restricted 2-ha/20 min method that has been widely used may be less efficient than broader-scale survey methods. The second Birds Australia Atlas program (Barrett *et al.* 2004) offered a survey option covering an area within 500 m of a central point within one day (up to 78 ha), which may be a good choice for further survey work to alleviate the limitations of the 2-ha/20 min method. This also provides a better trade-off in terms of gathering more data once a particular location is reached. Specific provisions may need to be made for gathering information from locations that are distant from established roads, and other difficult-to-reach locations. Abundance data are likely to be unreliable in many cases compared with presence data, so RR comparisons based on presences per sheet should probably underlie analyses. Bayesian analyses of RR comparisons that characterize deviations in the RR parameter, λ, from particular values seem appealing. For example, me might fix the acceptable RR to be $\lambda_{\text{Atlas I}}$, and use Bayesian updating to estimate the posterior-probability mass (p.p.m.) distribution for the parameter $\delta = (\lambda_{\text{current}} - \lambda_{\text{Atlas I}})$, where λ_{current} is the continuously updated value for RR for the species in question. If much of the p.p.m. for δ lies below zero, then we would have concerns that the RR is lower than the reference and an alarm should be raised. The choice of reference RR (here $\lambda_{\text{Atlas I}}$) might refer to a specific region (e.g. IBRA or degree-square) or more broadly, although sufficient data for the latter are likely to accrue at slower rates than for individual regions. The choice of reference RR is a difficult one and probably needs to be established by considering trends for the species between the Atlases and with a view to the status of the species (e.g. in conjunction with The Action Plan for Australian Birds, Garnett & Crowley 2000). Bayesian approaches are useful because of the focus on parameter estimation, continuous updating, logical consistency (Jaynes 2003) and capacity to handle multiple sources of uncertainty (Bernardo & Smith 1994).

CONCLUSIONS

Birds are an appropriate component of biodiversity to monitor in Australia's rangelands. Collectively, birds are a species-rich taxon, are usually relatively easily distinguishable to species level to a greater extent than many other taxa, and most are diurnally active. Other taxa, such as mammals and reptiles, typically require special and usually much more expensive survey methods (Mac Nally & Brown 2001), making broad-scale monitoring difficult. Many lay persons find birds to be engaging and are concerned about decreasing local diversity and abundance of the avifauna (e.g. the Birds on Farms participation). A huge advantage is that there is no secondary laboratory processing, a factor that plagues the use of invertebrates in general, in which one afternoon's collection might take 3 years to process. There are very extensive and generally broadly distributed datasets stemming from the Birds Australia Atlas programs and other schemes. Birds are typically at high trophic levels (as carnivores and invertebrate feeders), and although their high mobility can be construed as negative – making localization of specific stressors difficult – this mobility also means that birds are integrators of ecological conditions over potentially vast areas. It is likely that land-holders and skilled lay personnel will be needed to make monitoring more extensive for terrestrial bird species.

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