Trends in the use of private gardens by wild birds 1995-2002

Journal of Applied Ecology 2005
42, 659–671

Trends in the use of private gardens by wild birds in Great Britain 1995–2002

ANDREW R. CANNON,* DAN E. CHAMBERLAIN,† MIKE P. TOMS,† BEN J. HATCHWELL* and KEVIN J. GASTON*

**Department of Animal and Plant Sciences, University of Sheffield, Sheffield S10 2TN, UK; and †British Trust for Ornithology, The Nunnery, Thetford IP24 2PU, UK*

Summary

1. Wild birds are commonly observed in private residential gardens in Great Britain. However, little is known about how their use of this significant and increasingly important habitat is changing and how such changes relate to their population status.
2. Trends in the use of private residential gardens by wild birds in Great Britain were investigated using weekly bird records from 18 300 gardens over 8 years.
3. We showed that the use of this habitat is seasonal and cyclic, with the timing and regularity of its periodicity variable between species.
4. We evaluated the significance of the underlying trends in the cyclic reporting rates. Eighteen species showed clear trends, the three with the most negative year term parameter estimates being 'red-listed' as high conservation concern.
5. Examining correlations with national scale survey data suggested that garden reporting rates are related to general population trends in a number of species, including several of conservation importance. Other species exhibit important differences between national and garden trends.
6. *Synthesis and applications.* Our analysis demonstrates ecologically meaningful trends and provides novel insights into seasonal cycles of habitat exploitation, using relatively simple and cost-effectively collected data. This will lead to greater understanding of the relationships between gardens and general bird populations and of the times of year at which garden habitats are most important for birds. We have demonstrated the practicality and productivity of 'citizen science' in this context, and provided new information on the status of some birds of conservation concern.

Key-words: monitoring, periodic logistic regression, population trends, reporting rates, urban greenspace

Journal of Applied Ecology (2005) **42**, 659–671
doi: 10.1111/j.1365-2664.2005.01050.x

Introduction

In Great Britain private gardens contribute a significant proportion of the total habitat available to birds, garden ownership being a traditional and widely held aspiration. Owen (1991) estimated the total area of gardens in England and Wales as 485 000 ha or 3% of the total land area. Based on a net annual increase in residential land use of 5000 ha year^{-1} in England (ODPM 2003), and assuming around one-third of such development is garden, gardens may cover as much as 500 000 ha of England and Wales today. This compares well with the approximately 120 000 ha of national and local nature reserves in England (English Nature 2004) and 115 000 ha of Royal Society for the Protection of Birds (RSPB) reserves in the UK (RSPB 2002). Human residential habitats in which gardens are the primary resource support a significant proportion of the national populations of a number of wild bird species (Gregory & Baillie 1998), perhaps more important than previously suspected (Bland, Tully & Greenwood 2004), and appear to be refuges for some declining species, such as the song thrush *Turdus philomelos* CL Brehm (Mason 2000; Peach *et al.* 2004). Although most typical gardens support only a reduced avifauna as a result of a variety of factors, including high levels of disturbance and predation, lack of nesting cover and

Correspondence: Andrew Cannon, Biodiversity and Macroecology Group, Department of Animal and Plant Sciences, University of Sheffield, Sheffield S10 2TN, UK (e-mail andrew.cannon@shef.ac.uk).

predominantly alien plant species, some garden bird species are effectively subsidized in this habitat by artificial feeding and provision of nest boxes (Cannon 1999; Beebee 2001). Despite the recent upsurge in urban ornithological activity (Marzluff, Bowman & Donnelly 2001), most research has remained focused on larger green spaces, such as parks, and gradient studies. However, in many cities the overall area of private gardens may be very extensive; for example, gardens cover an estimated 23% of Sheffield, England (Gaston *et al.* 2005) and an estimated 27% of Leicester, England (Jeffcote 1993). Questionnaire data suggest that Sheffield's private gardens contain 25 000 ponds, 350 000 trees and 45 000 bird nest boxes (Gaston *et al.* 2005), a very significant habitat resource.

Gardens are amenable to detailed year-round monitoring by volunteers; this represents an obvious opportunity to increase public engagement in bird conservation and obtain data on a population that is otherwise inadequately monitored despite its potential conservation importance. Volunteer garden bird surveys have been attempted in a number of countries (Cannon 1999). Since 1970, in the UK, around 250 volunteers have recorded exact numbers of birds using feeding stations over the winter period for the British Trust for Ornithology (BTO) Garden Bird Feeding Survey (Toms 2003; Chamberlain *et al.* 2005), which offers a unique, long time series of feeder-use data, but it is limited by small scale and issues with modelling the free-format data. At the other extreme of scale, Project FeederWatch collects data from thousands of volunteers across North America, demonstrating continental-scale movements as well as trends (Wells *et al.* 1998), but unfortunately only in the winter half of the year. The Canberra bird survey (Veerman 2002) exemplifies the very comprehensive data available from smaller geographical scales, but resources generally limit this intensive approach to localized survey areas. We believe that the BTO/CJ Wildbird Foods Ltd Garden BirdWatch (GBW) in the UK is presently the only garden bird survey collecting systematic weekly data from thousands of sites at a national scale and throughout the year (Cannon 2000; Toms 2003). In this study, we used the first 8 years of data from GBW to determine seasonal patterns of garden usage and their variation between species, and the occurrence or otherwise of interannual directional trends and other temporal variations. By comparing GBW reporting rates with national population indices, we investigated the extent to which changes in garden usage by bird species reflect or differ from patterns in overall population levels, and consider the implications for species of conservation concern.

Methods

DATA COLLECTION: THE BTO/CJ GARDEN BIRDWATCH

GBW is an open-access project funded by volunteers' subscriptions; participation has grown from around 5000 sites in January 1995 to more than 16 000 at the time of writing (2004; (Fig. 1). We have used all available data for the years 1995–2002 in this analysis. The data are incomplete longitudinally as sites may join or leave the project at any time. Figure 2 shows the geographical distribution of GBW sites at the time of writing. Despite inevitable concentration in the areas of highest human population density (notably the south-east of England), the survey covers the whole of Great Britain reasonably well. Presence–absence data are collected weekly using a standard checklist of 40 bird species. This method overcomes the zero records problem observed by Rushton, Ormerod & Kerby (2004), in that absence records can be confidently interpreted as zeroes. Carrion crow *Corvus corone* L. and hooded crow

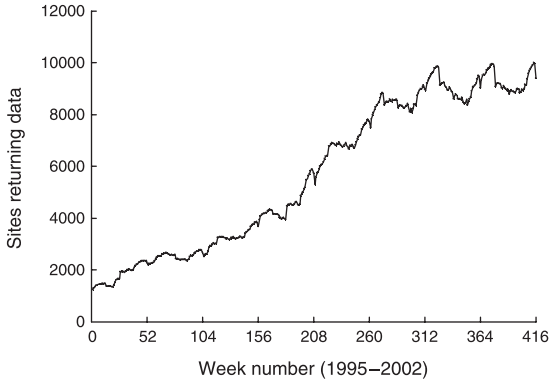

Fig. 1. Numbers of sites contributing data to the analysis each week. Week numbers 1–416 run from the start of 1995 to the end of 2002. Winter peaks in recruitment are the result of seasonal promotional activities; abrupt drops reflect the fact that survey forms are returned quarterly and a proportion of new recruits do not record for more than one quarter.

Fig. 2. Distribution of GBW sites in Great Britain, November 2003. Mapped using DMAP (www.dmap.co.uk).

Corvus cornix L. are recorded as a single species, as few hooded crows visit gardens and in this habitat the ecological role of the two is similar. Volunteers are instructed to define their own consistent 'study area' within which if a species is observed at any time during a week it will be recorded as 'present' for that week. No restrictions are imposed on the type of garden observed. Almost all volunteers provide artificial food of some kind and feeding stations are generally the focal point of the study areas. Volunteers are asked to maintain a consistent observation effort each week and encouraged to discard data from under- and over-observed weeks. Some variation in observer effort and competence is inevitable; however, we contend that this does not detract from the ecological interest of these data when examined at a national scale and over an 8-year time scale. The full instructions provided to volunteers are available at www.bto.org.

