Biological Conservation 226 (2018) 153–167

Contents lists available at ScienceDirect

Biological Conservation

journal homepage: www.elsevier.com/locate/biocon

Evaluating spatiotemporal trends in terrestrial mammal abundance using data collected during bird surveys

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A R T I C L E I N F O

Keywords:
Citizen science
Mammals
Population trends
Species distribution models

A B S T R A C T

Information on the status of biodiversity is crucial for species conservation and management. Large scale assessments are only feasible through citizen science but some taxa are poorly monitored because few people specialise in them. We explore alleviating this problem by using data collected for poorly monitored species as an add-on to existing bird surveys. Since 1995, participants in the annual Breeding Bird Survey have recorded the abundance of mammals during their surveys. We demonstrate the value of these data by developing spatial models of relative abundance for nine common and easily detected mammal species. Rabbit, brown hare and mountain hare all showed widespread declines. Conversely, deer showed increases throughout their ranges, with the exception of the red deer whose population was predominantly stable. The grey squirrel continues to increase in several areas. The red fox, the only carnivore with enough data, showed significant large declines. The collection of data on taxa other than the primary target has particular merit where the secondary taxa can be detected effectively by methods devised for the core survey. In such cases the data are inexpensive and inherit some of the benefits of the underlying structure and power of the core survey. However, the efficacy of the primary study design may vary for the members of secondary taxa and may not be temporally or spatially suitable for all of them. Although more volunteer training may be required, there are also opportunities to engage and enthuse people about conservation issues of other species groups.

1. Introduction

Knowledge of population abundance is central to conservation, sustainable harvesting and pest management (Shea et al., 1998) and is a proposed Essential Biodiversity Variable (Chandler et al., 2017). Attributes such as range size and total abundance and their temporal trends are used for assessments such as regional and global red listing exercises (Rodrigues et al., 2006; Eaton et al., 2005) and are required by statutory authorities for evaluating performance against sustainability targets (e.g. Conservation of Habitats and Species Regulations 2010). Many countries also have a requirement to monitor the status of non-native species. The bulk of this core monitoring work is typically achieved through volunteer-based schemes that provide annual or periodic information on occurrence or abundance that is used to generate trends (Toms and Newson, 2006; Hayhow et al., 2017; Rosenberg et al., 2017). Such schemes are very effective for taxonomic groups with large numbers of passionate observers (e.g. birds) but robust insights are more difficult for species groups with fewer specialist surveyors.

Mammals present particular challenges for monitoring: they are ecologically diverse, varying in size and activity leading to substantial differences in detectability. There is no single survey technique that is adequate for all (or even most) species and some are difficult to identify in the field. The UK's mammal community comprises 51 terrestrial mammals (The Mammal Society, 2018), which include several of conservation concern, such as brown hare (*Lepus lepus*) and red squirrel (*Sciurus vulgaris*), that have declined in recent decades (Battersby, 2005). There are also several introduced non-native mammals, some of which are known to cause problems for native species or to cause economic damage (e.g. grey squirrel (*Sciurus carolinensis*), Reeves's muntjac (*Muntiacus reevesi*), fallow deer (*Dama dama*), although the latter may be considered native as they went extinct in Britain during the last ice age; Harris and Yalden, 2008; Newson et al., 2010, 2012; The Mammal Society, 2018). A further group including red fox (*Vulpes vulpes*), mustelids and rodents are controlled as pests or disease vectors, and may cause issues for the protection of other species of conservation concern (Tapper et al., 1996; Douglas et al., 2014).

UK mammal monitoring is complex and is characterised as dispersed across many organisations, taxonomically incomplete, of varying spatial and temporal coverage, of varying quantitative refinement, incompletely reported and with varying engagement with

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https://doi.org/10.1016/j.biocon.2018.07.026
Received 17 January 2018; Received in revised form 19 July 2018; Accepted 20 July 2018
0006-3207/

volunteers (Macdonald et al., 1998; Toms et al., 1999). Although population trends of some mammal species are estimated using data from various bespoke schemes, such as the National Bat Monitoring Program (Bat Conservation Trust, 2018) or the National Gamebag Census (Aebischer et al., 2011), a comprehensive mammal monitoring scheme has yet to be developed in the UK. One way to improve data for poorly monitored species groups is to ask participants in other schemes to report sightings of under-reported taxa. One of the few examples of this approach is the collection of data on the relative abundance of mammals by participants in the UK Breeding Bird Survey (BBS; Harris et al., 2016). Starting in 1994, the BBS was designed to monitor widespread breeding birds across the UK. In 1995 it was extended to include mammals and has generated UK-level trends since then for nine mammal species. Although not the original focus of the scheme, many of the more detectable terrestrial mammals are easily identified and volunteers have provided reliable, consistent data on mammals (Newman et al., 2003). BBS methods do not provide absolute measures of abundance for any of the mammal species, and their efficacy may vary among species. Yet the fact the BBS uses a stratified random sample of almost 4000 1-km squares throughout the UK, with mammal data collected in a consistent manner across c.90% of these squares, means the survey can provide unique insights into likely population changes to complement intensive studies using bespoke methods. Mammal trends are estimated every year from BBS data by modelling counts at square level, as a function of square and year. Trends are produced for the UK as a whole, for the four individual countries and for nine regions within England, and are presented in an annual report (Harris et al., 2016).

