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Predicting the Response of Farmland Birds to Agricultural Change

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EXECUTIVE SUMMARY

1. The deleterious effects of agricultural intensification on the abundance and diversity of farmland birds are now generally accepted. The Department for Environment, Food and Rural Affairs has adopted as one of its eight Public Service Agreement (PSA) targets to reverse the long-term decline in the number of farmland birds by 2020. This is measured annually using the 'farmland bird index', which comprises population trends of 20 species: Kestrel, Grey Partridge, Lapwing, Stock Dove, Woodpigeon, Turtle Dove, Barn Owl, Skylark, Yellow Wagtail, Whitethroat, Jackdaw, Rook, Starling, Tree Sparrow, Greenfinch, Goldfinch, Linnet, Yellowhammer, Reed Bunting, Corn Bunting. To meet this target it will be necessary to determine: (i) the magnitude of changes in demographic rates required to reverse these declines and the habitat management changes required to achieve these; (ii) the extent to which current land management measures (within and outside agri-environment schemes) already meet species' needs and; (iii) identify any gaps in knowledge required to predict responses of these 20 bird species to habitat management change. The work reported here addresses the first issue using population and habitat association modelling, the second issue using a literature based review and draws on the results of both of these to address the third issue.

2. Population models were used to predict the changes required in current demographic rates to reverse the farmland bird index (defined here as achieving an indicator growth rate of greater than one, while assuring that the population growth rates for all the 10 currently declining species are also greater than one [Grey Partridge, Lapwing, Turtle Dove, Barn Owl, Skylark, Starling, Tree Sparrow, Yellowhammer, Reed Bunting, Corn Bunting] and population growth rates of currently increasing species remain the same). There are many combinations of population change of individual species that could, in theory, meet the PSA target. This scenario meets the criteria on achieving the target by non-perverse means but it is used only to provide a guide as to the scale of demographic changes required not as prescription for meeting the PSA target itself. Current values for survival and fecundity were estimated from British Trust for Ornithology national data sets (using means from the years 1990-2000) or from intensive studies in the published literature.

3. Estimates of the changes required in key demographic parameters suggest that, in general, the same population outcome is attained by a much smaller percentage change in survival than in productivity. All things being equal this implies that action to enhance survival may be more successful than action to enhance productivity. These models do not account for density-dependence, which will tend to force populations towards equilibrium (slowing declines as density falls and slowing increases as density rises). They provide estimates of minimum changes in key demographic rates required to halt the decline in the farmland bird index (estimates increase rapidly as the strength of density-dependence increases). For five species, reversing the population trend would require increasing survival for Barn Owl by 8.3%, Starling by 5.9%, Tree Sparrow by 8.6%, Yellowhammer 3.8% and Reed Bunting by 4.0%. For two species, it would require increasing the number of breeding attempts per year; for Turtle Dove by 20.2% and for Skylark by 6.7%. For Lapwing, the number of fledglings per breeding attempt would have to increase by 6.6%. For Corn Bunting survival would have to increase by 8.8% or number of breeding attempts 19.7% to halt the species' decline.

4. Breeding season habitat association models could be developed for seven of the 20 farmland bird index species, with greatest confidence in predictions for Lapwing, Skylark, Whitethroat and Yellowhammer. These models operate under the caveat that they assume unlimited potential for populations to respond to provision of breeding season resources, i.e. that there is no limitation of winter resources. Each model was developed for three baseline landscapes, representing pastoral, mixed and arable farming systems. Manipulations of habitat variables were limited to those considered likely to occur, either in future farming changes or as part of agri-environment schemes e.g. varying crop types, farming systems (organic/conventional), set-aside, hedgerow and field margin lengths. Significant increases in Skylark numbers were predicted if more land was managed under spring (rather than winter) cereal or as set-aside or organic farming (the latter particularly in the pastoral landscape). Lapwings were predicted to increase with increasing areas of set-aside and winter stubble. Boundary nesters like Yellowhammer and Whitethroat were predicted to increase with increasing incidence of hedgerows, ditches and wide uncropped field margins. Some of these increases were dramatic. Hedgerow occupancy by Whitethroat, for example, was predicted to increase by 130% if 2 m margins were added to 30% of all field boundaries.

5. Habitat-association models have to be interpreted and used with caution. They are based on correlation between habitat variables and bird numbers and do not link change in numbers to habitat change via an explicit effect on a demographic rate. A model that links population change to habitat change via demography was only possible for Grey Partridge. For this species, brood production rate and female winter survival rate are density dependent, whereas chick survival rate is not. Density dependence is related, in turn, to the availability of nesting cover, and the relationship differs with and without this change according to whether predator control. When modelled, Grey Partridge chick survival rate increased by 0.04 for every 1% increase in insect-rich arable habitat. Recovery of the population to its 1996 level depends on achieving around 6 km/km^2 of nesting cover and 3% of arable area as insect-rich brood-rearing habitat in the absence of predation control.

6. A literature review of the nesting and foraging requirements of the 20 farmland bird indicator species highlighted the diversity of resources required. Despite this diversity, matching resource requirements to those provided by major agri-environment schemes and other land management measures (e.g. Environmentally Sensitive Areas [ESAs], Countryside Stewardship Scheme [CSS], Tir Gofal, organic farming and integrated farming systems and set-aside) showed very few major gaps. The major exception was for options re-introducing small areas of arable land into grassland landscapes and perhaps also the reverse of grass pockets in arable land.

7. Thus the scope of current agri-environment schemes and farm management practices could potentially meet most of the needs of most of the bird species in the index. The failure to realise this potential is related to two key issues - the scale at which options are deployed and the quality of the habitats provided. Many of the current options/practices could potentially provide resources but the outcome is not certain. In some cases this is because our knowledge about species requirements or the impacts of management is insufficient to make predictions. This is particularly true, for example, for birds within grassland systems. In other cases this is because management prescriptions are not specific enough to guarantee they will deliver

benefits. This is particularly true, for example, in the lower tiers of ESAs. Increasing the extent to which high quality habitats could be achieved through (i) high quality on-farm advice and monitoring (as in the case for Cirl Buntings) although there are resource implications associated with this; (ii) tighter management prescriptions or more active promotion of higher tiers in schemes; (iii) more careful targeting of options towards species and/or regions, matching species and habitats present e.g. through farm audits with options; and (iv) in some cases, increased scientific input (from further research) to the development of options, as for the new arable options in CSS.

8. Habitat quality and quantity are, of course, related and similar results might be achieved either by managing small areas of high quality land or large areas of low quality land. Although for most species we do not know how much of a given resource is required, it is likely to be considerable. For example, the local recovery of Cirl Bunting populations was achieved through the provision of extensive areas of stubble and grass margins. Set-aside, which has accounted for sizeable areas of arable land for some time, has not resulted in marked changes in population trends of birds known to benefit from set-aside such as skylark. In order to affect national bird populations over a wide scale, as measured by the Breeding Bird Survey a combination of high quality targeted habitat management coupled with less costly options over extensive areas of agricultural land will be required. We suggest that this combination of measures will succeed in delivering the PSA target. There may be a dichotomy in the spatial distribution of resource requirements between summer and winter. Fewer, smaller areas of high quality resource may be sufficient in winter when birds are mobile and flock; ubiquitous habitat provision may be more appropriate in summer, when territorial birds exclude each other from resources.

9. The outputs of the demographic and habitat-association models allow us to assess whether, and to what extent, each species would benefit from a number of management options within arable and pastoral landscapes, assuming that the quality of the management was assured. In the summary chapter, we rank options by their delivery for the 14 declining or amber listed species included in the farmland bird index. Top options were arable or pastoral pockets (delivery for at least nine, possibly 12 species), non-cropped habitats such as hedges, trees and aquatic habitats (delivery for at least nine, possibly 11 species) and set-aside (delivery for at least eight, possibly 11 species). Other high-ranking options included winter stubbles, non-cropped field margins, wild bird cover, spring cereals, targeted pesticide restrictions and extensification of grassland management. It is also important to note that some highly targeted options, such as fallow plots for lapwing, though not delivering for a wide range of species, can deliver well for the species for which they are designed. It is important to note that sustained population recovery (rather than just the initiation of population recovery) will only be achieved by an integrated approach in which resources are provided for both the winter and breeding requirements of a range of species.

10. Further ecological and demographic research would strengthen and quantify management recommendations for birds. Three high priority needs, based on the extent to which they assist in meeting the PSA target, are: (i) a feasibility study to assess the extent to which a 'supplement' to the BBS could be used to assess the effectiveness of agri-environment schemes in delivering their 'bird objectives' and

provide data on the extent of habitat/resource required to reverse population declines; (ii) a resurvey of birds and habitats under the Arable Stewardship Pilot Scheme to assess the effectiveness of these options and the extent of habitat/resource required to enhance local bird populations; (iii) a large scale experiment designed to assess the cost-effectiveness of the introduction of arable pockets into grassland landscapes in the form of whole field, in-field or margin plots and; (iv) the development of models that link resource availability (i.e. management change) to demographic parameters and hence to population size for one to three key case species, or of individuals-based models to predict the amount of resource needed for granivorous passerine populations to survive over-winter.

CHAPTER 1 POPULATION MODELLING

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1.1 Introduction

The aim of this part of the project was to produce predictions of the future population trends of the farmland indicator species based on the best possible estimates of the current values of their demographic rates. The predictions would then allow us to investigate the changes in these demographic rates (as might be caused by agricultural change) that would be required to change each species' trend in order to meet the Public Service Agreement target of reversing the decline in the composite farmland indicator. We have considered 19 of the 20 species whose population trends are combined in the farmland indicator; the exception is Grey Partridge, which was considered independently using the data holdings of the Game Conservancy Trust and a more detailed model, as outlined in the Introduction (Section 2). The implications of the Grey Partridge results for our less detailed investigations into the other species, and of combining all 20 predicted trends into predictions for the indicator, are considered in the general discussion.

1.2 Models

We used a population modelling approach to predict future population trajectories for the farmland indicator species. This approach required data on the current values of the demographic rates (survival and productivity) operating in British populations of the 20 species. We could then use the following equation, adapted from those widely used in retrospective population modelling (e.g. Peach *et al.* 1999, Thomson *et al.* 1997, Siriwardena *et al.* 1999, 2001) to predict inter-annual changes in abundance from demographic rate data:

$$N_{t+1} = (N_t \times S^{AD}) + (N_t \times S^{FY} \times FPA \times NA \times S^{PF} \times 0.5) \qquad \text{eqn 1}$$

where N_t and N_{t+1} denote breeding abundance in one year and the next, S^{AD}, S^{FY} and S^{PF} are, respectively, annual adult, first-year (independence through to the second summer) and post-fledging (fledging to independence) survival rates, *FPA* is the number of fledglings produced per breeding attempt and *NA* is the number of breeding attempts made. The additional factor of 0.5 is needed because *FPA* and *NA* are measured per pair, rather than per individual. This model was derived by adapting those used in earlier integrated population modelling research at the BTO (e.g. Thomson *et al.* 1997, Peach *et al.* 1999, Siriwardena *et al.* 2001). Applying the model recursively from an arbitrary estimate of abundance in year t=1 provides predictions of abundance in any year t>1 relative to the level in the first year, assuming that all the demographic quantities remain constant.

1.3 Data Sources

Ideally, we would feed into eqn 1 national data on each of the demographic parameters from a period ending at the present and as short in length as possible (while providing numerically precise estimates that are not unduly influenced by annual fluctuations, such as unusually poor weather). Many of these data are available from the BTO's data sets and we have used these sources as a first option (Table 1), taking ring-recovery, nest record and Common Birds Census data from 1990-2000 as a compromise between numerical precision and relevance to present bird populations. In practice, however, national data on one or more of the demographic rates in eqn 1 were unavailable or insufficiently reliable for many species. Ring-recoveries potentially provide information on all three survival rates, but sample sizes are prohibitively small for some species (e.g. Skylark, Corn Bunting) and too small to produce reliable current estimates for others species (e.g. Turtle Dove). Further, the reliable estimation of post-fledging survival rates requires knowledge of the numbers of birds ringed each year, which is only available for a limited number of species (two in the farmland indicator set with adequate ring-recovery sample sizes). Nest records provide estimates of fledgling production for most of the species of interest, but no relevant data have been computerised for one species (Woodpigeon). For multi-brooded species, no data on numbers of breeding attempts made by individual pairs each year are collected currently under any BTO scheme.

Where data on demographic rates were needed that could not be supplied by the BTO's data holdings, we have sought alternative sources of information. No data sets exist that are comparable in scale to the BTO's, but data from local, intensive work and review articles in the recent literature provided useful alternatives/additions as indicated in Table 1. Given measured changes in abundance over a given period and corresponding data on all but one of the parameters in eqn 1, it is also possible to derive an estimate for the unknown parameter through calculations using appropriate re-arrangements of this model equation. We have used this approach for demographic rates for which estimates could not be derived from BTO data or from the literature.

The parameter in eqn 1 for which estimates are least often available is the post-fledging survival rate. We have been able to avoid the problem of estimating it for some species, however, by using ring-recovery information from young ringed in the nest to estimate survival rates over the first year of life. This combined, fledging-to-second-summer survival rate is referred to as "juvenile survival", as distinct from "first-year survival" from here on. Using juvenile survival produces the following revision of eqn 1:

$$N_{t+1} = (N_t \times S^{AD}) + (N_t \times S^{JV} \times FPA \times NA \times 0.5) \qquad \text{eqn 2}$$

where $S^{JV} = S^{FY}S^{PF}$. Passerines are typically ringed more commonly as fledged, independent juveniles than as nestlings (nest box species such as Tree Sparrow are sometimes an exception), so estimates of first-year survival tend to be more reliable than those of juvenile survival. However, the reverse is true for many non-passerines: we have used juvenile survival rates and eqn 2 for all species for which such a policy provided a more complete picture of its demography given the quality and quantity of the data available.

The full set of parameter estimates used in our initial models is shown in Table 1. These represent our best estimates of each parameter. BTO data from 1990-2000 were analysed specifically for this project as follows:

i. Ring-recovery data were analysed using program MARK (White & Burnham 1999) and models allowing only age-specific variation in survival, i.e. forcing each of adult and first-year or juvenile survival to be constant over the period of interest. See Siriwardena *et al.* (1998) for full details of the method.
ii. Nest Record Scheme data were analysed to provide constant estimates of each of clutch size, hatching success, brood size and the daily nest failure rates in egg and nestling periods, using simple averages or logistic models, as appropriate. Nest record cards were selected for analysis if they were recorded as coming from farmland.

These estimates were then combined to produce a constant estimate of fledgling production per breeding attempt for 1990-2000. See Siriwardena *et al.* (2000a) for full details of the method.

iii. Common Birds Census data from farmland plots were used to obtain an average recent population trend for each species, i.e. the linear slope (on the log scale) that best described the variation in abundance on CBC plots between 1990 and 2000. This slope was derived from log-linear Poisson regression models (ter Braak *et al.* 1994) in which CBC count was dependent on a categorical plot effect and a linear year effect. The parameter estimate for the year effect provided the slope, which was linear on the log scale and which could be converted to an estimate of the multiplicative population growth rate by inverting the logarithmic transform (i.e. taking an antilog).

Note that, even after we investigated BTO data in detail and examined the literature, there were some demographic rates for some species for which we were unable to find estimates. For these few parameters, we were forced to apply expert judgement based on data for related or ecologically similar species.