DATA ANALYSIS AND MODELLING

For each of the 40 species and 416 weeks, we calculated a weekly reporting rate, i.e. the proportion of the sites returning data that week at which the species was recorded. Using the procedure GENMOD in SAS, we fitted a generalized linear model (GLM) with a logit-link function and binomial error distribution (logistic regression) appropriate to the dichotomous dependent variable count, which has values of either 'present' or 'absent' for each species and week (Allison 1991). This procedure estimates the logarithm of the odds of occurrence in week i, $\ln(p_i/1 - p_i)$, where p_i is probability of occurrence in week i. A notable feature of the GBW reporting rates is a strong cyclic variation with a period of 1 year. The objective of the modelling being to examine longer term trends underlying this periodicity, we followed Flury & Levri (1999) in accounting for it in the model by using trigonometric terms. The initial form of the model for all species was:

$$\ln(p_i/1 - p_i) = \theta \qquad \text{eqn 1}$$

where

$$\theta = \alpha + \beta_1(\text{year}) + \beta_2(\text{year}^2) + \beta_3(\text{cosweek}) + \beta_4(\text{sinweek}) + \beta_5(\text{cosweek} \times \text{sinweek}) + \beta_6(\text{sinweek} \times \text{year}) + \beta_7(\text{cosweek} \times \text{year})$$

where α is the value of the GLM intercept returned by GENMOD and β_{1-7} are the GLM parameter estimates for each term. The term year took integer values between 1 and 8, representing 1995–2002; cosweek and sinweek were the cosine and sine, respectively, of the week number (1–52) within each year. Likelihood ratio statistics for each term were obtained using the TYPE3 option in GENMOD, terms for which $P_r > \chi^2 > 0{\cdot}05$ were successively removed from the model until all remaining terms were significant at this level (minimum adequate models), at which point estimated values for weekly occurrence probability p_i were calculated from the model parameter estimates using the formula:

$$p_i = e^{\theta}/1 + e^{\theta} \qquad \text{eqn 2}$$

As the same sites will provide data many times over the life of the project, the weekly observations are not independent. Therefore, rather than the default maximum likelihood estimation method in GENMOD we selected the generalized estimating equation (GEE) option for longitudinal data, which produces standard errors and test statistics that are adjusted for dependence (Zeger & Liang 1986) using GEE option TYPE = AR to impose a lag-1 autoregressive structure on the correlation matrix (Allison 1991).

Modelling was performed twice for each species, the first time using datasets containing all sites in order to model overall reporting rates (all sites models). For less frequently occurring species a proportion of the sites might be inherently unsuitable [such as urban gardens for treecreeper *Certhia familiaris* L. and reed bunting *Emberiza schoeniclus* (L.)], and hence interesting trends in the use of suitable sites might be obscured by the bulk of permanently negative observations. Accordingly, we repeated the modelling process using subsets of sites from which any gardens in which a species had never been recorded were removed (species positive models).

CORRELATIONS WITH NATIONAL POPULATION DATA

To examine the extent to which GBW reporting rates correlated with trends in national populations, we compared the annual means over weeks 14–26 of the GBW reporting rates predicted by the model with the population indices (also model-derived) for Great Britain from the BTO/Joint Nature Conservation Committee (JNCC)/RSPB Breeding Bird Survey (BBS), which takes place between April and June (Spearman rank correlation coefficients, PROC CORR in SAS, $n = 7$). We also calculated mean annual 'winter' predicted reporting rates for weeks 1–13 and correlated these with both the following BBS, i.e. that in the same calendar year (BBS/GBW winter$_{-1}$, $n = 7$), and the BBS from the preceding calendar year (BBS/GBW winter$_{+1}$, $n = 6$). The BBS field methodology and analytical procedures are described in detail elsewhere (Raven, Noble & Baillie 2003; Noble *et al.* 2005); indices for Great Britain (not reported elsewhere) were calculated using standard BBS modelling procedures that correct for regional differences in sampling effort. No BBS index was available for the year 2001.

Results

REPORTING RATES

Estimated values for weekly occurrence probability (predicted reporting rates), calculated from the model

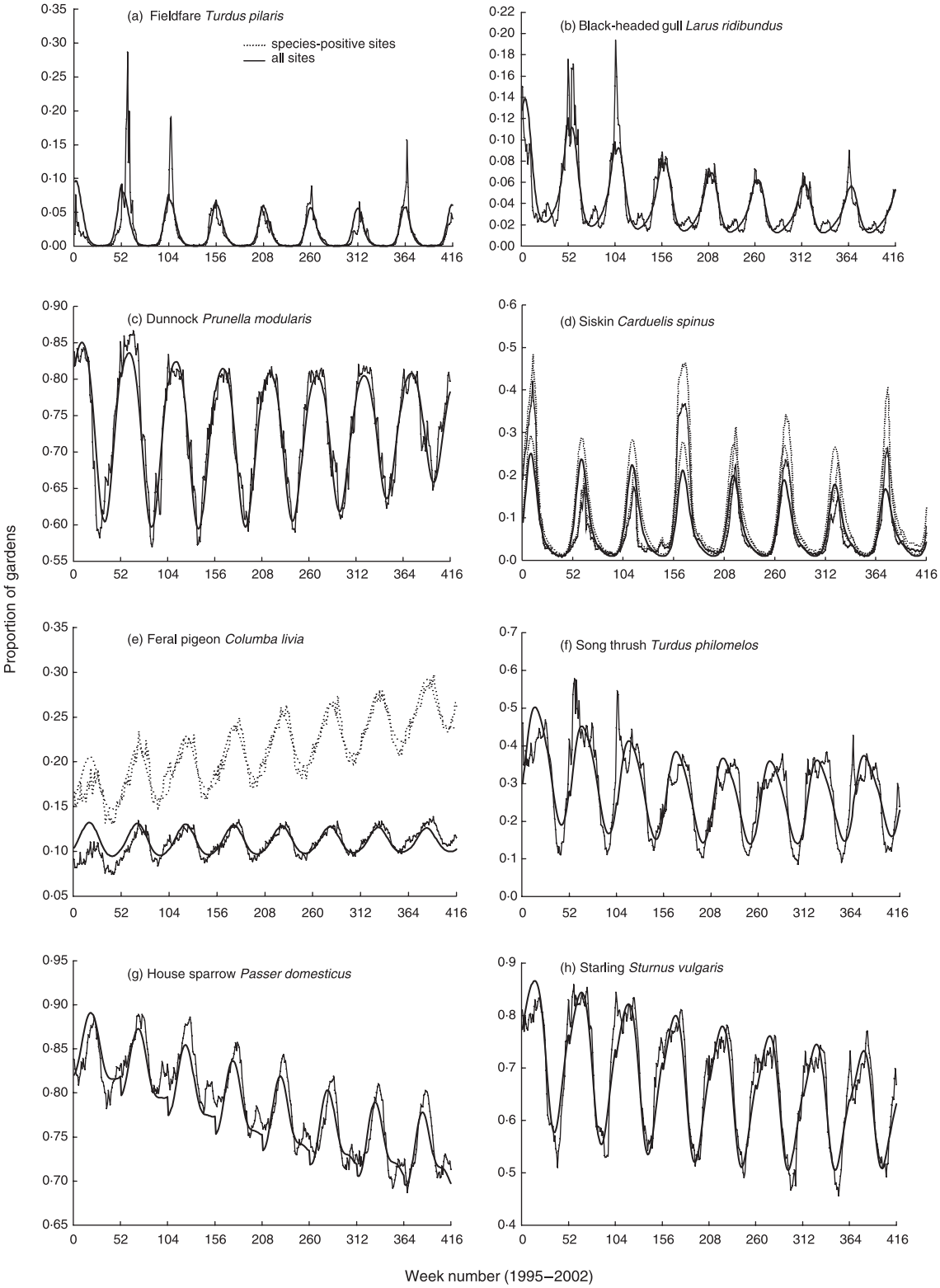

Fig. 3. Weekly reporting rates calculated from original data ('noisy' plots) and probabilities of occurrence predicted by 'minimum adequate' GEE models ('smooth' plots). Week numbers 1–416 run from January 1995 to December 2002. Models using all sites data (black lines) are plotted for all species. Raw data and model results using species positive data (dotted lines) are plotted additionally for species in which there is either a gain or loss of formal significance in the year term or an improvement in data dispersion, or an interesting difference in the form of the two plots.