In this paper we explore the benefits gained by adding mammal monitoring to an existing structured bird monitoring scheme by developing maps of spatial variation in relative abundance of mammals for different periods and, by comparing these, produce maps of change in abundance. Identifying where populations are stable versus declining is helpful when identifying locations for management interventions, or to assess where existing interventions are failing. Spatial variation in trends is increasingly recognised and may be helpful in identifying the underlying process causing change (Channell and Lomolino, 2000). For birds, such maps have been available for many decades and have been invaluable in providing greater insights into status than can be obtained from simple presence/absence maps (Gibbons et al., 1993). We show that spatial modelling of relative abundance can reveal hitherto unknown gradients in abundance and abundance change of value for management of mammal populations. We discuss the merits of asking bird observers to count mammals and consider the wider application of this approach to monitoring under-studied taxa.

2. Methods

2.1. Field surveys

The Breeding Bird Survey (BBS) is an extensive volunteer survey used to monitor the population changes of the UK's common breeding birds, and since 1995 volunteers have also provided information on mammals. The survey design employs a regional stratification to allow coverage to vary geographically in a planned manner to capitalise on larger volunteer pools in different areas of the UK. In each of 83 strata defined by administrative boundaries, the number of 1-km squares that were randomly selected was proportional to the number of potential volunteers (Gregory and Baillie, 1994). The difference in sampling effort across strata is taken into account by appropriate weighting when calculating the population trends. The BBS currently covers almost 4000 1-km squares (Harris et al., 2016). Field surveys are carried out twice per year, four weeks apart, between April and June. They start at around 6 am and normally last less than 2 h.

Within each square, volunteers walk two 1-km transects on each visit. Ideally, the two transects run parallel to one another, no closer than 500 m apart and 200 m from the edge of the square. In practice, the transect lines often deviate depending on access or terrain, but should never be closer than 200 m or intrude significantly into an adjacent square. As well as counting the birds encountered, volunteers can provide two different levels of information about mammals on these visits: the number of live mammals seen during the early and late BBS visits, and notes of any signs of mammals (tracks, scats etc.). Importantly, volunteers are also asked to submit ‘nil returns’ for visits where mammals and mammal signs were looked for, but none were seen. Additionally, any mammals seen in the square by the observer during additional visits can also be entered under ‘Additional visits’ and records deriving from local knowledge (e.g. in discussion with land owners), can also be submitted. Even though mammal recording has always been a voluntary addition to the scheme, mammal data have been collected on 80% of all BBS squares (average over 1995–2015).

2.2. Statistical analysis

To derive maps of relative abundance and change we followed an approach already successfully applied to bird count data from the BBS (Massimino et al., 2015). In outline this involved producing two distribution models for each species, relating the abundance of the species in an early period (1995–1999) and in a late period (2011–2015) to environmental variables. We then compared the resulting predicted abundance maps to identify geographical patterns in the magnitude of change. Each period spanned five years to reduce the influence of chance non-detections in occupied squares and to dampen the influence of population cycles.

We produced models for terrestrial mammal species with at least 100 occupied squares in both periods, excluding common shrew (*Sorex araneus*) and stoat (*Mustela erminea*) that were sufficiently commonly reported but were likely to have extremely low detection probability or identification issues such that we could not be confident that emergent trends were robust.

For each species and for each square, we extracted the maximum count per square on a single survey occasion within a year. As we were modelling abundance, we did not use other types of information on mammal presence (such as signs and local knowledge) which could not be easily converted into a measure of abundance. We then averaged maximum counts for each five-year period and we used this average as the response variable in the species abundance model. We then modelled species abundance using Generalised Additive Models (GAMs), specifying a logarithmic link function and Poisson error structure. Covariates were: (a) the percentage cover in the 1-km square of seven land cover classes (broadleaved/mixed woodland, coniferous woodland, mountain/heath/bog, improved grassland, semi-natural grassland, arable land, and built up area) and their quadratic terms, derived from the Land Cover Map 2000 (Haines-Young et al., 2000); (b) mean elevation of the 1-km square and its quadratic term from a digital elevation model (United States Geological Survey, 1996); (c) a two-dimensional (easting, northing) thin plate penalised spline to account for spatial patterns not linked to other covariates, as nearby areas were more likely to have similar abundance; and (d) a categorical variable taking a different value for each island in the UK to account for