Table 1 Demographic parameter estimates used for predictive population models. "First-year survival" was defined as survival from independence until the following breeding season, while "juvenile survival" was derived from samples of birds ringed as pulli, thus incorporating survival over the post-fledging period as well as over the birds' first winter. Adult, first-year and juvenile survival estimates were derived from BTO ring-recoveries unless noted. Post-fledging survival rate estimates (needed when first-year survival was used) were independent estimates where a superscript "e" appears and were derived from BTO abundance, survival and breeding performance data, using a re-arrangement of eqn 1, where a superscript "d" appears. All estimates of fledglings per attempt but one (see footnote 1) were derived from BTO nest record data. Average numbers of breeding attempts were taken from Crick *et al.* 1993 unless noted. Population growth rates were derived from CBC data. Where the data necessary to estimate particular parameters were unavailable, we used either expert judgement (denoted by superscript "g") or data for closely related species (see numbered footnotes). 95% confidence intervals were calculated for all the estimates derived from BTO data and were also available from some of the other data sources, but they are omitted here for clarity.

Species	**Adult survival**	**First-year survival**	**Juvenile survival**	**Post-fledging survival**	**Fledglings per attempt**	**Number of attempts**	**Population growth rate, 1990-2000**
Kestrel	0.599	-	0.297	-	3.85	1	1.001
Grey partridge							
Lapwing	0.644	-	0.426	-	1.63	1	0.977
Stock dove	0.469	-	0.406	-	1.32	3.5	1.045
Woodpigeon	0.590	-	0.532	-	1.8[1]	(2.25)	1.014
Turtle dove	0.525[2]	0.185	-	0.7[g3]	1.09	1.6[4]	0.933
Barn owl	0.544	-	0.26	-	3.15	1.25	-
Skylark	0.66[5]	-	0.25[5]	-	1.53	2	0.977
Yellow wagtail	0.532[6]	0.463[6]	-	0.5[g7]	1.43	1.25	1.010
Whitethroat	0.389	0.279	-	0.30[d]	2.87	2	1.051
Jackdaw	0.832	-	0.464	-	2.45	1	1.043
Rook	0.738	-	0.262	-	1.58	1[8]	1.033
Starling	0.571	0.366	-	0.386[e9]	3.63	1.5	0.945
Tree sparrow	0.552	-	0.493	-	3.35	1.61[10]	0.921
Greenfinch	0.418	0.366	-	0.566[e9]	2.10	2	1.033
Goldfinch	0.371[11]	0.337[11]	-	0.7[g12]	1.98	2	1.035
Linnet	0.330	0.286	-	0.85[d]	2.48	2.25	1.008
Yellowhammer	0.536[11]	0.529[11]	-	0.45[d]	2.77	2.5	0.963
Reed bunting	0.492	0.398	-	0.59[d]	1.99	2	0.961
Corn bunting	0.53, 0.5[13]	0.4[g14]	-	0.7[e15]	2.23	1.25	0.924

1. Nest record data for Woodpigeon are not computerised: this value (for a population in Cambridgeshire between 1970 and 1983) comes from Inglis *et al.* (1994) and refers to fledglings *per pair* rather than per breeding attempt, so incorporates the influence of the number of breeding attempts made (the latter is therefore shown in parentheses). This means that a population model combining the available survival and breeding success information would take account of the mortality of very young birds in two components of the model simultaneously and thus probably over-estimate it.
2. From data for 1962-1995, taken from Siriwardena *et al.* (2000b).
3. A realistic value for a bird larger than most passerines.
4. An estimate from a study based in East Anglia, 1998-2000 (Browne & Aebischer 2001).
5. Estimates from a local study in duneland in north-west England, 1995-1998, measured directly for adults and inferred for juveniles (Wolfenden & Peach 2001).
6. Estimates for the period 1962-1995 (G.M. Siriwardena, unpubl.): recovery data were too sparse to allow estimates to be made for 1990-2000.
7. A realistic value for a medium-sized passerine.
8. Cramp & Perrins (1994).
9. Estimates from new analyses of BTO ringing and ring-recovery data (R.A. Robinson, unpubl.).
10. Summers-Smith (1995).
11. Estimates from BTO ring-recoveries for 1962-1995 (Siriwardena *et al.* 1998): estimates for 1990-2000 were based on small ring-recovery sample sizes and were biologically implausible. Note that there is evidence that recent Yellowhammer survival has been considerably lower (Siriwardena *et al.* 1998).

12. The estimate of Goldfinch post-fledging survival derived from breeding performance, first-year survival and abundance data was implausible (0.997), so we used a more likely estimate mid-way between those for the congeneric Greenfinch and Linnet.
13. Møller (1983), Brickle (1999).
14. A realistic estimate in the context of the estimates of adult survival for the species.
15. Brickle (1999).

The data on population growth rates in Table 1 show a notable pattern: despite the broadly declining trend in the farmland indicator, only eight of the eighteen species considered actually had a declining population trend, on average, between 1990 and 2000. (Barn Owl is probably also still in decline, nationally, but no census data are available to confirm this (Toms *et al.* 2000, M.P. Toms, pers. comm.).) Species such as Yellow Wagtail, Linnet and Goldfinch underwent significant declines on farmland CBC plots in the 1970s and 1980s, but their populations have subsequently stabilised or begun to increase (Baillie *et al.* 2001). Conservation targets for these species might therefore best be phrased in terms of re-attaining previous population levels or maintaining current rates of recovery. More generally, it is important to realize that CBC declines are always likely to slow down and, perhaps, to stop as a species becomes restricted to a few key plots where the habitat and environmental conditions remain adequate and local abundance is therefore stable. This means that an unqualified interpretation of a target of achieving a population growth rate of one may not be the most appropriate in terms of conservation. A growth rate considerably higher than one, however, would show increasing abundance on the key plots and/or the (re-)colonisation of further plots.

1.4 Initial Model Results

Figure 1 shows the results of constructing models from entering the "best estimates" of demographic parameters presented in Table 1 into eqn 1 or 2, as appropriate, taking N_1 to be unity for each species and the year 2000 to be $t=1$. Given models derived from accurate estimates of the demographic rates during the period 1990-2000, the predicted model trends would be expected to follow on smoothly from the observed log-linear CBC trends. Figure 1 shows clearly that such smooth transitions are an exception rather than the norm. This pattern implies, therefore, that some of the demographic parameter estimates we used were either inaccurate or based on a sample of birds (probably related to habitat or geographical coverage) that is biased relative to the coverage of the CBC. (It is also conceivable that the CBC is a less accurate representation of the national population of a given species than our population model, but, given that the farmland indicator is based largely on CBC data and this project is therefore implicitly concerned only with the populations that contribute to the CBC, it seems sensible to treat the CBC trend as "the truth".)

It is unsurprising that there are inaccuracies and biases in our best demographic parameter estimates. Clearly, the parameters that were guessed or derived from related species could be wrong and most of the data taken from the literature were derived from studies that were not well matched to BTO data in terms of spatial scale, geographical location or time period. However, there are also known spatial biases in the coverage of the different national BTO data sets. The farmland CBC is representative of cropping patterns in south and east Britain, but not elsewhere (Fuller *et al.* 1985). Although we only selected nest record data that came from farmland, we could not control for any possible biases in the type of farmland (crop types, patterns of field boundary structure, etc.) or match them closely to those of the CBC (which are not known in such detail). In addition, meaningful spatial or habitat controls cannot be applied to survival estimation from ring-recovery data alone, and the geographical/habitat coverage of this data set is unknown, even at the level of farmland versus woodland or wetland. The latter could cause particular problems for those species that are commonly found and ringed in non-farmland habitats, such as Reed Bunting, Skylark and Linnet.

As a result, although the data in Table 1 represent the best estimates we could obtain, we do not believe that their precise values should be taken as "correct". Notwithstanding the possible biases described above, many of the estimates have associated 95% confidence intervals covering several tens of percentage points of their values. To illustrate the potential importance of the (im)precision of the parameter estimates, we plotted sets of trends equivalent to those in Figure 1 but using either all the lower 95% confidence limits or all the upper confidence limits for the various parameters in Eqn 1 (Figures 2 and 3, respectively). The 95% confidence intervals for each parameter are presented in Table 2. Figures 2 and 3 show the extremes of the likely ranges of population trends: the range of trends predicted by realistic estimates for the demographic parameters is clearly very large for many species. As a result, it would make little sense necessarily to persist with using exactly those "best" estimates of each demographic parameter that appear in Table 1 as a basis for our predictive models. However, the "real" current parameter values (i.e. those that apply to populations covered by the CBC) will almost certainly lie between the values used to generate the extreme trends. We therefore produced new predictive population models by adjusting the initial parameter estimates within reasonable ranges to produce trends that were more realistic in the context of the observed CBC trends.

Species	**Adult survival**	**First-year survival**	**Juvenile survival**	**Post-fledging survival**	**Fledglings per attempt**
Kestrel	0.564-0.634	-	0.268-0.328	-	3.68-4.02
Grey partridge					
Lapwing	0.575-0.743	-	0.354-0.501	-	
Stock dove	0.306-0.638	-	0.274-0.554	-	1.22-1.42
Woodpigeon	0.513-0.662	-	0.442-0.62	-	-
Turtle dove	0.381-0.665	0.026-0.663	-	-	0.80-1.38
Barn owl	0.505-0.582	-	0.239-0.282	-	2.80-3.50
Skylark	0.38-0.78	-	-	-	1.31-1.75
Yellow wagtail	-	-	-	-	?1.26-1.60?
Whitethroat	0.298-0.488	0.188-0.392	-	-	2.60-3.14
Jackdaw	0.728-0.902	-	0.338-0.596	-	2.27-2.63
Rook	0.557-0.864	-	0.129-0.46	-	1.21-1.85
Starling	0.537-0.604	0.329-0.406	-	0.329-0.447	3.44-3.82
Tree sparrow	0.35-0.739	-	0.329-0.658	-	3.18-3.52
Greenfinch	0.393-0.444	0.328-0.405	-	0.340-0.768	1.87-2.33
Goldfinch	0.336-0.408	0.294-0.382	-	-	1.64-2.32
Linnet	0.232-0.446	0.18-0.421	-	-	2.31-2.65
Yellowhammer	0.472-0.592	0.383-0.546	-	-	
Reed bunting	0.377-0.607	0.261-0.553	-	-	1.59-2.39
Corn bunting	-	-	-	-	1.90-1.56

Table 2 All available 95% confidence intervals for the demographic parameter estimates used in predictive population modelling. See text and legend to Table 1 for definitions of the parameters. Note that no confidence intervals were available for any estimates of the number of breeding attempts made.

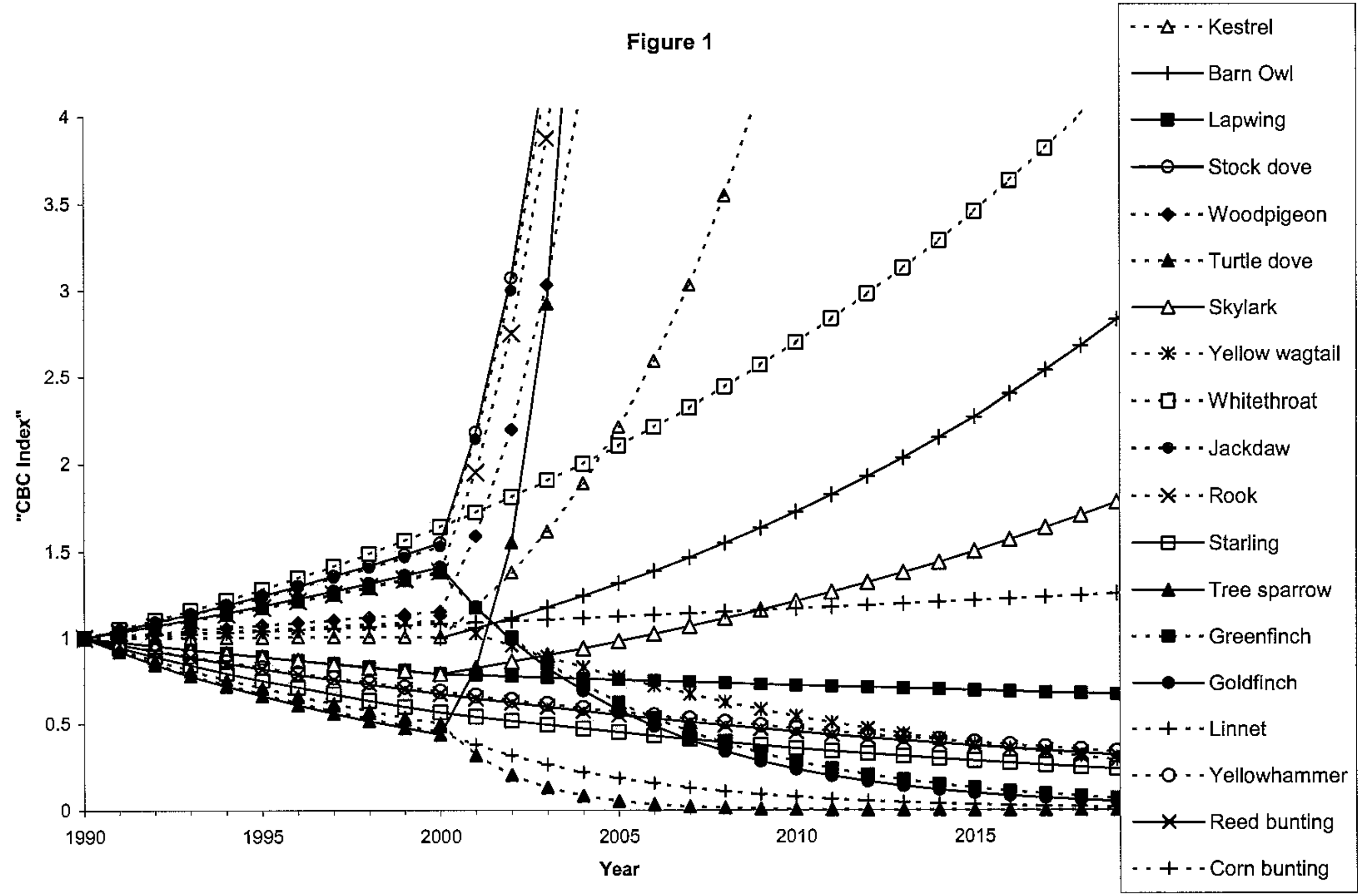

Figure 1 Plots of the average CBC population trends for the indicator species for 1990-2000 (linear trends on the log-scale), joined to trends as predicted by models based on the parameter estimates in Table 1. The CBC does not monitor Barn Owl, so no 1990-2000 trend could be calculated for this species.