parameter estimates, together with actual weekly reporting rates calculated from the raw data, were plotted for all 40 species. Fourteen examples are shown here (Fig. 3); figures for all 40 species are available as Figs S1·1–S1·40 in the online supplement and at www.bto.org/gbw/science. The two plots could be distinguished easily: the raw data plots were visibly 'noisy'; the modelled values were smoothed sinusoids. Cyclic patterns of

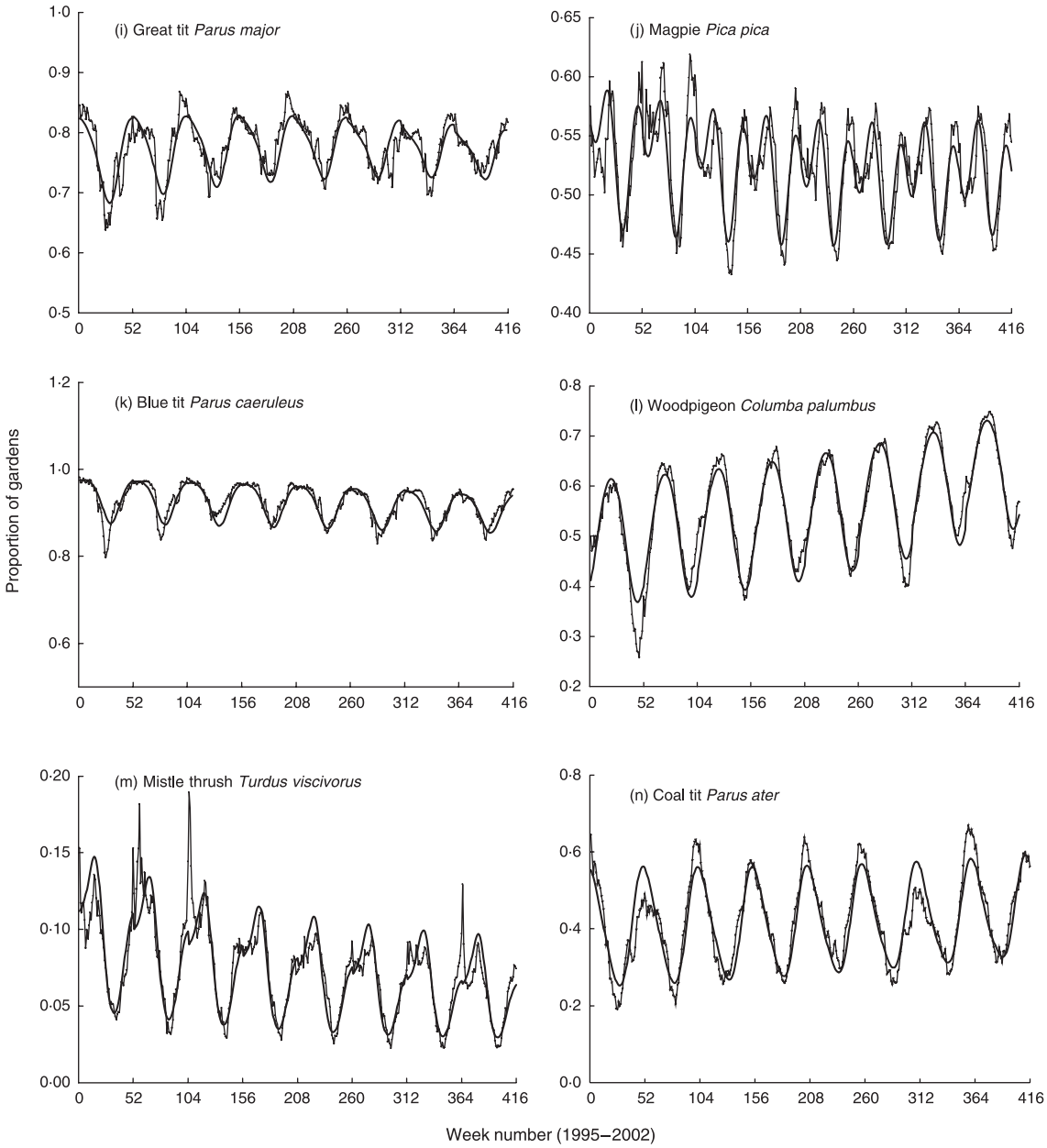

Fig. 3. *Continued*

seasonal habitat use were visible for almost all species, the primary cause of which was likely to be variation in exploitation of garden food supplies. Detailed examination of these cycles was beyond the scope of the present study, but other causal factors (varying in relative importance between species) probably included the effects of moult on bird visibility, preferential use of some types of artificial food by juveniles and the extent to which gardens were used as breeding, rather than merely feeding, habitat. Non-breeding winter visitors were clearly differentiated from residents, migrants [fieldfare *Turdus pilaris* L. (Fig. 3a), redwing *Turdus iliacus* L. and brambling *Fringilla montifringilla* L.] having reporting rates of zero in the summer months and species using gardens almost exclusively for winter food (black-headed gull *Larus ridibundus* L.; Fig. 3b) having very low summer reporting rates.

Several species showed progressive temporal reduction in the amplitude of their periodic seasonal cycles, notably dunnock *Prunella modularis* L. (Fig. 3c), robin *Erithacus rubecula* L., blue tit *Parus caeruleus* L., great tit *Parus major* L., greenfinch *Carduelis chloris* (L.) and perhaps starling *Sturnus vulgaris* L. and chaffinch *Fringilla coelebs* L. General population increases leading to increased garden residency were a possible explanation, as was a general increase in year-round provision of artificial food (Cannon 2000). Reporting rates for winter visitors, including siskin *Carduelis spinus* (L.) (Fig. 3d), which increasingly breeds in Great Britain, were irregular from year to year, as would be expected, but notable irregularity was also present in the data for pied wagtail *Motacilla alba* L., wren *Troglodytes troglodytes* (L.) and goldcrest *Regulus regulus* (L.), insectivorous species whose garden use is strongly

weather-dependent, and for jay *Garrulus glandarius* (L.), nuthatch *Sitta europaea* L. and bullfinch *Pyrrhula pyrrhula* (L.), species for which most gardens are probably 'emergency' habitat, there were a low reporting rates.

MATHEMATICAL MODELLING AND TRENDS

Table 1 lists all model terms and their GEE type 3 *P*-values for all 40 species and the all sites data. To enable comparison between species in the relative significance of their model terms, the values for just the initial model run are given, i.e. all terms were included in these models whether significant or not.

MODEL FIT AND DISPERSION

Although SAS procedure GENMOD does not provide GEE fit statistics (SAS Institute Inc. 2003), we have presented the deviance statistics from the initial maximum likelihood models divided by degrees of freedom in Table 1 as a 'rule of thumb' indicator of relative data dispersion (Der & Everitt 2002; SAS Institute Inc. 2003); from practical experience we have found that values between 0·5 and 2·0 suggest dispersion is reasonably controlled and provide a pragmatic assessment of model appropriateness when combined with visual inspection of predicted probability plots and residuals if necessary. For the all sites data, only 14 of the 40

Table 1. Model terms and their *P*-values for all sites. Values are from model first runs, i.e. all terms included in the model whether significant or not. Terms in bold type are formally significant at $P = 0{\cdot}05$. Significance, magnitude and sign of year and year^2 parameter estimates indicate trends