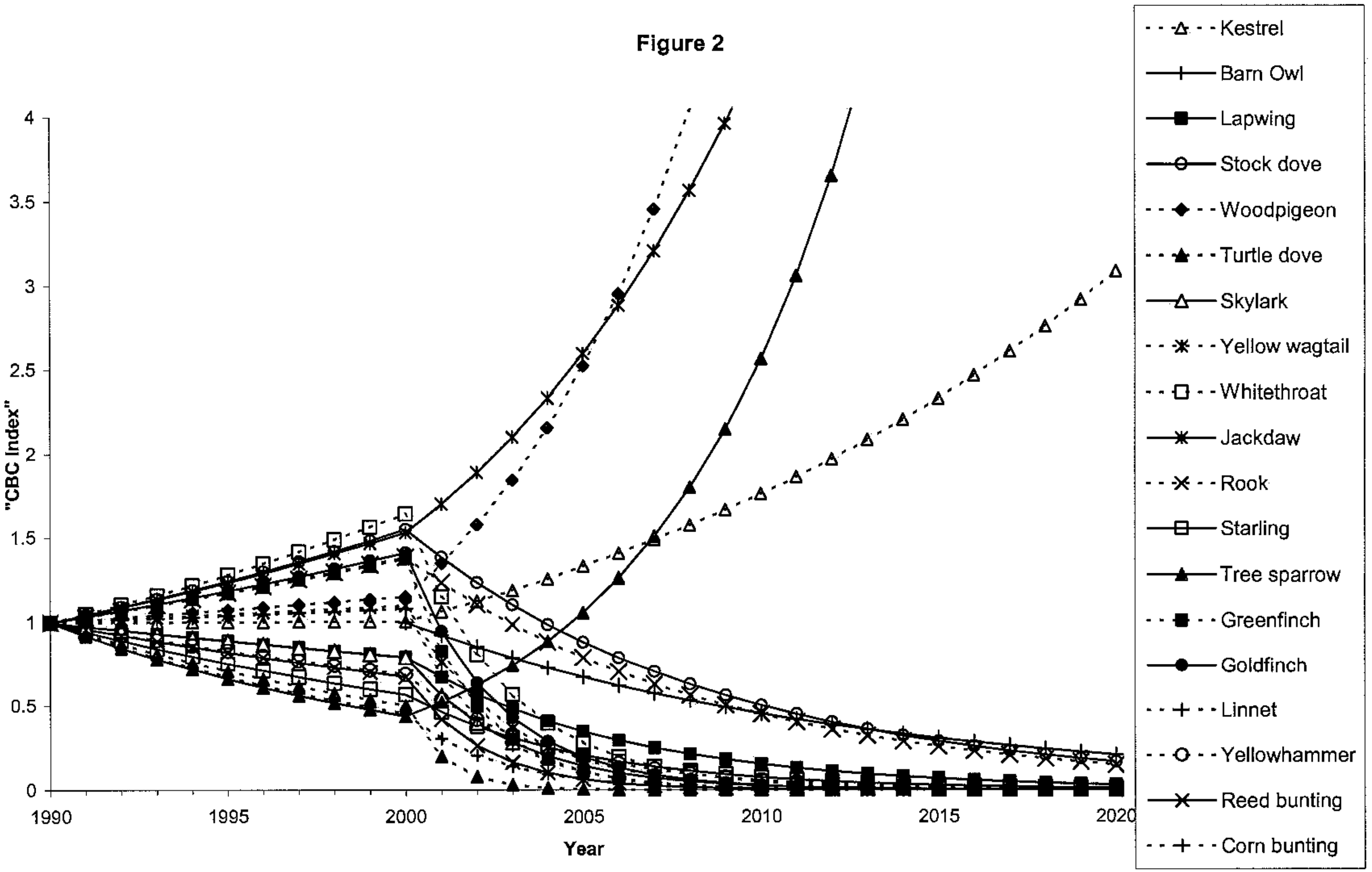

Figure 2 Plots of the average CBC population trends for the indicator species for 1990-2000 (linear trends on the log-scale), joined to trends as predicted by models based on lower 95% confidence intervals for the parameter estimates in Table 1. Parameter estimates for which no confidence interval data were available were left as the values shown in Table 1.

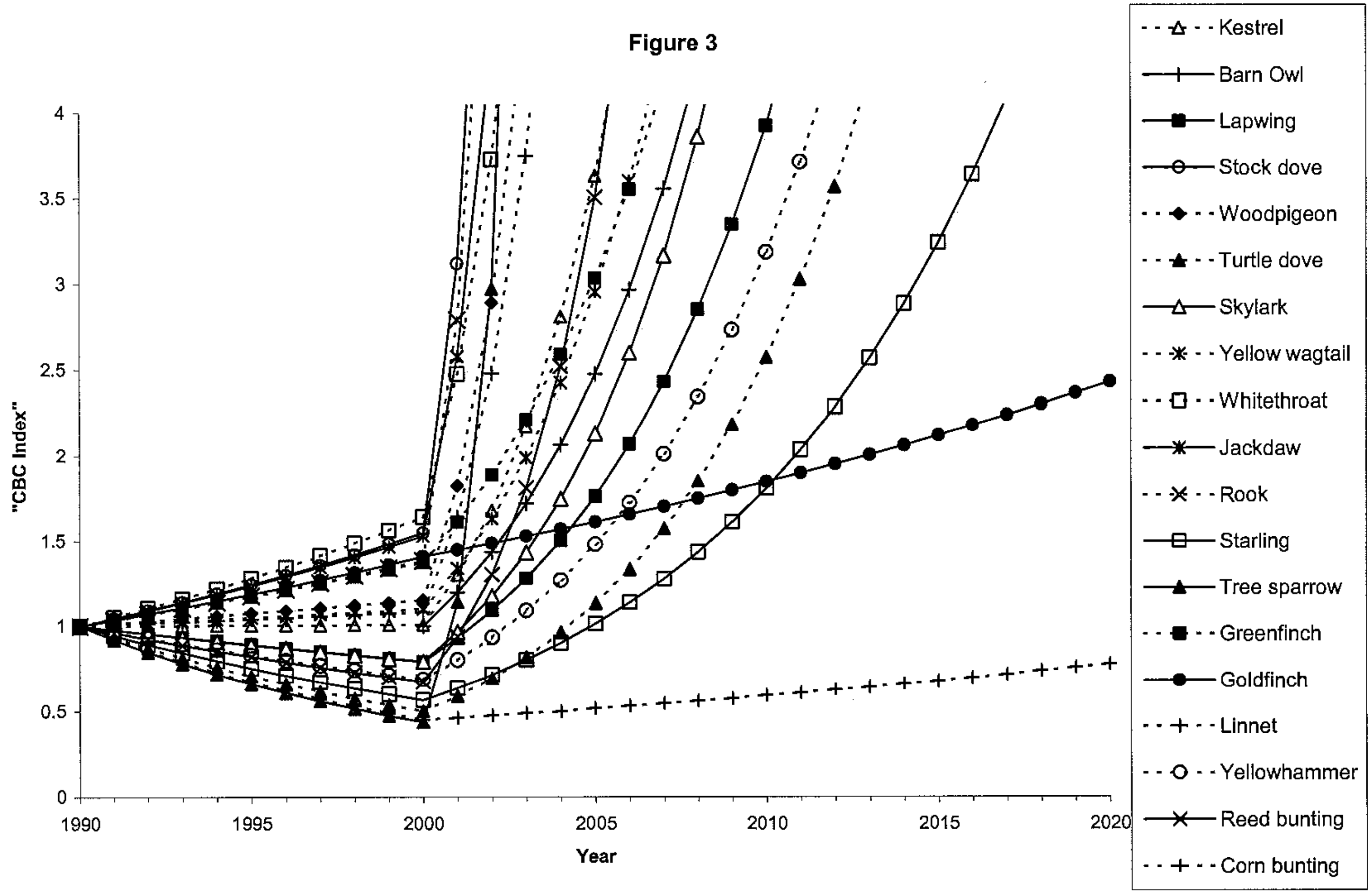

Figure 3 Plots of the average CBC population trends for the indicator species for 1990-2000 (linear trends on the log-scale), joined to trends as predicted by models based on upper 95% confidence intervals for the parameter estimates in Table 1. Parameter estimates for which no confidence interval data were available were left as the values shown in Table 1.

1.5 Adjusted Model Results

For each species, we adjusted the values of each demographic rate input into our models by small amounts until the population growth rate predicted by the models matched that measured by the CBC between 1990 and 2000. We began by varying the parameter estimates that were the least reliable (i.e. ones based on other species or on guesswork, or those with the widest 95% confidence intervals), and then moved onto other parameters if it proved necessary. For Barn Owl, our original demographic parameter estimates suggested that the population is increasing, whereas it is believed that there has been a continuing decline, although firm evidence is lacking (Toms *et al.* 2000, M.P. Toms, pers. comm.). We therefore generated a more plausible, declining model trend for the species using the parameter values representing the lower end of the 95% confidence intervals for survival and fledgling production (no other confidence intervals were available). The juvenile survival and fledgling production estimates for Woodpigeon both incorporated early juvenile mortality and one or both could therefore be regarded as being too low (Table 1). The initial estimates for these parameters produced a *steeper* population increase, however, than the one observed (Figure 1), so demographic rates had to be reduced rather than increased to produce a fit to the CBC trend. We opted to reduce all three Woodpigeon demographic rates by a small amount because any or all could have been too high. Among the other species, the size of the adjustments to the demographic parameters varied, but it was only necessary to use values outside the range of the measured 95% confidence intervals for any parameter (where confidence intervals were available) for Kestrel, Jackdaw and Tree Sparrow. Note that the parameter estimates for Kestrel may have suffered from particular problems with sampling bias (see below). As colonial species that commonly use nest-boxes, Jackdaw and Tree Sparrow are also species for which sampling biases could easily occur when significant amounts of ringing or nest record information come from just a few colonies. The final set of adjusted parameter estimates is presented in Table 3.

Table 3 Adjusted demographic parameter estimates used to produce population models that predict future CBC trajectories continuing with the same trend as found for 1990-2000 in the absence of any demographic changes.

Species	Adult survival	First-year survival	Juvenile survival	Post-fledging survival	Fledglings per attempt	Number of attempts
Kestrel	0.548	-	0.259	-	3.50	1
Grey Partridge						
Lapwing	0.635	-	0.42	-	1.63	1
Stock Dove	0.41	-	0.34	-	1.25	3
Woodpigeon	0.52	-	0.45	-	1.7*	-
Turtle Dove	0.58	0.42	-	0.8	1.30	1.6
Barn Owl	0.505	-	0.239	-	2.80	1.25
Skylark	0.63	-	0.24	-	1.45	2
Yellow Wagtail	0.552	0.477	-	0.52	2.95	1.25
Whitethroat	0.389	0.279	-	0.826	2.87	2
Jackdaw	0.71	-	0.323	-	2.07	1
Rook	0.78	-	0.3	-	1.7	1
Starling	0.568	0.364	-	0.38	3.63	1.5
Tree Sparrow	0.35	-	0.29	-	3.03	1.3
Greenfinch	0.435	0.39	-	0.67	2.27	2
Goldfinch	0.408	0.382	-	0.7	2.32	2
Linnet	0.33	0.286	-	0.851	2.48	2.25

Species	Adult survival	First-year survival	Juvenile survival	Post-fledging survival	Fledglings per attempt	Number of attempts
Yellowhammer	0.536	0.529	-	0.452	1.43	2.5
Reed bunting	0.492	0.398	-	0.593	1.99	2
Corn bunting	0.51	0.42	-	0.7	2.23	1.25

* This figure refers to fledged young per pair because the data input into the model for Woodpigeon were in this form (Table 1).

Table 3. Continued.

The result of entering the parameter estimates in Table 3 into predictive models is shown in Figure 4 and the indicator trend produced by combining these trends is shown in Figure 5. These species-specific and combined trends show realistic possible population trends if nothing changed in the demography of each species, except that they take no account of the density-dependence that is likely to affect most parameters in some way, at least at very high or very low densities. Density-dependence cannot be accounted for in these models in any simple way, so we will first examine the implications of the models without considering it; the subject is dealt with in greater detail below.

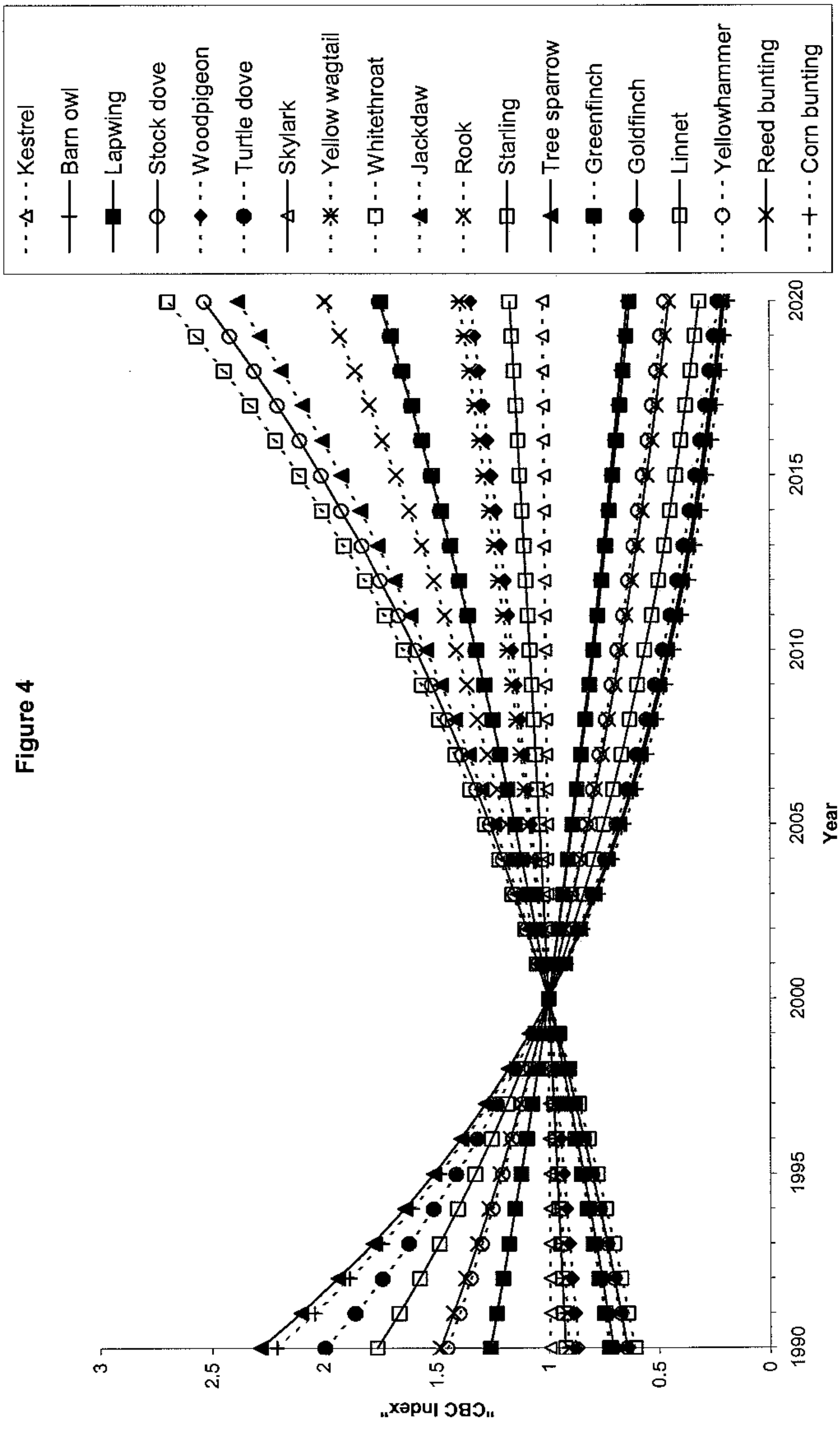

Figure 4 Projected population trends for each species using the adjusted demographic parameter estimates listed in Table 3.