	Model terms									
Species	Deviance/ d.f.	Year	Parameter estimate	Year^2	Parameter estimate	cosweek	sinweek	cosweek × sinweek	year × sinweek	year × cosweek
(a) Trend unambiguously identified. Presented in ascending order of year parameter estimate, i.e. fastest declining first										
Turdus philomelos	1·1252	**< 0·0001**	**−0·243**	**< 0·0001**	**0·0209**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**< 0·0001**
Passer domesticus	1·0717	**< 0·0001**	**−0.1818**	**< 0·0001**	**0·0088**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**0·0013**
Sturnus vulgaris	1·2500	**< 0·0001**	**−0·1663**	**< 0·0001**	**0·0093**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**0·0002**
Motacilla alba	0·6422	**< 0·0001**	**−0·1424**	**< 0·0001**	**0·0118**	**< 0·0001**	**< 0·0001**	**0·0012**	**< 0·0001**	**< 0·0001**
Prunella modularis	1·1354	**< 0·0001**	**−0·102**	**< 0·0001**	**0·0108**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**< 0·0001**
Corvus frugilegus	0·5693	**< 0·0001**	**−0·0988**	**0·0002**	**0·0076**	**0·0003**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**< 0·0001**
Parus caeruleus	0·5488	**0·0002**	**−0·0806**	**0·9315**	**0·0002**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**< 0·0001**
Corvus monedula	1·0381	**< 0·0001**	**−0·0781**	**< 0·0001**	**0·0081**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**< 0·0001**	0·2026
Corvus corone/cornix	1·1166	**< 0·0001**	**−0·0758**	**< 0·0001**	**0·007**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**< 0·0001**
Pica pica	1·3805	**0·0006**	**−0·0462**	**0·0034**	**0·0035**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**< 0·0001**
Troglodytes troglodytes	1·3161	**0·0025**	**−0·0355**	**< 0·0001**	**0·0054**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**0·0013**
Carduelis chloris	1·1588	**0·0286**	**−0·032**	**0·1405**	**0·0019**	**0·3624**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**< 0·0001**
Fringilla coelebs	1·1442	**0·0259**	**0·0329**	**0·0008**	**−0·0044**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**< 0·0001**
Accipiter nisus	0·6420	**0·0049**	**0·0449**	**< 0·0001**	**−0·0064**	**< 0·0001**	**< 0·0001**	**< 0·0001**	0·6892	**< 0·0001**
Streptopelia decaocto	1·1246	**0·0052**	**0·0454**	**0·0092**	**−0·0037**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**< 0·0001**	0·6469
Dendrocopos major	0·9914	**0·0003**	**0·0555**	**0·0411**	**−0·0028**	**< 0·0040**	**< 0·0001**	**0·0338**	**< 0·0001**	0·0709
Parus major	1·0502	**< 0·0001**	**0·0666**	**< 0·0001**	**−0·0071**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**< 0·0001**
Aegithalos caudatus	0·8375	**< 0·0001**	**0·0962**	**0·0043**	**−0·0037**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**< 0·0001**
(b) No trend identified or trend debatable. Presented in systematic order										
Larus ridibundus	0·2946	**< 0·0001**	**−0·2593**	**< 0·0001**	**0·0171**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**< 0·0001**
Columba palumbus	1·3253	0·1492	0·0192	**< 0·0001**	**0·0068**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**0·0106**	0·2655
Columba livia	0·7003	0·2632	0·0241	0·8450	0·0004	**< 0·0001**	**< 0·0001**	**< 0·0001**	**< 0·0001**	0·0886
Strix aluco	0·2691	**0·0217**	**0·0871**	0·1818	−0·0045	**< 0·0001**	**< 0·0001**	**< 0·0001**	0·4523	0·5892
Erithacus rubecula	0·8203	0·1119	−0·0231	**< 0·0001**	**0·0064**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**< 0·0001**
Turdus merula	0·5008	0·1392	0·0278	0·0510	−0·0032	**0·0008**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**< 0·0001**
Turdus pilaris	0·1518	**< 0·0001**	**−0·2785**	**< 0·0001**	**0·0202**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**0·0064**
Turdus iliacus	0.1724	**< 0·0001**	**−0·3339**	**< 0·0001**	**0·0199**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**< 0·0001**	0·1722
Turdus viscivorus	0·4900	**< 0·0001**	**−0·1317**	**0·0003**	**0·0066**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**< 0·0001**	0·4224
Sylvia atricapilla	0·4232	**< 0·0001**	**−0·1584**	**< 0·0001**	**0·0128**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**0·0068**	**< 0·0001**
Regulus regulus	0·3777	**0·0291**	**0·0517**	**0·0003**	**−0·0075**	**< 0·0001**	**0·0005**	**< 0·0001**	0·1876	0·1126
Parus ater	1·3281	0·6259	0·0062	**0·0282**	**0·0025**	**< 0·0001**	**0·0008**	**< 0·0001**	**< 0·0001**	**< 0·0001**
Sitta europaea	0·7314	0·2881	0·0213	0·6403	0·0008	**< 0·0001**	**< 0·0001**	**< 0·0001**	0·2706	**0·0134**
Certhia familiaris	0.1776	**< 0·0001**	**−0·1547**	**0·0002**	**0·0117**	**< 0·0001**	**< 0·0001**	**< 0·0001**	0·6924	0·4326
Garrulus glandarius	0·7200	0·6698	0·0077	0·3480	−0·0015	**< 0·0001**	**0·0139**	**< 0·0001**	0·1887	**0·0257**
Passer montanus	0·3687	**0·0223**	0·0785	0·2889	−0·0033	0·2036	**< 0·0001**	**< 0·0001**	**< 0·0001**	0·5976
Fringilla montifringilla	0·1257	**< 0·0001**	**0·1695**	**< 0·0001**	**−0·0247**	**< 0·0001**	**< 0·0001**	**< 0·0001**	0·0582	0·1953
Carduelis carduelis	0·9890	0·1811	0·0208	**< 0·0001**	**0·014**	**< 0·0001**	**< 0·0001**	**< 0·0001**	0·0812	**< 0·0001**
Carduelis spinus	0·4412	**0·0115**	**−0·0502**	0·0747	0·0033	**< 0·0001**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**< 0·0001**
Pyrrhula pyrrhula	0·4342	**< 0·0001**	**−0·1258**	**< 0·0001**	**0·0136**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**< 0·0001**	**0·0048**
Emberiza citrinella	0·2124	**0·0063**	**−0·1034**	0·3534	0·0032	**0·0009**	**< 0·0001**	0·5504	0·8295	**0·0035**
Emberiza schoeniclus	0·0803	**0·0005**	**−0.1910**	0·0644	0·0093	**< 0·0001**	**< 0·0001**	**0·0200**	0·3642	0·7923

species had deviance/d.f. figures outside this range; five of these (black-headed gull, fieldfare, redwing, brambling and siskin) were winter visitors with particularly sharp and sudden reporting rate peaks that were clearly less amenable to sine/cosine modelling. Tawny owl *Strix aluco* L., goldcrest, treecreeper, tree sparrow *Passer montanus* (L.), bullfinch, yellowhammer *Emberiza citrinella* L. and reed bunting were low reporting rate species (generally < 0·1), while mistle thrush *Turdus viscivorus* L. and blackcap *Sylvia atricapilla* (L.) had fairly low general reporting rates with some irregular peaks. The data for the latter two species were only marginally underdispersed (deviance/d.f. = 0·49 and 0·43, respectively).

LONG-TERM TRENDS FROM THE ALL SITES DATA SET

The magnitudes and directions of the GEE parameter estimates for the terms year and year^2 indicated whether the models had identified significant overall trends. From Table 1, 32 of the 40 species modelled using data from all sites had a statistically significant year term in their models (type 3 GEE $P_r > \chi^2 = 0{\cdot}05$). Of the eight remaining, four had significant year^2 terms and the year^2 term for blackbird *Turdus merula* L. was only marginally non-significant ($P = 0{\cdot}051$). For only three species, feral pigeon *Columba livia* Gmelin (Fig. 3e), nuthatch *Sitta europaea* L. and jay, had this modelling method clearly failed to identify significant year-on-year change across all sites over the sampling period. These species had uniformly low reporting rates (*c.* 0·1).

For 18 species (Table 1a) the trends were unambiguous, that is data dispersion was clearly within an acceptable range and type 3 GEE scores for year were significant. The three species most clearly identified as negative were song thrush (Fig. 3f), house sparrow *Passer domesticus* (L.) (Fig. 3g) and starling (Fig. 3h). These trends were apparent visually from the reporting rate plots, that for song thrush showing a recent upturn that was reflected in the high positive parameter estimate for year^2. Positive trends were apparent visually in the three species with the most positive year terms, great spotted woodpecker *Dendrocopos major* (L.), great tit and long-tailed tit *Aegithalos caudatus* (L.), that for great tit (Fig. 3i) possibly more the result of progressive flattening of intrayear reporting rate variation. The parameter estimate for year^2 for great tit was considerably more negative compared with the other two, and indeed some decrease in the rate of change was suggested by the plot. Pied wagtail and dunnock had relatively high positive year^2 parameter estimates as well as negative year terms; their reporting rate trends were mitigated by an apparent recovery towards the end of the survey period.

Negative year trends for four corvids, rook *Corvus frugilegus* L., jackdaw *Corvus monedula* L., crow and magpie *Pica pica* (L.) (Fig. 3j), were somewhat surprising given anecdotally reported increases in garden use; their plots supported the suggestion of a slightly downturning overall trend, although the year^2 parameter estimates were all positive, albeit relatively small. A slight downward trend for blue tit (Fig. 3k) was apparent from the plot, but the dispersion statistic was relatively poor, perhaps the result of progressive flattening of the periodicity in this species. The reporting rate trajectory for wren reflected a known population decline and recovery, as discussed in Cannon (2000); an overall trend for this species over this time scale is probably meaningless. The suggested negative trend for greenfinch might be accounted for by a similar flattening of the reporting rate periodicity as in blue tit (the year^2 term was exceptionally non-significant in both species, perhaps suggesting some similarity in data form). The *P*-value for year was not impressive by the standards of this model; the same could be said about chaffinch, for which a positive overall trend was suggested.

A number of species had trends that were clearly apparent to the eye from their reporting rate plots but are not reflected in the *P*-values for both year and year^2 terms. Perhaps the most notable were woodpigeon *Columba palumbus* L. (Fig. 3l, positive), coal tit *Parus ater* L. (positive), goldfinch (positive) and mistle thrush (Fig. 3m, negative). Year and year^2 parameter estimates for these species are shown in Table 1b. Mistle thrush was excluded from Table 1a because of a relatively poor dispersion statistic. However, the plot suggested this may have resulted from a number of irregular reporting rate spikes. If these are disregarded, the significant and relatively large negative parameter estimate for the year term probably reflects an authentic downward trend. The accelerating reporting rate increase for goldfinch was reflected in the highly significant and relatively large positive parameter estimate for year^2, but two anomalous reporting rate peaks, one high (spring 1996) and one low (spring 2002), might explain the inability of the model to resolve a significant year term. The summer reporting rate minima for woodpigeon showed interyear variation (Fig. 3l). This could have compromised the ability of the model to reflect the clearly apparent upward trend, although again a highly significant year^2 term partly redeemeds this. Poor *P*-values for both year and year^2 (the latter formally significant but unimpressive in the context of this model) in coal tit were disappointing, given an apparent upward trend; however, both the winter maxima and summer minima for this species were notably variable from year to year.

COMPARISON WITH LONG-TERM TRENDS IN THE SPECIES POSITIVE DATA SET

As can be seen from Table 2a, all four of the species that had clearly visible trends in their raw data but no unambiguous trends in the all sites models acquired either significant year terms (woodpigeon, coal tit, goldfinch) or an acceptable dispersion statistic (mistle thrush) when modelled using only species positive data. Significant upward trends were also acquired by the only three species for which the model previously failed to

Table 2. Effect of removing any sites where a species has never occurred from the modelled data set on the data dispersion statistic and on the GEE model *P*-values and parameter estimates (est) for terms year and year^2. D, deviance

	All sites data set					Species positive data set				
Species	D/d.f.	Year	*P* est	Year^2	*P* est	D/d.f.	Year	*P* est	Year^2	*P* est
(a) Species gaining significance										
Columba palumbus	1·29	0·1492	0·0192	**< 0·0001**	0·0069	1·30	**0·006**	0·0371	**< 0·0001**	0·0067
Columba livia	0·70	0·2632	0·0241	0·8450	0·0004	1·06	**0·0003**	0·0828	0·7035	−0·0008
Parus ater	1·33	0·6259	0·0062	**0·0282**	0·0025	1·33	**0·0068**	0·0351	0·1617	0·0016
Sitta europaea	0·73	0·2881	0·0213	0·6403	0·0008	1·18	**0·0001**	0·0831	0·4602	−0·0014
Garrulus glandarius	0·72	0·6698	0·0077	0·3480	−0·0015	0·95	**0·0025**	0·0548	0·0766	−0·0029
Carduelis carduelis	0·99	0·1811	0·0208	**< 0·0001**	0·014	1·04	**0·0322**	0·0332	**< 0·0001**	0·0146
(b) Species losing significance										
Corvus monedula	1·24	**< 0·0001**	−0·0781	**< 0·0001**	0·0081	1·24	0·0771	−0·0301	**< 0·0001**	0·0075
Carduelis chloris	1·14	**0·0286**	−0·0320	0·1405	0·0019	1·14	0·1029	−0·0240	0·0977	0·0022
Carduelis spinus	0·44	**0·0115**	−0·0502	0·0747	0·0033	**0·56**	0·7384	0·0065	0·2215	0·0022
(c) Species with notably improved data dispersion										
Emberiza citrinella	0·21	**0·0063**	−0·1034	0·3534	0·0032	**0·58**	0·4236	−0·0301	0·5052	0·0024
Emberiza schoeniclus	0·08	**0·0005**	−0.1910	0·0644	0·0093	0·35	0·2006	−0·0684	0·1641	0·0071
Strix aluco	0·27	**0·0217**	0·0871	0·1818	−0·0045	**0·65**	**< 0·0001**	0·1553	0·0674	−0·0062
Turdus viscivorus	0·49	**< 0·0001**	−0·1317	**0·0003**	0·0066	**0·63**	**< 0·0001**	−0·0877	**0·0002**	0·0069
Passer montanus	0·37	**0·0223**	0·0785	0·2889	−0·0033	**0·91**	**0·0046**	0·1065	0·4686	−0·0025
Pyrrhula pyrrhula	0·43	**< 0·0001**	−0·1258	**< 0·0001**	0·0136	**0·58**	**< 0·0001**	−0·0940	**< 0·0001**	0·0146

identify any significant trend [feral pigeon (Fig. 3f), nuthatch and jay]. Conversely, for jackdaw, greenfinch and siskin the year terms lost formal significance when the models were applied to the species positive data set. For siskin, the replacement of a significant negative year term in the all sites data with a small (although highly non-significant) positive year term in the species positive data might be predicted from comparing the plots (Fig. 3d). In addition to mistle thrush, five other low reporting rate species acquired a more respectable dispersion statistic (Table 2c) although that for reed bunting remained unacceptably low. Formal significance of the year terms for tawny owl, tree sparrow and bullfinch did not change, although the *P*-values greatly improved.

COMPARISON WITH NATIONAL POPULATION TRENDS

In seven of the 37 species for which BBS indices were available, their values correlated significantly with the GBW mean predicted reporting rates over the same period (Table 3 and Figs S2·1–S2·8 in the online supplement). Woodpigeon and great spotted woodpecker had very strong positive correlations and rising trends, black-headed gull and house sparrow had strong positive correlations and falling trends. In greenfinch, blackbird and blackcap, rising BBS trends were negatively correlated with falling trends in GBW, although relatively poor data dispersion and irregularity of winter peaks cast some doubt on the modelled GBW means for blackcap. BBS indices for all these species showed similarly strong correlations with winter GBW predicted means, although the correlation for blackbird reversed, becoming negative. Crow and jackdaw also showed significant negative correlation between winter GBW means and rising BBS trends but their breeding season correlations were non-significant, that for jackdaw strikingly so. For starling, all three correlations were positive and marginally non-significant, both BBS and GBW figures clearly decreasing. Correlations for yellowhammer were also marginally non-significant and consistent. Robin had a significant positive winter$_{-1}$ correlation (rising trend), while tawny owl had a significant positive winter$_{+1}$ correlation, although the clear opposition of the overall GBW (rising) and BBS (declining) trends suggested the latter could be explained by the fall in BBS index having occurred only in the most recent 2 years. The year with the biggest fall (2002) was ignored in the winter$_{+1}$ calculation; prior to 2000 there was a rising trend in BBS as in GBW. If notional BBS and GBW values for the missing BBS year (2001) were interpolated by simple averaging, the correlation lost significance (r_s = 0·393, P = 0·383).