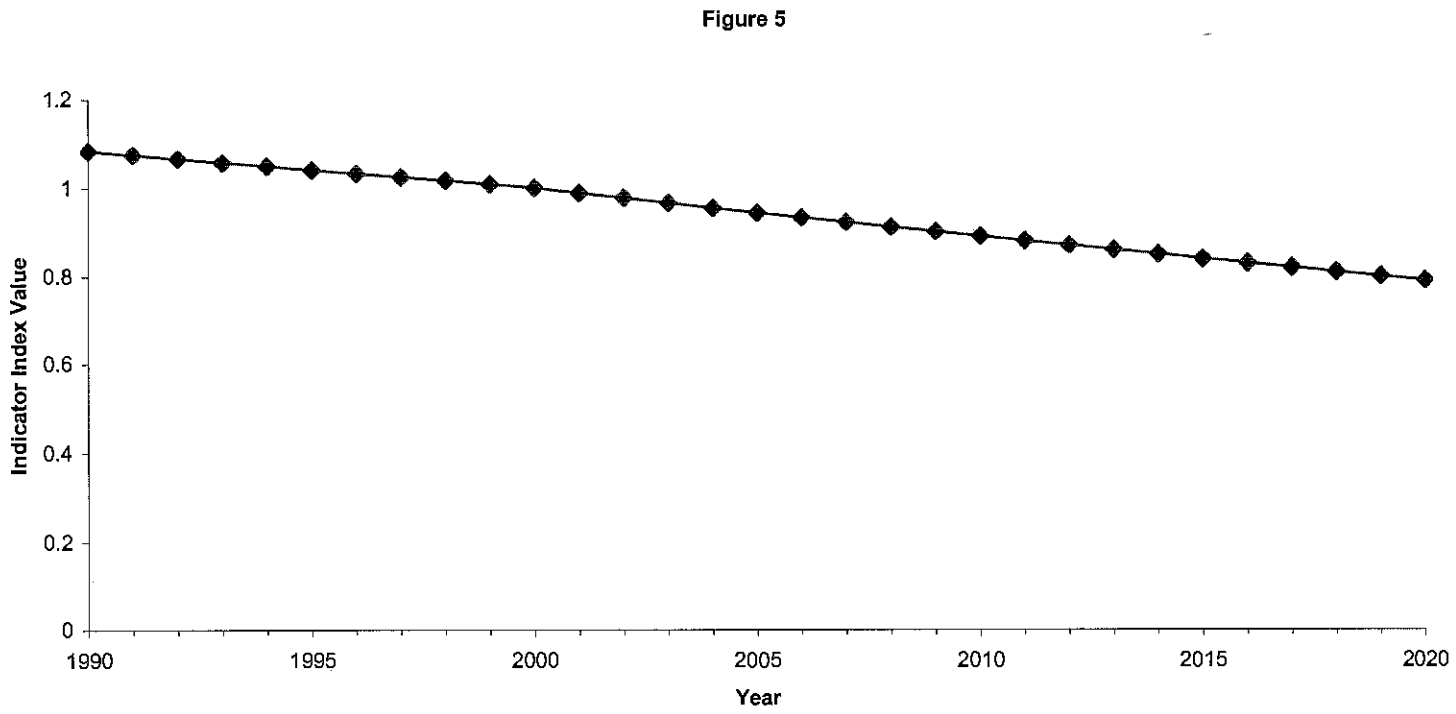

Figure 5 Projected indicator trend derived from the species-specific trends shown in Figure 4.

1.6 Key Demographic Rates in Population Change

The causes of population declines (and increases) can be elucidated by identifying the demographic rate that changed to drive the decline. It is then reasonable to suppose that the most easily achievable and effective demographic change to attain a conservation objective – probably the most cost-effective one in terms of conservation or management action – would be the reverse of the one that had driven the population change. Note, however, that any other change with an equivalent effect on abundance could occur and be equally satisfactory as a conservation outcome. We have reviewed the literature to identify, as far as they are known, the key demographic rates underlying the major population changes over the last 30 years or so of the 20 species in the farmland indicator set. The results of this review will be valuable in determining which demographic rates are likely to form the best targets for effective conservation action for each species.

Relevant studies have previously been conducted using BTO data sets for most of the species for which the Trust holds adequate data, and many other studies provide further information on these species, as well as insights into the demography of others. An exception to this pattern is the Kestrel, for which no full demographic analysis had previously been conducted but for which BTO historical data are available and should be adequate. We have therefore conducted a full integrated population modelling analysis for Kestrel in an attempt to identify the key demographic parameter(s) in the species' decline. A draft manuscript describing this work is included in this report as Appendix 1. The results suggest that changes in both juvenile survival and in fledgling production were probably responsible for the cessation of the species' decline and the ensuing period of stability. However, some bias or sampling error in one or more of the CBC, NRS and ring-recovery data sets prevents the drawing of firm conclusions about the demographic mechanism behind the onset of the Kestrel's principal period of decline.

The information we have collected on the key demographic rates for all 20 species is summarised in Table 4. It suggests that no single demographic rate can be implicated as key to population change across species and therefore that targeting one parameter, across all species, for conservation action (say, through seasonally concentrated habitat management) is unlikely to be effective. A recent review of the demography of farmland birds found that, at the national scale, survival was more often the key demographic rate in population change for declining, passerine species, but that nidifugous and increasing species tended to have been affected more significantly by changes in breeding success (Siriwardena *et al.* 2000b). However, there is also evidence for several species that changes in breeding performance have been important at the local scale (e.g. Brickle *et al.* 2000).

Species	Key demographic rate
Kestrel	Juvenile survival and/or fledglings per attempt
Grey Partridge	Fledglings per attempt
Lapwing	Fledglings per attempt
Stock Dove	Survival?
Woodpigeon	Survival pre-1970, fledglings per attempt post-1970
Turtle Dove	Number of breeding attempts
Barn Owl	Survival?
Skylark	Number of breeding attempts
Yellow Wagtail	?
Whitethroat	Adult/first-year survival
Jackdaw	Fledglings per attempt?
Rook	?
Starling	Survival?
Tree Sparrow	Survival?
Greenfinch	None (basically stable population)
Goldfinch	Survival
Linnet	Fledglings per attempt
Yellowhammer	Survival?
Reed Bunting	Survival
Corn Bunting	Survival? Number of breeding attempts?

Table 4 Key demographic rates in the population changes of the 20 farmland bird indicator species. References: O'Connor & Mead (1984), Baillie & Peach (1992), Potts (1986, and related papers), Inglis *et al.* (1994), Browne & Aebischer (2001), Wilson *et al.* (1997), Siriwardena *et al.* (1998b, 1999, 2000ab, 2001), Peach *et al.* (1999), Brickle *et al.* (2000), Baillie *et al.* (2001) and this project. Question marks following parameter descriptions indicate patterns that are suggested by the data but for which the evidence is not strong. Question marks are used alone where we could find no studies of the demography of population change.

1.7 Changes in Demography Required to Halt Population Declines

The PSA target with respect to the farmland bird indicator is to reverse the declining trend in the index by 2020. Although this could be done by any mathematically adequate set of increases in population growth rate across the 20 species, the increases would have to be achieved in the currently declining species for the meeting of the target to be consistent with the spirit in which it was established. We have therefore defined the goal of changing the indicator trajectory as achieving an indicator growth rate of greater than one with all the population growth rates for the now-declining species also being greater than one. In the first instance, we have postulated that the population growth rates of any species that are currently increasing (at whatever rate) remain the same in the future. If we assume that the changes in trend direction occurred "now", i.e. from 2000 in the context of the most up-to-date data available for our analyses, the postulated species-specific population trends would appear as in Figure 6. Combined together, these trends mean that the indicator trend would have to be increasing (at a growth rate of at least 1.014), as shown in Figure 7, for the PSA target to be met.

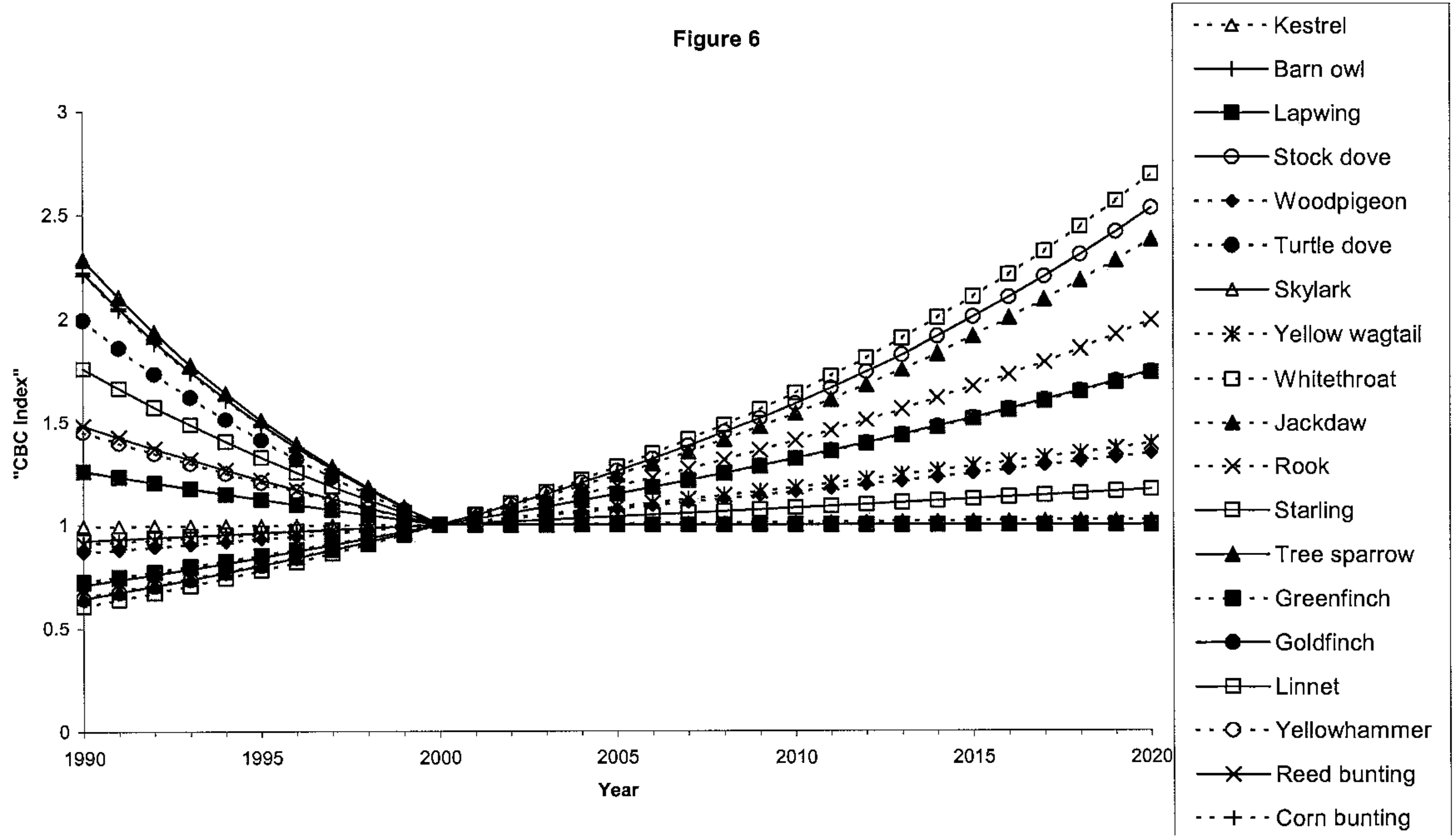

Figure 6 Projected population trends for each species that would meet the PSA target: all declines converted to stability. The trends shown postulate a change in trend direction "now", i.e. immediately after the time represented by the most recent available population data, but this change could occur at any time before 2020 and the PSA target would still be met.

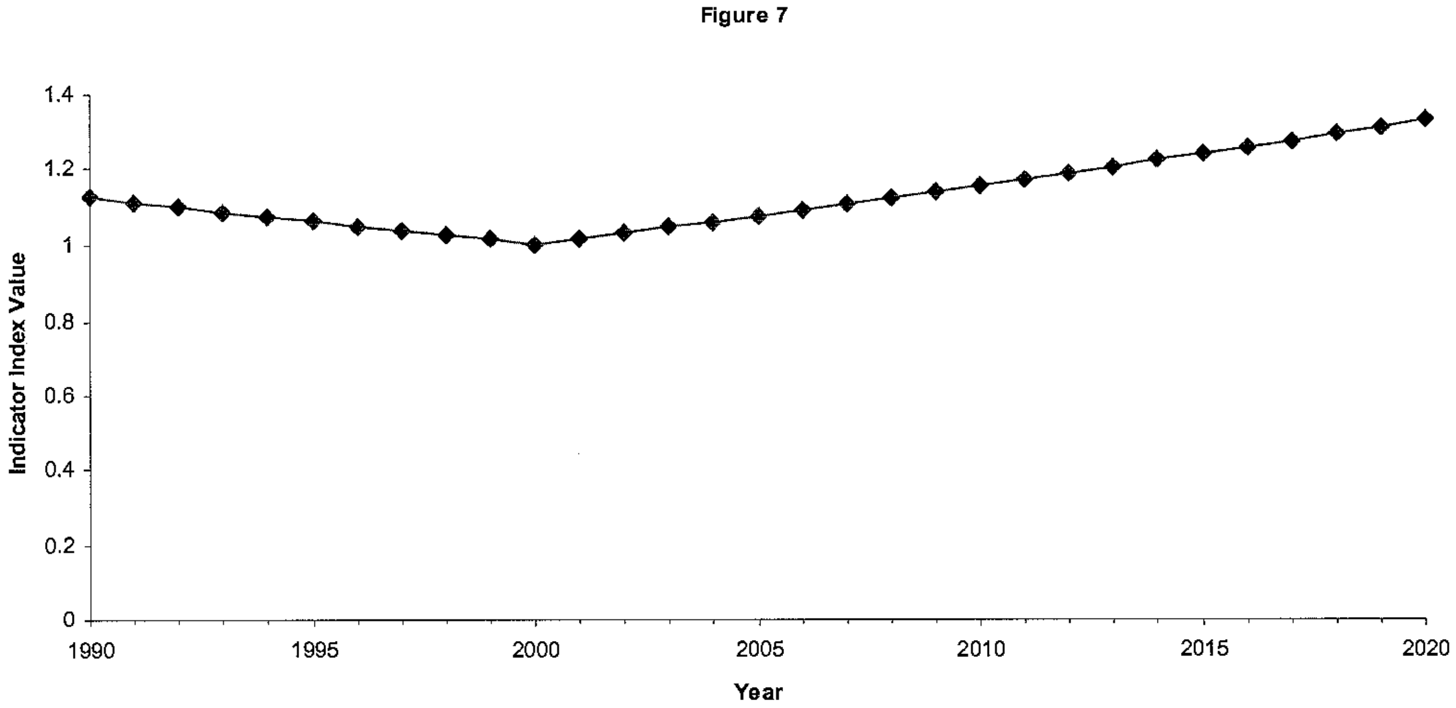

Figure 7 Projected indicator trend derived from the species-specific trends shown in Figure 6.