When sites at which species never occurred were removed (species positive data sets), results for the seven species with the strongest GBW/BBS correlations were largely unchanged except that for blackcap the winter$_{-1}$ correlation became non-significant, probably because irregularity in winter reporting rates was damped by the bulk of never-visited sites in the all sites data set. Some species did gain or lose significant correlations (Table 4). Siskin acquired very strong significant correlations in the species positive data, the rising BBS trend correlating positively with GBW breeding season predicted means in gardens favoured by the species, despite an overall negative GBW trend in the all sites data. It can be seen from Fig. 3d that the breeding season reporting rate in the species positive gardens was creeping up. The correlations were strongly negative

Table 3. Correlations (Spearman's rank) between BBS indices (Great Britain only) and GBW mean reporting rates predicted by model for weeks 14–26 (breeding), weeks 1–13 same calendar year (winter$_{-1}$) and weeks 1–13 following calendar year (winter$_{+1}$), data from all sites. Percentage difference between last year and first year values of GBW means and GB BBS indices for the period are given, and overall UK BBS trends are provided for reference. Results in bold type are significant at $P = 0{\cdot}05$

Species	BBS/GBW breeding		BBS/GBW winter$_{-1}$		BBS/GBW winter$_{+1}$		% change in GBW means weeks 14–26	% change in GBW means weeks 1–13	% change in GB BBS index 1995–2002	UK BBS status 1994–2002
	r_s	P	r_s	P	r_s	P				
(a) Positive correlations										
Columba palumbus	**1**	**< 0·0001**	**1·00000**	**< 0·0001**	**1·00000**	**< 0·0001**	**20·39**	**25·28**	22·09	**+5·0**
Dendrocopus major	**0·96429**	**0·0005**	**0·96429**	**0·0005**	**0·94286**	**0·0048**	**12·15**	**11·56**	53·57	**+71·5**
Larus ridibundus	**0·82886**	**0·0212**	**0·82886**	**0·0212**	**0·84067**	**0·0361**	**–53·24**	**–60·14**	–46·43	**–33·2**
Passer domesticus	**0·81537**	**0·0254**	**0·81537**	**0·0254**	**0·89865**	**0·0149**	**–13·00**	**–15·58**	–10·48	**–7·3**
Erithacus rubecula	0·57143	0·1802	**0·92857**	**0·0025**	0·54286	0·2657	**7·30**	**0·78**	4·50	**+16·1**
Strix aluco	0·17857	0·7017	0·17857	0·7017	**0·88571**	**0·0188**	**32·69**	**32·43**	–11·11	**–35·5**
Sturnus vulgaris	0·75	0·0522	0·75000	0·0522	0·77143	0·0724	**–15·22**	**–17·51**	–20·75	**–13**
Emberiza citrinella	0·72075	0·0676	0·72075	0·0676	0·77143	0·0724	**–32·47**	**–42·94**	–8·51	**–13·4**
Accipiter nisus	0·70921	0·0743	–0·43644	0·3276	–0·63775	0·1731	**1·79**	**–15·37**	–2·30	–13·2
(b) Negative correlations										
Carduelis chloris	**–0·91896**	**0·0034**	**–0·91896**	**0·0034**	**–0·92763**	**0·0077**	**–3·14**	**–7·27**	21·70	**+30·8**
Turdus merula	**–0·85714**	**0·0137**	**0·85714**	**0·0137**	0·77143	0·0724	**–0·81**	**0·14**	16·16	**+16·1**
Sylvia atricapilla	**–0·82143**	**0·0234**	**–0·89286**	**0·0068**	–0·77143	0·0724	**–35·14**	**–20·14**	32·41	**+46·2**
Corvus corone	–0·68471	0·0897	**–0·95499**	**0·0008**	**–0·81168**	**0·0499**	**–6·06**	**–11·39**	16·16	**+15·1**
Corvus monedula	–0·07143	0·8790	**–0·92857**	**0·0025**	**–0·82857**	**0·0416**	**4·24**	**–8·59**	8·41	**+12·0**
Regulus regulus	–0·45047	0·3104	–0·45047	0·3104	**–0·92763**	**0·0077**	**–9·29**	**–9·13**	15·17	**+64·9**
(c) Non-significant and poor correlation										
Passer montanus	0·64286	0·1194	0·64286	0·1194	0·42857	0·3965	**22·84**	**21·59**	44·33	**+55·1**
Parus major	0·59462	0·1591	–0·45047	0·3104	–0·77143	0·0724	**2·23**	**–2·37**	13·59	**+18·7**
Carduelis carduelis	0·54554	0·2053	0·54554	0·2053	0·26482	0·6121	**118·76**	**149·38**	21·65	**+18·2**
Garrulus glandarius	0·52254	0·2289	–0·52254	0·2289	–0·23191	0·6584	**2·84**	**–2·49**	46·15	**+16·3**
Certhia familiaris	0·5	0·2532	0·50000	0·2532	–0·08571	0·8717	**–28·39**	**–28·10**	–5·74	**+18·7**
Pyrrhula pyrrhula	0·44475	0·3174	0·55594	0·1950	0·52179	0·2883	**–2·38**	**–11·20**	–8·54	**–26·2**
Streptopelia decaocto	0·43644	0·3276	0·43644	0·3276	0·23540	0·6534	**0·52**	**0·40**	20·39	**+25·8**
Troglodytes troglodytes	0·35714	0·4316	0·00000	1·0000	0·77143	0·0724	**4·24**	**0·13**	–1·74	**+13·7**
Aegithalos caudatus	0·21622	0·6414	0·21622	0·6414	0·49281	0·3206	**42·22**	**56·21**	–17·70	–2·7
Parus caeruleus	0·17857	0·7017	0·17857	0·7017	0·37143	0·4685	**–3·98**	**–3·63**	3·85	**+8·8**
Carduelis spinus	0·14286	0·7599	0·14286	0·7599	–0·02857	0·9572	**–6·87**	**–34·70**	7·89	–18
Prunella modularis	0·09009	0·8477	–0·61264	0·1436	–0·28989	0·5774	**–1·68**	**–4·63**	7·77	**+12·5**
Corvus frugilegus	0·09009	0·8477	0·48651	0·2682	0·17393	0·7417	**–12·95**	**–27·68**	–5·94	–0·3
Motacilla alba	–0·07143	0·8790	–0·10714	0·8192	0·08571	0·8717	**–19·48**	**–29·59**	–3·91	**+23·4**
Emberiza schoeniclus	–0·10714	0·8192	–0·10714	0·8192	0·08571	0·8717	**–53·64**	**–53·35**	–3·92	+3·1
Fringilla coelebs	–0·18019	0·699	–0·41443	0·3553	–0·37143	0·4685	**–0·99**	**–4·39**	7·22	**+5·3**
Parus ater	–0·21822	0·6383	0·41825	0·3504	–0·52179	0·2883	**12·84**	**–0·37**	18·45	**+28·4**
Pica pica	–0·30632	0·5040	0·01802	0·9694	–0·57977	0·2278	**–3·70**	**–8·10**	–3·85	**+2·4**
Sitta europaea	–0·34236	0·4523	0·34236	0·4523	–0·05798	0·9131	**–5·06**	**4·72**	34·58	**+43·6**
Turdus viscivorus	–0·37062	0·4131	–0·37062	0·4131	–0·63775	0·1731	**–32·94**	**–39·63**	0·00	+0·8
Columba livia	–0·39641	0·3786	–0·39641	0·3786	–0·77143	0·0724	**–3·61**	**–4·18**	0·00	–6·3
Turdus philomelos	–0·60714	0·1482	–0·42857	0·3374	0·02857	0·9572	**–27·57**	**–22·42**	13·00	**+13·4**

Table 4. Changes in GBW/BBS correlations when sites at which species never occurred were removed from data (species positive data set). Correlations acquiring or losing significance are shown in bold. Correlations changing sign are shown in italic

	BBS/GBW week 14–26		BBS/GBW winter$_{-1}$		BBS/GBW winter$_{+1}$	
Species	r_s	P	r_s	P	r_s	P
(a) Species gaining significance						
Carduelis spinus	**0·90094**	**0·0056**	**−0·90094**	**0·00056**	**−0·94286**	**0·0048**
Parus major	**0·81084**	**0·0269**	−0·10911	0·8175	−0·48571	0·3287
Streptopelia decaocto	**0·80013**	**0·0307**	0·70921	0·0743	**0·85331**	**0·0307**
Regulus regulus	**0·81084**	**0·0269**	**0·81084**	**0·0269**	0·55078	0·2574
Corvus monedula	**0·85714**	**0·0137**	*0·82143*	*0·0234*	*0·94286*	*0·0048*
(b) Species losing significance						
Sylvia atricapilla	**−0·89286**	**0·0068**	**−0·32143**	*0·4821*	0·25714	0·6228
Larus ridibundus	0·70273	0·0782	0·82886	0·0212	0·84067	0·0361
Corvus corone/cornix	*0·18019*	*0·699*	**−0·45047**	0·3104	*0·08697*	*0·9699*

with both winter GBW means, but the way the model appears to have smoothed this species' irregular and strongly weather-dependent winter peaks into a steady declining trend may be misleading.

For jackdaw, the species positive breeding season GBW/BBS correlation became highly significant and was positive, in contrast to the negative all sites correlation; both winters also changed sign, to become highly significant and positive with respect to the rising BBS trend. Oddly, both winter correlations for crow (significant in the all sites data) became very poor and highly non-significant. The winter$_{+1}$ correlation for goldcrest lost significance when only species positive sites were used, but breeding and winter$_{+1}$ correlations became significant (positive, rising trend), which made sense considering the significant positive year term in the model (albeit with poor dispersion). Finally, in their species positive data sets great tit and collared dove *Streptopelia decaocto* (Frivaldsky, 1838) acquired significant positive breeding season correlations with a rising BBS trend.

To provide a very simple, indicative comparison of overall variation, percentage differences between 1995 and 2002 in the 13-week 'winter' and 'breeding' GBW predicted reporting rate means were calculated and are tabulated in Table 3 together with simple arithmetic percentage differences between 1995 and 2002 in BBS indices for Great Britain. For comparison, we also reprint in Table 3 the percentage changes in BBS index for the UK as a whole between 1994 and 2002 (from Raven, Noble & Baillie 2003), which were model-derived and significance-tested rather than being simply arithmetical.

INTERYEAR CHANGES IN THE TIMING OF THE SEASONAL CYCLE

Interaction terms (year × sinweek, year × cosweek) were included in the model in the hope of detecting significant timing shifts in the seasonal reporting rate cycles. In only four species were both these model terms non-significant (tawny owl, goldcrest, brambling and reed bunting); in a further 13 species one of these two interaction terms was non-significant. It was hard to see any systematic relationships between model term significance and plotted reporting rates. For example, in pied wagtail there was clear variation between the timing of the sharp and irregular reporting rate peaks and the smoothed peaks of the predicted reporting rate plot, however in dunnock (Fig. 3c) and robin the peaks were broad and there was little obvious variation in the phase relationship of the actual and modelled data plots.

Discussion

ANALYTICAL APPROACH

These results suggest that our modelling method can resolve underlying trends in these highly cyclical data, despite the dominance of the sine and cosine terms, which for most species produce GEE parameter estimates an order of magnitude greater than those for the year and interaction terms. However, there are some discrepancies between the timing of modelled occurrence probability peaks and reporting rate peaks in the raw data (e.g. blue tit, house sparrow). Logistic regression models might be expected to detect statistical significance for even quite weak trends in such a large data set, but in some cases visually apparent trends in raw data did not produce correspondingly significant model terms (e.g. coal tit). It seems this modelling method may require further refinement to capture trends adequately in species whose reporting rates are significantly irregular or irruptive. There is also the issue of assessing fit. For example, reed bunting shows a clearly apparent decreasing trend in the raw data and although this is reflected as expected in a significant negative year term in the model, the dispersion statistic is outside normal limits of acceptability. Although the GEE technique allows for temporal autocorrelation, spatial autocorrelation is also a theoretical issue but to date we are unaware of a generally available modelling method

that can allow for this in binary data. Given the 8-year time scale and the wide distribution of sites at a national scale (Fig. 2), we contend that broad-scale long-term trends will greatly outweigh local spatial autocorrelation effects in these data. Adding further complication to the modelling would be practically unfeasible; as it is, data from a total of 16 172 different sites were included in data sets having typically between 2·3 and 2·5 million observations; calculating GEE models with autoregression across 416 weeks for data sets of this size was extremely computationally intensive. The high sensitivity of the models to small interactions leaves us unable to assess meaningfully interyear changes in the timing of garden use by this method alone, although inspection of the GBW data suggests that such changes are becoming apparent in some species and will be worthy of further investigation. However, Fig. 3g shows that for house sparrow the model fails to keep pace with the peak in actual reporting rates, which is becoming steadily later each year, and also has trouble tracking the small post-breeding reporting rate peak despite the excellent dispersion statistic (1·07).

POTENTIAL BIASES IN THE DATA

Given the very large sample size and excellent geographical coverage we contend that GBW is effectively monitoring the specific target habitat, private residential gardens. One point frequently raised in discussion of volunteer garden surveys is that virtually all GBW volunteers provide supplementary food. Given that supplementary food is extremely widespread and frequently superabundant in British gardens, and that preliminary survey data from Sheffield (A. R. Cannon, unpublished data) suggest that around one-third of British households provide food for birds, we make no apology for monitoring 'gardens with supplementary food' as an important habitat in their own right. Furthermore, most British gardens are smaller than the normal territory (and considerably smaller than the winter feeding range) of most species considered here. A mobile bird exploiting supplementary food in one garden will also be exploiting the two unfed gardens either side, hence presence–absence data from fed gardens is likely to be representative of the British garden habitat as a whole, certainly as far as the 40 more common species covered by GBW are concerned. There have been some changes in the timing and types of food provided over the period considered (Toms 2003) and it is possible that these are affecting reporting rates. For example, the reduction in amplitude of the seasonal cycles noted in several species might be partly the result in a shift from winter-only to year-round provisioning. If the relative proportions of various garden types in the survey changed as participation increased over the period considered, this might affect reporting rates for some species, but we have no evidence of this and the recruiting methods and target audience have remained largely unchanged.

TRENDS IN GARDEN USE: INTERESTING TIMES?

The reporting rate data document a period of almost universal change in garden usage rate by most of the more common species that exploit British gardens regularly, and we have successfully captured significant trends for several important species in our mathematical models. Of the 18 species with trends clearly identified by the model, the three with the most negative year term parameter estimates are 'red-listed' as being of high conservation concern (song thrush, house sparrow and starling) and that with the fifth most negative (dunnock) is 'amber listed' as being of medium concern (Gregory *et al*. 2002) Five are designated 'pest' species of economic importance (rook, jackdaw, crow, magpie and collared dove) (DEFRA 2004); for all of these the model has successfully identified significant year trends, all negative except for collared dove, which is positive, as would be expected from the > 25% national population increase over this period. Crow, magpie and jackdaw all have generally increasing populations, so the negative trends in the garden reporting rates are interesting and conflict with anecdotal evidence of a general increase in corvids' use of residential gardens. The models failed to identify significant trends across all sites in four red-listed (tree sparrow, bullfinch, yellowhammer and reed bunting) and five amber-listed (black-headed gull, redwing, fieldfare, mistle thrush and goldcrest) species. None of these species is a typical garden resident, being either a seasonal visitor or primarily associated with other types of habitat. Three 'pests' also fail to show a significant trend across all sites, although all three (woodpigeon, feral pigeon and jay) do have significant and positive year terms in the species positive gardens.