From here on, we consider the demographic changes necessary to change the eight species' declines into stability (because it is easier to define a growth rate of one in equations than any "slight increase"): any increment in the demographic rates leading to stability will tend to lead to a population increase for the species. The approach we have taken is to identify the factors by which survival or productivity would have to be increased to generate a population growth of one. For survival, we have considered changing the adult and first-year/juvenile values in tandem as well as each independently. Following the algebraic definitions used in eqn 1 and eqn 2 above, we defined the population growth rate PGR as N_{t+1}/N_t and productivity P as $FPA \times NA \times S^{PF}$ (where first-year survival was used) or $FPA \times NA$ (where juvenile survival was used). We can then define algebraically a "survival inflation factor" α, denoting the amount that survival has to be increased to meet the target PGR; re-arranging eqn 1 and eqn 2, using the current demographic rates gives:

$$PGR = \alpha S^{AD} + (0.5 \times \alpha S^{FY/JV} \times P),$$

so

$$\alpha = \frac{PGR}{S^{AD} + (0.5 \times S^{FY/JV} \times P)}$$

for changes in both adult and first-year/juvenile survival. Likewise,

$$PGR = \alpha S^{AD} + (0.5 \times S^{FY/JV} \times P),$$

so

$$\alpha = \frac{PGR - (0.5 \times S^{FY/JV} \times P)}{S^{AD}}$$

for changes in adult survival only and

$$PGR = S^{AD} + (0.5 \times \alpha S^{FY/JV} \times P),$$

so

$$\alpha = \frac{PGR - S^{AD}}{0.5 \times S^{FY/JV} \times P}$$

for changes in first-year/juvenile survival only. Defining PGR in terms of the same demographic parameters and a "productivity inflation factor" β, analogous to α for survival,

$$PGR = S^{AD} + (0.5 \times S^{FY/JV} \times \beta P),$$

so

$$\beta = \frac{PGR - S^{AD}}{0.5 \times S^{FY/JV} \times P}.$$

(Note that the calculation of β for productivity is identical mathematically to that for α for first-year/juvenile survival, although the two parameters are quite different biologically.) By setting PGR=1 and entering the demographic parameter estimates from Table 3 into these equations, we estimated the percentage changes in survival and productivity required to produce stable trends and the concomitant absolute parameter values. The results of this process are shown in Table 5.

Species	**Demographic rate being changed**								
	Adult and first-year/juvenile survival			**Adult survival only**		**Juvenile survival only**		**Productivity**	
	% increase	**Necessary adult survival**	**Necessary juvenile survival**	**% increase**	**Necessary adult survival**	**% increase**	**Necessary jnvenile survival**	**% increase**	**Necessary productivity**
Lapwing	2.3	0.650	0.430	3.6	0.658	6.6	0.448	6.6	1.738
Turtle Dove	7.6	0.624	0.452	12.2	0.651	20.2	0.505	20.2	2.000
Barn Owl	8.3	0.547	0.259	15.2	0.582	18.4	0.283	18.4	4.142
Skylark	2.4	0.645	0.246	3.7	0.653	6.7	0.256	6.7	3.083
Starling	5.9	0.602	0.385	9.8	0.624	14.8	0.418	14.8	2.374
Tree Sparrow	8.6	0.380	0.315	22.5	0.429	13.8	0.330	13.8	4.483
Yellowhammer	3.8	0.556	0.549	6.9	0.573	8.6	0.574	8.6	1.754
Reed Bunting	4.0	0.512	0.414	7.9	0.531	8.2	0.431	8.2	2.553
Corn Bunting	8.8	0.555	0.457	15.8	0.591	19.7	0.503	19.7	2.333

Table 5 Changes in and absolute values of annual survival and productivity required to generate stable trends for the eight declining species. (Note that the percentage increases for each of adult and juvenile survival alone are identical because they are identical mathematically – see text.).

It is clear from Table 5 that considerably smaller percentage changes need to be made in survival than in productivity to achieve the same outcome in terms of the population trend. This is the case especially when adult and first-year/juvenile survival are varied in tandem. In reality, the survival prospects of an individual are likely to depend partly on its dominance status and competitive ability, which are probably related to age (among other factors), but the survival rates of different age-classes are unlikely to be entirely independent. A likely scenario is that both adult and first-year survival would be affected by a cold winter or habitat deterioration, but that the effect on the younger birds would be more pronounced. In the context of this study and the categories in Table 5, this means that real changes in the environmental influences on survival (such as enhanced winter food availability) will affect populations in a manner somewhere between adult and first-year/juvenile survival being varied by the same factor and first-year/juvenile survival alone changing. The relative effect on the survival of the different age-classes will vary from species to species according to the age-classes' relative competitive abilities and the extent to which individuals compete directly (which will be higher, for example, for more social species).

Species	Observed percentage change in demographic rate		
	Adult survival	First-year survival	Fledglings per breeding attempt
Stock Dove	-	-	7.7
Turtle Dove	-	-	34.2
Swallow	3.3	5.2	
Skylark	-	-	36.5
Pied Wagtail	4.6	25.6	-
Wren	5.1	36.5	-
Dunnock	4.3	14.6	-
Robin	7.8	9.8	-
Blackbird	8.9	11.9	-
Song Thrush	23.6	42.3	-
Mistle Thrush	38.7	48.2	-
Sedge Warbler	38.5	10.3	-
Lesser Whitethroat	129.7	73.7	-
Whitethroat	122.8	59.4	-
Blackcap	7.1	34.3	-
Chiffchaff	98.4	89.0	-
Willow Warbler	25.1	12.0	-
Long-tailed Tit	33.2	41.5	-
Blue Tit	11.9	12.6	-
Great Tit	24.3	34.3	-
Starling	16.8	17.6	-
Tree Sparrow	39.8	16.6	17.9
House Sparrow	16.4	58.0	-
Chaffinch	12.2	0.6	10.8
Greenfinch	13.8	9.4	9.4
Goldfinch	27.5	13.1	50.3
Linnet	6.4	14.7	27.6
Redpoll	30.8	22.8	-
Bullfinch	12.9	10.9	62.1
Yellowhammer	24.1	26.4	192.5
Reed Bunting	0.7	30.7	46.2
Corn Bunting	-	-	114.7

Table 6 Percentage changes in demographic rates between periods of different population trend direction. Data on changes in survival and breeding performance per breeding attempt were taken from Siriwardena *et al.* (1998, 2000a) and the difference between the maximum and minimum values of each parameter across between two and six historical time periods, each of which had a different population trend, was recalculated as a percentage of the minimum value.

The greater percentage changes needed to achieve changes in population trends through adjusting productivity suggest that (all else being equal) trying to improve survival might prove to be the more effective conservation policy. However, it is noteworthy that the achievement of a single extra breeding attempt per year for the average pair of a multi-brooded species like Skylark could lead to an increase in productivity of the order of 20%. In addition, and more generally, quantitative data are lacking on how much any demographic rate might be increased at the national scale by any given change in the environment such as the provision of extra habitat or resources. In other words, we do not know whether achieving a certain percentage increase in survival is easier or more difficult than achieving the same increase in productivity. Nevertheless, variations of up to ±10% would fall within the 95% confidence intervals for many of the survival and fledgling production parameter estimates we have used, including many we would regard as being reasonably precise. Comparison of the percentage changes in Table 5 with those previously measured in historical analyses of BTO data (Table 6) also show that changes of that order have recent precedents, suggesting that they should be achievable in practice.

1.8 Accounting for Density-Dependence

1.8.1 Methods

A potential problem with all the models we have presented thus far is that demographic rates are actually likely to change with density so that, as abundance changes under the influence of given values of the various demographic rates, the demographic rates themselves will be affected. The effect of such density-dependent variation on population trajectories will be to tend to force them towards equilibrium values, i.e. progressively slowing declines as density falls and progressively slowing increases as density rises. Predicted trends that take no account of density-dependence may therefore be misleading. Its effects are likely to be greatest at very low and very high densities: the latter may be particularly likely to complicate our models for those species for which we have predicted continuing declines.

Although we can recognise the potential importance of density-dependence in demographic rates, it is difficult to ascertain the form of the relationships. As well as a genuine dependence on density, i.e. the influence of the relative intensity of intraspecific competition on demography, patterns such as the elimination during a decline of poorer quality individuals before higher quality ones or the increase in the proportion of the remaining population that is in good quality habitat (or the converse patterns during an increase) may be indistinguishable mathematically at the population level (e.g. Siriwardena *et al.* 2000a). However, we can acquire some idea of the range of likely strengths of density-dependent relationships (in the broad sense of some negative relationship between a demographic rate and density at a large scale) from the literature. Such relationships are often parameterised in the form of *k*-values in a key factor analysis (Varley & Gradwell 1960, Varley *et al.* 1973, Southwood 1978, Begon *et al.* 1986), and we have followed this scheme to investigate the possible effects of the changes in density on our models.

We used equations derived from the *k*-value approach to ask what the implications of density-dependence were for the changes in demographic rates that our models suggested were required to reverse population declines among the farmland indicator species. These equations described the demographic parameter values required to produce a population level at equilibrium equal to the goal abundance level used above (reflecting stability at current levels), parameterised in terms of density, the demographic parameter values estimated to be necessary to produce stability in a density-independent model (Table 5) and a given strength of density-dependence.

We investigated the potential effects of density-dependence on the key demographic rates of adult survival, first-year or juvenile survival and productivity (production of young birds up to the beginning of the period over which first-year or juvenile survival was measured, defined as P above). We considered density-dependence from a starting point of relationships between k-values (mortality or loss factor) and density (taken to be approximately equivalent to abundance on farmland because the national area of the latter is unlikely to have changed significantly). We used linear models relating k-values for a given demographic rate to the logarithm of abundance (McCleery & Perrins 1985, Baillie & Peach 1992), as follows:

$$k = \text{slope} \times \log_{10}(\text{abundance}) + \text{constant}.$$

For adult and first-year/juvenile survival rate k-values k_s, this expression took the form:

$$k_s = (d_s \times \log_{10}D) + C_s, \qquad \text{eqn 3}$$

where D represents autumn density (of adults and juveniles, combined), d_s is the density-dependent slope parameter and

$$C_s = \log_{10}(1/S_c) - (d_s \times \log_{10}D_c), \qquad \text{eqn 4}$$

where S is the (adult or juvenile) survival rate of interest (value predicted to reverse the decline in the absence of density-dependence, Table 5) and D_c is current density, estimated as the density of breeding individuals plus their annual productivity P (as defined above) for the period 1990-2000 (a combination of the demographic rate data in Table 3 and the mean abundance index for the period as indicated by the constant population growth rate models whose results are summarised in Table 1) (Baillie *et al.* 2000). The k-value for a survival rate is related to the survival rate itself according to the following equation:

$$S = \frac{1}{10^{k_s}} \qquad \text{eqn 5}$$

We could then calculate the value for S that would produce a final density D_f (level of abundance) that would indicate net population stability by entering this value of D_f and a pre-determined intensity of density-dependence d_s (see below) into eqns 3 and 5, as follows:

$$S = \frac{1}{10^{(d_s \times \log D_f) + C_s}}$$

Defining a k-value with respect to productivity is more difficult than with respect to survival, because productivity (fledgling production per breeding attempt × number of attempts × post-fledging survival) does not have an obvious maximum value. We defined the maximum productivity of autumn juveniles as the product of the maximum clutch size (reflecting the likely maximum number of fledglings per breeding attempt) and the likely maximum number of broods, thereby incorporating a post-fledging survival rate of 100% for species for which this parameter was separated from first-year survival. (For the other species, the density-dependence in productivity refers to the production of fledglings and the density-dependence in first-year (juvenile) survival to the entire first year from fledging onwards, but this difference has no effect on the workings of our models.) Thus, the k-value for productivity k_p that we have used was defined as follows:

$$k_p = \log_{10}((\text{maximum clutch size} \times \text{maximum number of attempts}) / (\text{observed clutch size} \times \text{observed number of attempts} \times \text{observed post-fledging survival probability})).$$

The maximum clutch sizes and numbers of attempts we have used are shown in Table 7.

Species	Maximum clutch size	Maximum number of breeding attempts
Kestrel	6	1
Lapwing	5	1
Stock Dove	3	4
Woodpigeon	3	3
Turtle Dove	3	3
Barn Owl	11	2
Skylark	7	4
Yellow Wagtail	7	2
Whitethroat	7	2
Jackdaw	7	1
Rook	7	1
Starling	8	2
Tree Sparrow	9	3
Greenfinch	7	3
Goldfinch	7	3
Linnet	7	3
Yellowhammer	6	3
Reed Bunting	8	2
Corn Bunting	7	2

Table 7 Values for maximum clutch size and numbers of breeding attempts used in the calculation of *k*-values for productivity. Values were determined by consultation of a combination of Cramp *et al.* (1977-1994) and the standard maximum parameters used in the BTO's Nest Record Scheme.

The form of the equation for the relationship between the productivity *k*-value k_p and density is the same as that used for survival and described above, i.e.

$$k_p = (d_p \times \log_{10} D) + C_p, \qquad \text{eqn 6}$$

but, in this case, d_p refers to effects on productivity, D represents density at the start of the breeding season (the density of adults before any juveniles are produced) and C_p is defined as

$$C_p = \log_{10}(MP/OP_c) - (d_p \times \log_{10} D_c), \qquad \text{eqn 7}$$

where MP denotes maximum productivity, OP_c denotes observed productivity (value predicted to be sufficient to generate stability in the absence of density-dependence, Table 4) and D_c denotes the current mean breeding density (estimated using the mean abundance index from linear CBC models for 1990-2000). The *k*-value for productivity is related to the observed productivity value itself as follows:

$$OP = \frac{MP}{10^{k_p}} \qquad \text{eqn 8}$$

We then estimated the value of OP required to produce a final abundance level D_f indicating net stability under a given strength of density-dependence d_p by entering the latter two values into eqns 6 and 8, as follows:

$$OP = \frac{MP}{10^{(d_p \times \log_{10} D_f) + C_p}}$$

Just as it was important to use estimates of demographic parameters that were as accurate as possible, it was important that we modelled density-dependence using realistic estimates of its likely strength. This presents problems because the difficulties in studying true density-dependence mean that relevant studies investigating the variations in demographic rates with respect to density tend to be rare, and studies on the species of interest at a large scale are even rarer. We used whatever measurements of variations in demographic rates with density that we could find in the literature to identify a realistic range of possible strengths of density-dependence in survival and productivity. If these measurements were not in terms of *k*-values, we converted them to this format, i.e. slopes of the relationship between the *k*-value and density (i.e. values for d_s and d_p). We then examined the range of values of this slope and selected one value indicating each of relatively high-, medium- and low-intensity density-dependence which we then applied to all the species considered.

The published data we referred to in investigating evidence of density-dependence in survival is summarised in Table 8; from these estimates we chose values of *d* of 0.15, 0.50 and 0.85 to represent the three different strengths of density-dependence.

The data we use to investigate the range of intensities of density-dependence in productivity are summarised in Table 9. Negative values in Table 9 indicate patterns consistent with changes in productivity having *caused* population change (rather than *responding* to it). The other values suggest that reasonable values for low-, medium- and high-strength density-dependence in productivity would be 0.05, 0.40 and 0.80, respectively.

Species	Slope for survival k-value against $\log_{10}$(density)	Source
Pied Flycatcher	0.07	Virolainen 1984
Blackcap	0.20	Baillie & Peach 1992
Swallow	0.22	Møller 1989
Sedge Warbler	0.34	Baillie & Peach 1992
Pied Flycatcher	0.47	Järvinen 1987
Whitethroat	0.48	Baillie & Peach 1992
Willow Warbler	0.50	Baillie & Peach 1992
Great Tit	0.66	McCleery & Perrins 1985
Redstart	0.93	Järvinen 1987
Pied Flycatcher	1.00 / 0.26*	Stenning *et al.* 1988

* the latter value refers to the full period of this study, the former to a restricted run of years

Table 8 Values for the strength of density-dependence in annual survival taken from the literature.