GBW AS A MONITORING TOOL

The presence and detectability of underlying trends in the GBW data enable us to examine whether trends in this habitat differ from those in general populations in direction or timing, perhaps because gardens act as a refuges or because of temporal resource partitioning. Of the species with significant BBS/GBW correlations, positive correlations on rising trends (e.g. woodpigeon, great spotted woodpecker) suggest that increasing populations are colonizing gardens, while those with falling trends (e.g. house sparrow, starling) suggest that garden reporting rates can reflect known serious declines in species of conservation concern. Other species have negative correlations, notably greenfinch and, perhaps surprisingly for a well-established garden-breeding species, blackbird (although the GBW decline for the latter is very slight, suggesting this correlation should be treated with caution). Negative winter correlations in jackdaw, crow and goldcrest support a conclusion that some species are using gardens differently from others. That greenfinch (negative correlation) and house

sparrow (positive correlation) are using gardens in very different ways is supported by the large timing difference between their reporting rate peaks. For some species, garden usage may well reflect breeding populations, whereas for others it may primarily reflect winter feeding behaviour. A previous study found significant correlations between average percentages of winter garden feeders visited and US BBS indices across 13 states for nine North American species (Wells *et al*. 1998) but temporal correlations across a 7-year period were less successful, significant only for Carolina wren *Thryothorus ludovicianus* Latham and house sparrow, whereas we have found significant correlations for 15 species over 8 years. The detection of significant trends and the demonstration of so many correlations with trends in the general population over a relatively short time scale show that GBW has great potential as a monitoring tool for a habitat that is important in its own right and not adequately covered by other monitoring schemes. It provides a wealth of data that it would not be affordable or practicable to collect by any other means, and it is ongoing, so the value and information content of the data can only increase.

Acknowledgements

Individual GBW volunteers, 18 300 in total, contributed data to this analysis for a mean of 130 weeks each, and we are grateful to them all, from the 48 supplying only a single week's data to the 19 who supplied data for all 416 weeks. Jacky Prior, Carol Povey, Tracey Brookes and Derek Toomer worked on GBW during this period, and many other BTO staff and volunteers were involved in the processing of survey forms. Nigel Clark, Chris Whittles and the late Chris Mead were instrumental in establishing and supporting the project, which is funded by volunteers' subscriptions and supported by CJ Wildbird Foods Ltd. Stuart Newson supplied the BBS indices. The BBS is organized by the BTO and funded by the BTO, the Joint Nature Conservation Committee (on behalf of the Countryside Council for Wales, English Nature, Scottish Natural Heritage and the Department of the Environment for Northern Ireland) and the Royal Society for the Protection of Birds. Robert Freckleton and two referees provided useful comments. Steve Freeman and Rob Robinson advised on modelling. A. R. Cannon is funded by a Natural Environment Research Council studentship with the BTO as CASE partner.

Supplementary material

The following supplementary material is available for this article online.

Figures S1·1–S1·40. Weekly reporting rates calculated from original data ('noisy' plots) and probabilities of occurrence predicted by 'minimum adequate' GEE models ('smooth' plots)

Figures S2·1–S2·8. Mean values of predicted GBW reporting rates over weeks 14–26, plotted with corresponding Great Britain BBS indices, years 1995–2002 (2001 missing) for eight species with significant and/or interesting correlations

References

Allison, P.D. (1991) *Logistic Regression Using the SAS System.* SAS Institute Inc., Cary, NC.

Beebee, T.J. (2001) British wildlife and human numbers: the ultimate conservation issue? *British Wildlife*, **13**, 1–8.

Bland, R.L., Tully, J. & Greenwood, J.J.D. (2004) Birds breeding in British gardens: an underestimated population? *Bird Study*, **51**, 96–106.

Cannon, A. (1999) The significance of private gardens for bird conservation. *Bird Conservation International*, **9**, 287–298.

Cannon, A. (2000) *The Garden Birdwatch Handbook*, 2nd edn. British Trust for Ornithology, Thetford, UK.

Chamberlain, D.E., Vickery, J.A., Glue, D.E., Robinson, R.A., Conway, G.J., Woodburn, R.J.W. & Cannon, A.R. (2005) Annual and seasonal trends in the use of garden feeders by birds in winter. *Ibis*, in press.

DEFRA (2004) *General Licences Under the Wildlife and Countryside Act 1981*. Department for Environment, Food and Rural Affairs, London, UK. http://www.defra.gov.uk/wildlife-countryside/vertebrates/General-licence.htm [web site visited 14 April 2004].

Der, G. & Everitt, B.S. (2002) *A Handbook of Statistical Analyses Using SAS*, 2nd edn. Chapman & Hall/CRC, Boca Raton, FL.

Dunnett, N.P. & Beer, A. (2001) The forgotten elements: the contribution of suburban residential gardens to achieving greater environmental sustainability. *Developing a New Architecture for Suburbia.* Robert Gordon University, Aberdeen, UK.

English Nature (2004) *Fact and Figures*. English Nature, London, UK. http://www.english-nature.org.uk/about/annual_report/facts.htm [web sited visited November 2004].

Flury, B.D. & Levri, E.P. (1999) Periodic logistic regression. *Ecology*, **80**, 2254–2260.

Gaston, K.J., Warren, P.H., Thompson, K. & Smith, R.M. (2005) Urban domestic gardens. IV. The extent of the resource and its associated features. *Biodiversity and Conservation*, in press.

Gregory, R.D. & Baillie, S.R. (1998) Large-scale habitat use of some declining British birds. *Journal of Applied Ecology*, **35**, 785–799.

Gregory, R.D., Wilkinson, N.I., Noble, D.G., Robinson, J.A., Brown, A.F., Hughes, J.D.A.P., Gibbons, D.W. & Galbraith, C.A. (2002) The population status of birds in the United Kingdom, Channel Islands and Isle of Man: an analysis of conservation concern 2002–07. *British Birds*, **95**, 410–450.

Jeffcote, M.T. (1993) Wildlife conservation and private gardens in the city of Leicester. *Urban Nature Magazine*, **1**, 151–154.

Marzluff, J.M., Bowman, R. & Donnelly, R. (2001) *Avian Ecology and Conservation in an Urbanizing World.* Kluwer Academic Publishers, Norwell, MA.

Mason, C.F. (2000) Thrushes now largely restricted to the built environment in eastern England. *Diversity and Distributions*, **6**, 189–194.

Noble, D.G., Newson, S.E., Baillie, S.R., Raven, M.J. & Gregory, R.D. (2005) Recent changes in UK bird populations measured by the Breeding Bird Survey. *Bird Study*, in press.

ODPM (2003) *Land Use Change in England.* No. 18A. Office of the Deputy Prime Minister, London, UK http://www.odpm.gov.uk/stellent/groups/odpm_planning/documents/page/odpm_plan_024948.pdf [web site visited 14 April 2004].

Owen, J. (1991) *The Ecology of a Garden*. Cambridge University Press, Cambridge, UK.

Peach, W.J., Denny, M., Cotton, P.A., Hill, I.F., Gruar, D., Barritt, D., Impey, A. & Mallord, J. (2004) Habitat selection by song thrushes in stable and declining farmland populations. *Journal of Applied Ecology*, **41**, 275–293.

Raven, M.J., Noble, D.G. & Baillie, S.R. (2003) *The Breeding Bird Survey 2002*. Report No. 8. British Trust for Ornithology, Joint Nature Conservation Committee and Royal Society for the Protection of Birds, Thetford, UK.

RSPB (2002) *RSPB Home Page*. Royal Society for the Protection of Birds, Sandy, UK. http://www.rspb.org.uk [web site visited 9 December 2002].

Rushton, S.P., Ormerod, S.J. & Kerby, G. (2004) New paradigms for modelling species distributions? *Journal of Applied Ecology*, **41**, 193–200.

SAS Institute Inc. (2003) *Technical FAQ No. 1808: Is There a Goodness-of-fit Statistic or Test?* SAS Institute Inc., Cary, NC. http://support.sas.com/faq/FAQ01808.html [web site visited 19 November 2003].

Toms, M.P. (2003) *The BTO/CJ Garden Birdwatch Book*. British Trust for Ornithology, Thetford, UK.

Veerman, P.A. (2002) *Canberra Birds. A Report on the First 18 Years of the Garden Bird Survey.* Canberra Ornithologists' Group, Canberra, Australia.

Wells, J.V., Rosenberg, K.V., Dunn, E.H., Tessaglia-Hymes, D.L. & Dhondt, A.A. (1998) Feeder counts as indicators of spatial and temporal variation in winter abundance of resident birds. *Journal of Field Ornithology*, **69**, 577–586.

Zeger, S.L. & Liang, K.-L. (1986) Longitudinal data analysis for discrete and continuous outcomes. *Biometrics*, **42**, 121–130.

Received 16 September 2004; final copy received 28 February 2005
Editor: Rob Freckleton