Species	Slope for productivity k-value against $\log_{10}$(density)	Source
Whitethroat	-0.13	Baillie & Peach 1992
Linnet	-0.13	Siriwardena *et al.* 2000a
Stock Dove	-0.13	Siriwardena *et al.* 2000a
Redstart	-0.10	Järvinen 1987
Blackcap	-0.05	Baillie & Peach 1992
Swallow	0.03	Møller 1989
Sedge Warbler	0.03	Baillie & Peach 1992
Chaffinch	0.04	Siriwardena *et al.* 2000a
Pied Flycatcher	0.05	Stenning *et al.* 1988
Pied Flycatcher	0.06	Järvinen 1987
Greenfinch	0.10	Siriwardena *et al.* 2000a
Pied Flycatcher	0.13	Virolainen 1984
Willow Warbler	0.19	Baillie & Peach 1992
Tree Sparrow	0.19	Siriwardena *et al.* 2000a
Reed Bunting	0.25	Siriwardena *et al.* 2000a
Bullfinch	0.31	Siriwardena *et al.* 2000a
Skylark	0.40	Siriwardena *et al.* 2000a
Turtle Dove	0.48	Siriwardena *et al.* 2000a
Goldfinch	0.83	Siriwardena *et al.* 2000a
Yellowhammer	0.83	Siriwardena *et al.* 2000a
Corn Bunting	2.76	Siriwardena *et al.* 2000a

Table 9 Values for the strength of density-dependence in annual survival taken from the literature. Data that were not already in the *k*-value format were converted to it using the maximum productivity approach described above. Note that the extreme value for Corn Bunting was based on a regression between only two points. The negative values show cases where variation in productivity may have driven changes in density rather than have *been* driven by them. The latter illustrate that the *k*-value approach is coarse and that the slope values shown incorporate the net effects of environmental influences and other factors in population dynamics together with those of density.

1.8.2 Results

The increases in survival or productivity needed to halt the nine declining species' population declines under different intensities of density-dependence are shown in Tables 10 and 11. These results suggest one clear message: as the strength of density-dependence that affects a demographic parameter being targeted increases, ever greater increases must be made in that parameter to achieve the same goal in terms of abundance. As an example, if Corn Bunting first-year survival were density-independent, it would need to be increased from around 42% (Table 2) to 50% (Table 10), but if it were subject to strong density-dependence, it would have to be increased to 72% (Table 10). The increases here represent instantaneous values that survival would have to take initially; any measured values would be lower, reflecting decreases as density rises towards population equilibrium. Given stronger density-dependence, therefore, the increases in demographic rates suggested in Table 5 could be seriously misleading under-estimates of the changes needed to halt population declines. We consider further the intensity of density-dependence that is likely, in reality, to affect farmland birds in the Discussion, with reference to the results of detailed modelling of data on Grey Partridge (Section 2).

Species	Strength of density-dependence							
	None		0.15		0.50		0.85	
	Adults	First-years/ juveniles	Adnlts	First-years/ juveniles	Adults	First-years/ juveniles	Adults	First-years/ juveniles
Lapwing	0.658	0.448	0.670	0.456	0.698	0.475	0.728	0.495
Turtle Dove	0.651	0.505	0.688	0.534	0.782	0.607	0.890	0.691
Barn Owl	0.582	0.283	0.621	0.302	0.722	0.351	0.839	0.408
Skylark	0.653	0.256	0.665	0.261	0.693	0.272	0.723	0.283
Starling	0.624	0.418	0.652	0.437	0.724	0.485	0.805	0.539
Tree Sparrow	0.429	0.330	0.459	0.353	0.536	0.413	0.627	0.483
Yellowhammer	0.573	0.574	0.590	0.591	0.631	0.633	0.675	0.677
Reed Bunting	0.531	0.431	0.547	0.444	0.588	0.477	0.632	0.513
Corn Bunting	0.591	0.503	0.630	0.536	0.732	0.623	0.850	0.723

Table 10 Effects of density-dependence on the survival rates necessary to produce stability in abundance. Adult and first-year/juvenile survival estimates are shown for a range of strengths of density-dependence (the survival of younger birds refers to first-years or juveniles according to the age-class for which data were available and shown in Table 3). The latter represent slopes for the relationship between a survival *k*-value and the logarithm of density (see text for details). The results for the situation with no density-dependence are taken from Table 5.

Species	Strength of density-dependence			
	None	0.05	0.40	0.80
Lapwing	1.74	1.75	1.82	1.91
Turtle Dove	2.00	2.04	2.32	2.69
Barn Owl	4.14	4.23	4.92	5.85
Skylark	3.08	3.10	3.23	3.39
Starling	2.37	2.41	2.68	3.02
Tree Sparrow	4.48	4.58	5.36	6.41
Yellowhammer	1.75	1.77	1.90	2.05
Reed Bunting	2.55	2.58	2.77	3.01
Corn Bunting	2.33	2.38	2.77	3.29

Table 11 Effects of density-dependence on productivity necessary to produce stability in abundance. Productivity estimates are shown for a range of strengths of density-dependence. The latter represent slopes for the relationship between the productivity *k*-value and the logarithm of density (see text for details). The results for the situation with no density-dependence are taken from Table 5.

1.9 Demographic Changes and the Indicator Trend

We have estimated the demographic changes that would be necessary to reverse the declines of the farmland indicator species and we have identified the demographic rates that might best be targeted in the first instance. This information is summarised in Table 12. The PSA target, as we have interpreted it, treats any currently increasing trends as satisfactory, so we have not suggested any demographic changes for species with such trends. There may, however, be valid conservation goals in making some of those trends more positive (see Discussion). For the declining species, we have suggested the type and order of demographic change that is, perhaps, most likely to be effective in changing their fortunes (Table 12), although our investigations of density-dependence suggest that larger changes may be required in practice. If the PSA target is to be met, the next challenge for conservation action is to identify how the demographic changes listed in Table 12 can be achieved. Reference to the appropriate research cited in Table 4 would suggest qualitative solutions to the species-specific problems; if quantitative information is not available, then field research measuring the demographic results of controlled conservation action would supply it.

Species	**Suggested demographic change**	**Minimum percentage change required***	**Minimum absolute change in the average of the suggested demographic rate***	
			Current value	**Required value**
Kestrel	Do nothing	-	-	-
Grey Partridge				
Lapwing	Increase fledglings per attempt	6.6	1.63	1.74
Stock Dove	Do nothing	-	-	-
Woodpigeon	Do nothing	-	-	-
Turtle Dove	Increase number of breeding attempts	20.2	1.6	1.92
Barn Owl	Increase survival	8.3	0.505, 0.239	0.547, 0.259
Skylark	Increase number of breeding attempts	6.7	2	2.13
Yellow Wagtail	Do nothing	-	-	-
Whitethroat	Do nothing	-	-	-
Jackdaw	Do nothing	-	-	-
Rook	Do nothing	-	-	-
Starling	Increase survival	5.9	0.568, 0.364	0.602, 0.385
Tree Sparrow	Increase survival	8.6	0.35, 0.29	0.380, 0.315
Greenfinch	Do nothing	-	-	-
Goldfinch	Do nothing	-	-	-
Linnet	Do nothing	-	-	-
Yellowhammer	Increase survival	3.8	0.536, 0.529	0.556, 0.549
Reed Bunting	Increase survival	4.0	0.492, 0.398	0.512, 0.414
Corn Bunting	Increase survival	8.8	0.51, 0.42	0.555, 0.457
	Increase number of breeding attempts	19.7	1.25	1.50

* Changes in survival refer to changes in adult and first-year/juvenile survival in tandem, with figures relating to adult survival given before those for first-year survival in each column

Table 12 Summary of the minimum changes in key demographic rates necessary to halt the decline in the farmland indicator and meet the Government's Public Service Agreement target. Significantly larger changes may be necessary if appreciable density-dependence occurs. Where pairs of figures are given for survival, adult survival rates are given before first-year/juvenile survival rates. Full sets of current parameter values are show in Table 3.

1.10 Discussion

1.10.1 How reliable are the models and predicted trends?

The utility of our results in informing policy clearly depends on the models and their resultant predicted trends being reliable. We discussed above some of the caveats that must be placed on our initial models in respect of data quality: those estimates were adjusted to make them consistent with one another and to provide a more realistic overall fit to the observed CBC data. However, the adjusted estimates do not necessarily represent the true demographic rates for the population monitored by the CBC. Any inaccuracy in these values will have reduced the accuracy of our predictions; some is inevitable when we were forced to use, at best, averages of the demographic parameters over a ten-year period. The only way to obtain more accurate estimates of current demographic parameter values is likely to be to use intensive field studies measuring the important variables at the national scale, especially for those species and demographic rates for which sample sizes in BTO data sets are always likely to be prohibitively small (e.g. Skylark survival). As well as this potential problem with accuracy in terms of the time period from which data are drawn, there is the more general issue of matching between data sets that we have already discussed. Any geographical mismatch between the populations sampled for data on abundance, survival and breeding success would make our models less reliable. Overall, we simply do not know how representative of national populations of the species concerned are our models combining data from different areas/habitats and time periods; we can only re-iterate that we estimated, selected and adjusted parameter values in order to make the models as reliable as possible. In the long term, many of these problems could be addressed by new analyses investigating the extent of biases in BTO and other data sets, either by cross-referencing the data with other environmental information or by conducting calibration studies using especially collected data.

The accuracy of the modelling process depends also on the structure of the models themselves. This structure was very simplistic, primarily because, given the levels of uncertainty already present in predictions from such simple models, adding complexity would only multiply the uncertainty and would probably not provide useful additional insights. Nevertheless, we must acknowledge that complicating factors could well occur, such as simultaneous changes in more than one demographic rate and further changes in a species' demography subsequent to the single change we have postulated. Our predictions must therefore be viewed as significant simplifications of the trends likely to occur in reality.

As we have discussed above, density-dependence could play an important role in determining population trends and the influence of any changes in demographic rates upon them. We fitted linear functions to the relationships between *k*-values and density, thus assuming that the relationship has a simple linear form. This assumption is typical of studies of density-dependence (e.g. McCleery & Perrins 1985, Baillie & Peach 1992) but is unlikely to hold at extremely low or extremely high densities, where the effects of small changes in density could well be, respectively, much smaller or much larger. Where there is a high risk of the production of extreme density values by predictive models, it is possible to try to account for such effects by fitting specific, realistic, non-linear functions to the density/*k*-value relationship (e.g. Baillie *et al.* 2000). For simplicity, however, and because our predictive models were based on real, national estimates of demographic rates and so should not produce extreme predictions, we restricted ourselves to the linear modelling approach.

We also opted for simplicity in terms of considering whether density-dependent reductions could occur in demographic rates other than the one being targeted and changed. For example, if survival increased and caused abundance to increase productivity might then fall as a result of the increased breeding density. We have ignored such scenarios because they would be too complicated to model and because they would require further unverifiable assumptions about relationships with density to be made, but we must acknowledge that they could occur. In practice, patterns like this would mean that there would be a value of the demographic rate being changed above which further increases would have no effect on abundance. If further increases in abundance were still desirable, conservation action would then have to address the factors limiting other demographic rates.

Our results suggest that the strength of any density-dependent effects on a demographic parameter could have a large influence on the outcome of conservation action designed to increase that demographic rate. Determining the intensity of density-dependence that actually operates for each species at the national scale could therefore be critical in making decisions about implementing conservation measures. It is inevitable that some influence of density on demographic rates occurs, although this influence could be too small to be detectable (density-dependence is generally hard to detect in population data: e.g. Holyoak & Baillie 1996a,b). We have little strong evidence about the real strengths of density-dependence at the national scale: the data in Tables 8 and 9 came either from local studies or from correlative data at the national scale from only a small number of data points. Density-dependence may well be stronger in local studies because environmental effects will be more homogeneous at this scale and relationships with density may be diluted as the scale increases. Extreme values are also more likely to result when relationships are estimated from only two to six points (e.g. the data in Table 9 from Siriwardena *et al.* 2000a). In the absence of firm evidence about the strength of density-dependence, however, the best advice may be to "aim high" in attempting to increase demographic parameters to reverse declines.

1.10.2 What do particular increases in demographic rates mean in terms of (required) environmental change?

For most species and for most demographic rates, we simply do not know the relationships between habitat features or environmental conditions and the values of the parameter in question. Some studies, particularly those of breeding birds, have compared demographic data among study areas with particular land-use types or with respect to variations in management in space or time (see Table 4). These could be used to predict the order of change in the variables considered that might bring about a given desired demographic enhancement, but such predictions would usually entail considerable extrapolation in terms of spatial scale. For future changes for which previous studies do not provide analogues at an appropriate scale, the demographic effects can only be the subject of expert judgment. Qualitative predictions such as that "increased winter food availability will increase winter survival for certain species" can be made fairly easily and are likely to be reliable, but quantitative predictions will be more difficult to get right. It would be useful if future research could assess the results of particular agri-environment provisions in this context, for example by comparing demography at an appropriate scale between matched areas with and without agri-environment enhancement.

1.10.3 The PSA target – its biological meaning and attainability

Notwithstanding the caveats we have identified above, the results suggest that the order of changes in demographic rates needed to meet the PSA target should be readily achievable through national conservation or policy action. The demographic changes required, especially if density-dependence tends to be weak, tend to fall within the range of variation that has been observed in the recent past (Table 12), suggesting that producing adequate increases should be feasible. Meeting the PSA target as defined here would mean that all the current declines (of the nine species listed, for example, in Table 11, plus Grey Partridge: see General Discussion) had been reversed: this would certainly represent an important success in the conservation of the species concerned.

One important caveat on the interpretation of trends and targets with respect to the indicator concerns one declining and one increasing species: Barn Owl and Rook. Both of these species are considered in the indicator as linear extrapolations of periodic data (Breeding Bird Atlas numbers of occupied 10×10 km squares and BTO Rook Survey counts, respectively), rather than as annual CBC indices. (CBC indices *can* be calculated for Rook but are unreliable.) Such extrapolation means that the contributions of these species to the indicator cannot change over time, whatever really happens to their populations in Britain. This makes the inclusion of such population trends potentially misleading, in terms of demographic changes, with respect to the indicator and the PSA target: even if conservation action produced a rapidly increasing trend in Barn Owl numbers, the indicator would be unaffected. The same would apply to a change in the fortunes of Rooks in Britain whereby their abundance started to fall. It is not the purpose of this study to review the methods used to construct the indicator or to suggest improvements, but it is undeniable that such extrapolated trends cannot play any useful role in the monitoring of individual species' future population changes or of biodiversity as a whole. A key point here is that the BTO/JNCC/RSPB Breeding Bird Survey (BBS) has now replaced the CBC as the principal monitoring scheme for terrestrial breeding birds in the UK, and will henceforth form the basis for the indicator. The BBS will not provide monitoring data for Barn Owls, but should provide a better guide to inter-annual changes in Rook numbers than the CBC and prevent any need for extrapolating data from the Rook survey (although the latter will remain a more accurate and complete measure in the years when it is conducted). Note, however, that the BBS has only been in operation since 1994, so baseline information on Rook numbers from which to determine targets will only ever be available from this date, unless the indices are calibrated against results from several future Rook Surveys. It would not be valid merely to add the BBS trend to a linear extrapolation of Rook Survey data from earlier years; recent research at the BTO has shown that BBS indices showed some agreement with CBC ones over the period of overlap between the surveys for the vast majority of species tested (not including Rook: Freeman *et al.* 2002), but this does not necessarily apply to other surveys. More generally, it would be better simply to omit from the indicator any species for which long-term monitoring data that are of reasonable quality and comparable to those for the other species are not available.

We have assumed that the population growth rates of the increasing species will remain constant: if these increases or the recoveries of some of the declining species became faster, the rate of increase of the indicator would go up, so that the target could be exceeded without all the individual species' declines having been halted. This kind of situation could well occur if environmental action to benefit target species as per Table 12 also benefited other, already increasing species. Any such effects would make it advisable to re-assess the indicator target to make sure that no continuing declines were obscured by the further increases. In general, it might be best, to consider species-specific trends and species-specific targets for each case of conservation concern, rather than composite indicator trends that tend to obscure the information that is really of interest for conservation and management.

Another point to note in the interpretation of the PSA target is that it deals only with reversing current downward population trends, rather than regaining previous levels of abundance. This has particular relevance for species which have previously declined, but which are now at or near a much lower equilibrium population size (leading to weakly increasing or only weakly declining recent population growth rates). In such cases, perhaps including Yellow Wagtail and Linnet, a target of population stability would not be very challenging and would not represent a particularly satisfactory outcome in terms of conservation. It might therefore be preferable to adopt more positive, species-specific targets for these species.

Finally, it is important, in general, to realise that the meaning of the indicator is limited by the data that go into it: meeting the PSA target as written would mean only that the 10 declining species were no longer declining and that the other 10 had not begun to decline. Many farmland bird species are not included in the indicator that has been used to set the PSA target. Some of these, such as Song Thrush, have been at the forefront of the research and publicity that biodiversity on farmland has attracted; others, such as Swallow can be regarded as farmland specialists at least as justifiably as species like Reed Bunting that *are* in the indicator, or are common in a range of habitats but with the bulk of their British populations being found on farmland (e.g. Blackbird, Bullfinch), something that could also be said for Starling. Because the indicator and therefore the PSA target do not include these species, it is quite possible that conservation problems on farmland that are indicated by their populations could persist after *indicator* decline has been reversed. Of course, it is also possible that ensuring that population growth occurs for the species that do contribute to the indicator will have a similar effect on other declining farmland species because the conservation or management action undertaken proves to have more wide-reaching benefits.

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CHAPTER 2 GREY PARTRIDGE

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2.1 Summary

- Brood production rate and female winter survival rate of the Grey Partridge are dependent on density whereas chick survival rate is not. Density dependence is bound up with the availability of nesting cover.
- The annual rate of change in the Grey Partridge population over the last decade, calculated from long-term partridge demographic data from Sussex, was very similar to the value of 0.926 obtained from BTO national CBC data.
- Based on data from the GCT Conservation Headland research programme, Grey Partridge chick survival rate was recalibrated in terms of the percentage of arable area made up of insect-rich brood-rearing habitat. Chick survival rate increased by 0.04 for every 1% increase in percentage arable area that is insect-rich.
- Ignoring density dependence in the partridge model leaves unrecognised the role of nesting cover as an important determinant of Grey Partridge density in the agricultural environment.
- Stabilising the population at its current level requires an increase in average amount of nesting cover from 4 to 4.3 km/km^2, or management of 3% of arable area as insect-rich brood-rearing habitat.
- Recovery of the population to its 1996 level depends on achieving 6.5 km/km^2 of nesting cover together with 3% of arable area as insect-rich brood-rearing habitat. Set-aside could be managed towards this end under existing prescriptions.

2.2 Introduction

The Grey Partridge *Perdix perdix* is one of the 20 species incorporated in the UK Government's farmland bird headline quality-of-life indicator. Because it is a quarry species and therefore of considerable economic importance (at least until its decline), it has been the subject of over 70 years of research. As a result, it is the most-studied of the 20 farmland bird species, and the only one for which detailed information on density dependence is available (Potts 1980, 1986). This report reviews the density-dependent relationships underpinning a full population model of Grey Partridge population dynamics, compares model predictions with ones obtained in the absence of density dependence, and examines how changes in the agricultural environment affect Grey Partridge population density.

2.3 Density-Dependent Demographic Model

This study draws heavily on the detailed information published by Dr G.R. Potts in the course of his long-term study of Grey Partridge population dynamics in a 62-km^2 area of the South Downs, Sussex (e.g. Potts 1986). This study began in 1968 and is ongoing. Every year after harvest, partridges have been counted, aged as adults or young, and sexed (adults only). Using the formulae in Potts (1986), these counts enable the following demographic rates to be estimated:

- brood production rate (number of spring pairs that produce chicks)
- nest survival rate (probability that a female that survives the summer will produce chicks)
- female summer survival rate (proportion of females present in the spring alive in the autumn)
- chick survival rate (proportion of chicks that survive from hatching to six weeks of age)
- female winter survival rate (proportion of females present in the autumn alive the following spring – includes immigration and emigration)

On the basis of these definitions, brood production rate is equal to the product of female summer survival rate and nest survival rate. Using key factor analysis (Varley & Gradwell 1963), Potts (1980, 1986) found that brood production rate and female winter survival rate varied with partridge density, whereas chick survival rate did not. He discovered that the relationship between brood production rate and density changed depending on whether predators were controlled or not, and that density dependence was bound up with the availability of nesting cover. As predator control is now much less widely practised than in the past, and as the focus here is on how Grey Partridge density responds to changes in the agricultural environment, this paper restricts itself to the situation without predator control. The density-dependent relationships reported by Potts (1980, 1986) are as follows, noting that k denotes a k-factor, i.e. $-\log_{10}(\text{survival rate})$:

Brood production rate:
$k_1 + k_2 = -0.22 + 0.58 \log_{10}(C \text{ pairs / km cover per km}^2)$ (1)
where k_1 is the k-factor for nest survival, k_2 is the k-factor for female summer survival and C is the mean clutch size (14 eggs).

Female summer survival rate:
$k_2 = 0.31\ (k_1 + k_2)$ (2)

Female winter survival rate:
$k_5 = -0.07 + 0.39 \log_{10}((\text{autumn females / km cover per km}^2) + 1)$ (3)

Taken together with shooting losses, which are assumed to occur (if they occur) before winter losses, and which have also been shown to be density-dependent, these relationships have been brought together in a model of partridge population dynamics (Potts 1980, 1986) that forms the basis for the current work. The original model took as input observed annual chick survival rates, but these were replaced here by a constant average rate so that effects of varying the average rate could be examined.

2.4 Demographic Parameters Ignoring Density Dependence

Typically, little is known about density dependence in the population dynamics of the other 19 species that make up the farmland bird quality-of-life indicator, so it is instructive to compare results from the Grey Partridge density-dependent model with ones from a Grey Partridge model that ignores density dependence.

Demographic parameters for such a model were estimated from the annual Sussex partridge counts for the last decade (1990-2000), which was thought to be representative of the current agricultural environment of the Grey Partridge. Demographic parameters was calculated on an annual basis, then averaged across years to give a mean and standard deviation. For comparability with BTO demographic parameters for other species, adult survival and first-year survival rates were calculated as the product of female summer survival rate and female winter survival rate, fledglings per attempt was calculated as the product of average clutch size (14 eggs) and nest survival rate, and juvenile survival rate was equated to chick survival rate (Table 1).

Based on these parameters, the annual rate of population change was 0.929, which is very close to the value of 0.926 estimated from BTO national CBC data for the Grey Partridge over the same period. This is equivalent to a decline of 77% over 20 years.

The joint effect of varying average female annual survival rate and annual young production per surviving female (fledglings per attempt x juvenile survival) on the annual rate of change is displayed in Figure 1. The limits of the axes were chosen to reflect approximately 95% of observed variation in annual means. The direction of the contours suggests that the rate of change is equally sensitive to changes in adult annual survival and production.

	Adult survival	First-year survival	Juvenile survival	Post-fledging survival	Fledglings per attempt	Number of attempts
Mean	0.411	0.411	0.294	-	8.57	1
SD	0.094	0.094	0.065	-	1.62	0

Table 1 Demographic parameter estimates for the Grey Partridge. Estimates are averages of annual values derived from autumn count data from the Sussex study, 1990-2000.

2.5 Comparison Between Models with and Without Density Dependence Effects of Changes in the Agricultural Environment

In the presence of density dependence, it becomes meaningless to talk of rate of population change because the modelled population no longer changes with a constant rate, but seeks to return to an equilibrium level where, by definition, the rate of change is zero (Aebischer 1991). Because population size at equilibrium is stable, the density-dependent demographic parameters become fixed, so that it is not possible to present a figure akin to Figure 1 for the density-dependent model (for the Grey Partridge, both adult survival and production contain density-dependent elements).

To compare outputs from the two types of model, what is needed therefore is a representation of population change in relation to variables that do not themselves depend on density. An obvious one is chick survival rate, the sole partridge demographic parameter that does not vary with density (Potts 1980, 1986). Another one is the amount of nesting cover (km/km^2), an environmental variable that enters into the density-dependent survival relationships (equations (1) and (3)).

Using nesting cover has the further advantage that it provides a direct link to changes in the agricultural environment, as it represents the density of non-cropped linear features in the landscape that are suitable for nesting, such as hedgerows and grass banks. Rands (1986a, 1987) showed how the attractiveness of different areas to Grey Partridges was determined by the amount and quality of available nesting habitat. It is also known that chick survival rate varies in relation to the availability of chick-food invertebrates in cereals, itself dependent on the pesticide regime (Potts 1973, Potts 1977, Green 1984, Potts 1986, Rands 1986b, Potts & Aebischer 1991). It seems therefore appropriate to attempt a recalibration of chick survival rate in terms of the amount of insect-rich brood-rearing habitat available in the landscape, to provide a second link to changes in the agricultural environment.

The GCT Conservation Headland research programme (Boatman & Sotherton 1988, Sotherton *et al.* 1989, Sotherton 1991, Sotherton *et al.* 1993) offers a sound scientific basis for the recalibration exercise. Conservation Headlands are the outer 6 m of a cereal crop that receive only selective herbicide treatment and no summer insecticide treatment, so that the crop understorey is rich in broad-leaved weeds and associated invertebrates. This constitutes an ideal insect-rich brood-rearing habitat for Grey Partridges. Sotherton *et al.* (1993) found experimentally that, over eight years, chick-survival rate averaged 0.23 on fully sprayed areas and 0.39 on areas with Conservation Headlands within the same East Anglian farms. They also reported similar data on mean brood size at six weeks in Hampshire and eastern England, with an average of 4.8 chicks on fully sprayed blocks and 7.3 chicks on blocks with Conservation Headlands. Using the conversion formula in Potts (1986), these are equivalent to chick survival rates of 0.19 and 0.37 respectively. Further information on chick survival rate in a fully sprayed situation come from Sussex, where chick survival rate averaged 0.22 over six years when broad-spectrum herbicides and insecticides had been used intensively on an area of 7 km^2 (Aebischer & Potts 1998).

It seems reasonable to conclude that under an intensive spraying regime, chick survival rate will be close to 0.20, which can be taken as the rate corresponding to zero availability of insect-rich brood-rearing habitat within the arable area. With Conservation Headlands, chick survival rate was around 0.38. Landscape data collated by Oxford under this DEFRA project shows that in a typical arable situation, 54% of land is arable, of which 69% is in cereals, with an average field size of 8.24 ha. Assuming square cereal fields with Conservation Headlands along three sides of each field, the amount of insect-rich brood-rearing habitat would be 6.3% of cereal area, or 4.3% of arable area. Thus a 1% increase in such habitat on the arable area was equivalent to an increase in chick survival rate of around 0.04. By extrapolation, having 6% of arable area as insect-rich brood-rearing habitat would result in a chick survival rate of around 0.44, close to that recorded during the pre-pesticide era (Potts 1986). The current average chick survival rate of 29.4% observed in Sussex (Table 1) corresponds to 2.35% of the arable area being insect-rich.

The Oxford landscape data also indicated that the average amount of hedgerow in a typical arable situation was 4 km/km^2. Using this value as a measure of nesting cover, together with an average chick survival rate of 29.4% (Table 1), resulted in an equilibrium density of 5.45 pairs/km^2 according to the density-dependent model. This was taken as the standard reference level against which to compare equilibrium levels for all combinations of nesting cover (from 2 to 10 km/km^2) and percentage of insect-rich arable area (from 0% to 6%), so that the latter were expressed as percentage differences relative to the reference level (Figure 2). The same combinations of nesting cover and percentage of insect-rich arable area were used as input into the density-independent model for comparison (Figure 3). To improve comparability, the output, which was an annual rate of change, was transformed to a relative difference over five years, this being the approximate time for the density-dependent model to reach equilibrium.

When density dependence was included in the model, the contours indicated that when insect-rich brood-rearing habitat was scarce, increasing its proportional coverage had a similar positive effect on density as increasing the amount of nesting cover. When the insect-rich brood-rearing habitat reached around 3% of arable area, the positive response in density was strongest in relation to increasing nesting cover. The model predicted that a 1% increase in insect-rich brood-rearing habitat, from 2.35% (equivalent to the current average chick survival rate) to 3.35%, would yield a 13.8% increase in population equilibrium when nesting cover is kept at its average of 4 km/km^2. Conversely, keeping insect-rich habitat constant at 2.35% of arable area, a 1% increase in nesting cover (from 4 to 4.04 km/km^2) would produce a population increase of 1.1%.

When density dependence was omitted from the model, the pattern of contours was very different. Availability of nesting cover no longer played a role in determining density (all contours horizontal). Generally the amount of insect-rich brood-rearing habitat required to avoid a decline was higher than when density dependence was present.

2.6 Conclusions

1. Ignoring density dependence in the partridge model leaves unrecognised the role of nesting cover as an important determinant of Grey Partridge density in the agricultural environment.
2. To improve prospects for the recovery of the Grey Partridge, equal emphasis should be given to improving the amount of arable land containing insect-rich brood-rearing cover and nesting cover.
3. Considering that the average annual rate of population change over the last decade was 0.926, stabilising the population (BAP Objective 4.2, Anon. 1995) requires a relative increase of 8% (100/0.926-100). This can be achieved by having 3% of arable area as insect-rich brood-rearing habitat, or by increasing nesting cover from 4 to 4.3 km/km^2.
4. The average annual rate of population change over the last decade of 0.926 amounts to a change of 0.58 (42% decline) between 1995, when the BAP targets were set, and 2002. Recovery of the population to its 1995 level (BAP target 4.3, Anon. 1995) requires a relative increase of 72% (100/0.58-100). It depends on achieving 6.5 km/km^2 of nesting cover together with 3% of arable area being insect-rich.
5. Given that set-aside currently represents on average 11% of arable land (Oxford data, this DEFRA project), its judicious use could satisfy these requirements, e.g. by managing it as adjacent strips of tussocky grass for nesting and cereal mixtures for brood-rearing (total width 20 m) distributed over the farm. Such management is already permitted under existing prescriptions.

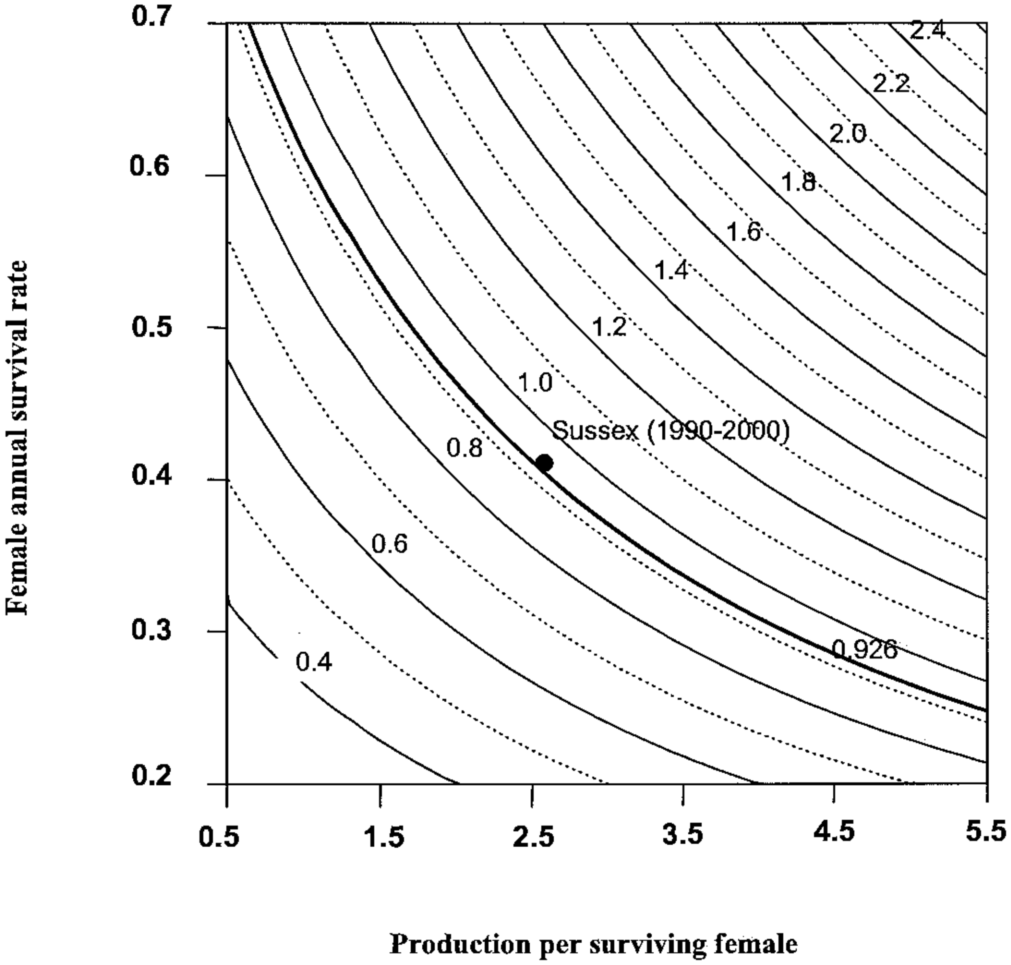

Figure 1 Contour map of the annual rate of population change for the Grey Partridge in relation to female annual production and survival, assuming no density dependence in either production or survival. The contour for the 1990-2000 annual rate observed in the BTO CBC data is superimposed (0.926), as is the point corresponding to the average parameters in Table 1.

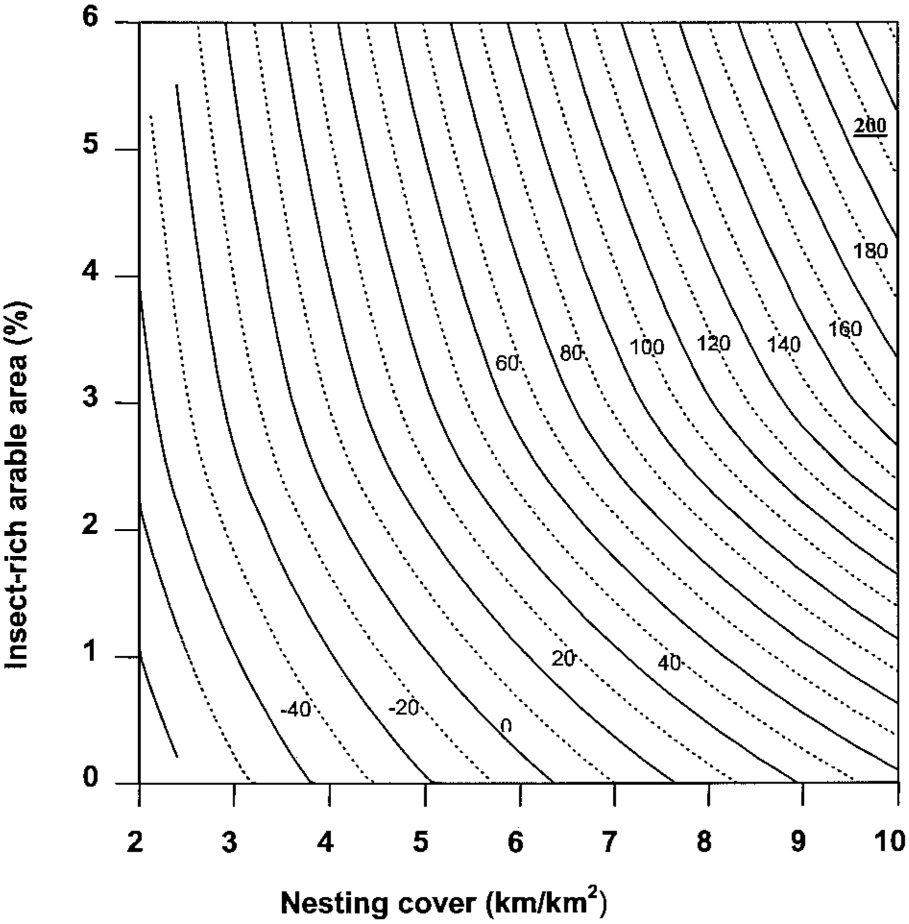

Figure 2 Density-dependent model. Contour map of relative differences (%) in equilibrium levels according to the amount of nesting cover and the percentage of arable area made up of insect-rich brood-rearing habitat. The baseline (contour 0) passes through the density predicted on an "average" arable farm over the last decade.

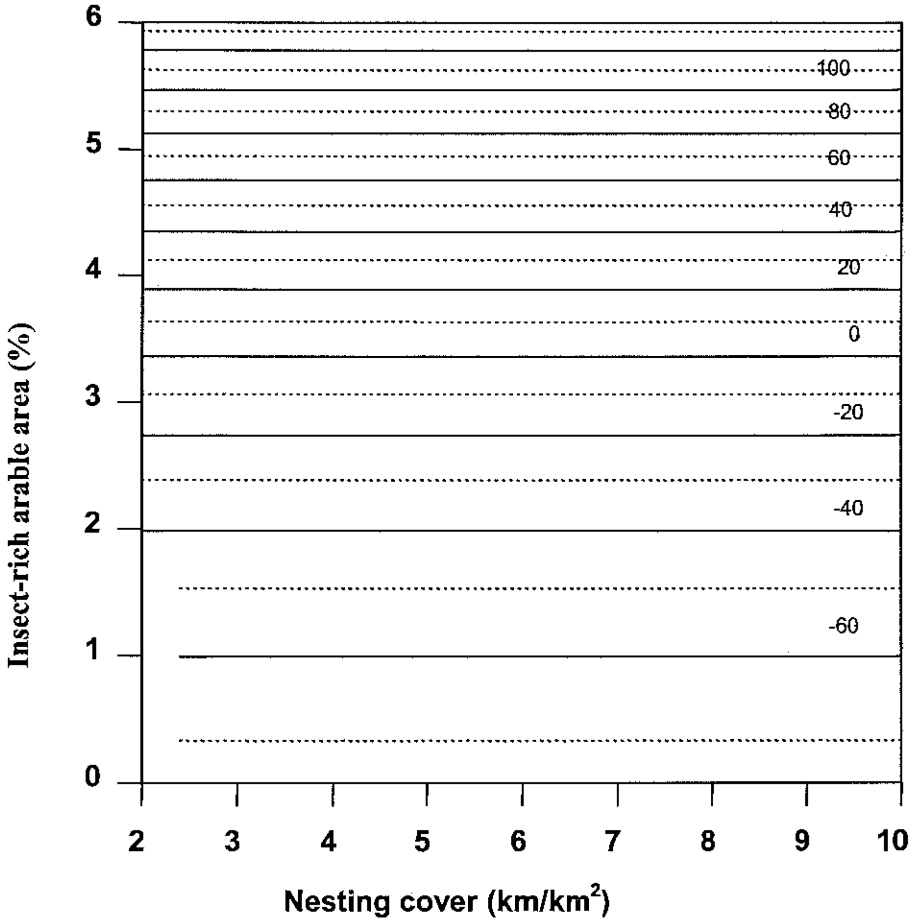

Figure 3 Density-independent model. Contour map of relative differences (%) after five years according to the amount of nesting cover and the percentage of arable area made up of insect-rich brood-rearing habitat.

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CHAPTER 3 USING HABITAT ASSOCIATION MODELS TO PREDICT IMPACTS OF AGRICULTURAL LANDSCAPE CHANGES ON FARMLAND BIRD POPULATIONS

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3.1 Introduction

Models that relate numbers of birds to the availability of certain habitats ("habitat association models") can be used to predict the effects of changes in habitat on breeding bird populations. These models, constructed using multivariate regression techniques, can relate any measure of bird abundance (for example, number of individuals, of singing males, of nests, or simply the presence or absence of birds) to predictor variables which explain a high proportion of variation in bird abundance.

Habitat association models have been used to show, for example, that breeding Cirl Buntings *Emberiza cirlus* population densities are positively associated with winter stubble and rough field margin availability, the subsequent provision of which has enhanced Cirl Bunting populations (Peach *et al.* 2001). Thus far, habitat association models have been used to make *qualitative* predictions about the direction of response in population size to management change. Here we aim to use existing habitat association models to *quantify* the extent of population recovery following specific management changes. Particular emphasis has been placed on predicting the impacts of changes in the agricultural landscape that may occur as the result of current agricultural trends, or that may be achieved through the mechanism of options in current or future agri-environment schemes.

These models have the advantage that they can relate abundance directly to habitat parameters that are open to manipulation by management change. Their disadvantage is that they do not make an explicit link between habitat change and the impacts on demographic rates that result in the observed population change. There is only one species for which this link has been made, the Grey Partridge, which is subject to a separate part of this report by the Game Conservancy Trust. This study involves using habitat-association models developed in one agricultural landscape to make predictions in other, different, landscapes. The results of such extrapolation must be treated with caution as they depend on the assumption that factors controlling bird distribution remain constant in different landscapes. However, they may be prone to lacking geographical generality as mentioned above (the degree of generality of such intensive models is currently the subject of a joint RSPB/Oxford University/BTO study, funded by BBSRC).

Due to variation in the quality of data available for each of the 20 species in the farmland bird index there is considerable variation in the style of models used in this study. Some, such as those for Lapwing *Vanellus vanellus*, Skylark *Alauda arvensis* and Yellowhammer *Emberiza citrinella*, are based on intense autecological surveys conducted in a relatively small geographical area. These models have a high precision of measurement of the habitat variables that predict population size, and these variables may be relatively "fine-scale", such as margin width and hedge height, making them suitable output for conservation recommendations.

The model used for Rook was developed using multivariate regression, but using a far larger spatial unit for measurement of numbers, so instead of using fine-scale habitat variables to predict abundance in a single field or boundary it uses agricultural statistics. Finally, although no "classic" multivariate habitat-association model was available for Yellow Wagtail, habitat-specific density estimates were used to generate estimates of mean density over a mosaic of different land-use types.

Of the 20 species in the farmland bird index, it was only possible to use habitat-association models for seven in this report. Weak predictive models were available for several more species such as Starling, Greenfinch and Linnet, but were not considered to predict bird distribution accurately enough that faith could be placed in any predictions obtained from their manipulation. The availability, suitability and sources of models for each of the 20 farmland bird species are outlined below.

3.2 Species Models

The farmland bird index is based on 20 species. However, for many of these species habitat-association models are unavailable, either because such research has not been conducted or because attempts to produce models have been unsuccessful. The species, and models that were utilised in each case, are described below.

Kestrel *Falco tinnunculus*

No model is available for this species. Village (1989) gives density estimates for Kestrel in different farming systems (mixed, pastoral, arable) in Scotland, and other authors do likewise for other habitats and/or regions, but these were not considered accurate or reliable enough to use in a predictive manner.

Grey Partridge *Perdix perdix*

Comprehensive habitat-association modelling of Grey Partridge has been conducted by the Game Conservancy Trust. This is presented elsewhere in this report.

Lapwing *Vanellus vanellus*

A Lapwing model was developed for the purpose of this report using data collected by a recent PhD study in the West Midlands (R. Sheldon, unpublished data). Data on habitat variables were collected in 2000 from 61 fields where lapwings attempted to breed, as well as 61 randomly-selected neighbouring fields where breeding was not attempted. Attempts were made to develop models with either number of nesting attempts or number of fledged chicks in each habitat unit (field) as the response variable, using a Poisson error distribution and a log link (Crawley 1993). Although statistically significant models were obtained, using predictor variables such as field area, crop type, presence/absence of woodland bordering the field and height of the field boundary, these models gave a disappointingly poor fit to the actual data. Goodness-of-fit of these minimum adequate models (MAM's) was tested by examining the ratio of residual deviance to residual degrees of freedom (as the use of a Poisson error distribution prohibits the calculation of a meaningful statistic for the proportion of variance explained by the model). Models using both response variables were over-dispersed (had deviance/degrees of freedom values over two, Crawley 1993), indicating that they provided a poor-fit to the data due to either a failure of the assumption of Poisson errors or the failure to include important predictor variables.

To simplify the data a binary logistic model was developed using presence/absence of nesting attempts in each field (presence = 1, absence = 0) as the response variable, a binomial error distribution and a logit link. Quadratic terms were included to test for nonlinear effects of continuous variables, and intercorrelation between predictor variables tested for; variables that were found to be highly intercorrelated were tested in the model separately and the most significant retained.

A backward selection procedure was used, with the significance of each predictor variable being tested by the change of deviance of the model (ΔD) on the removal of that variable. At each iteration, the variable whose removal caused the smallest change in ΔD was removed. This was repeated until all remaining predictor variables were significant at $P \leq 0.05$. The minimum adequate model (MAM) is given below:

$\mathrm{Log}_e(p/1\text{-}p) = \text{crop} + (\text{field area} \times 0.1232) - (\text{wood} \times 1.7547) - 1.1653$

Where p = the probability of Lapwings attempting to breed in any given field, wood = presence/absence of woodland adjoining any boundary of the field (presence = 1, absence = 0). Field area was measured in hectares. The intercept value, crop, varies between the seven categories of crop type: winter cereals = -0.7728, spring cereals = -0.3143, other crops = 0.0363, set-aside = 1.9134, option 1b in the pilot Arable Stewardship Scheme = 8.2619, stubbles = 1.2711, grassland = -1.1563.

The goodness of fit of this model was tested using a classification table, classifying the probabilities of occurrence predicted for each field into a binary prediction of either present or absent. At the least stringent level of classification, with $p \geq 0.5$ being "present" and $p < 0.5$ "absent", this model predicted presence or absence correctly in 73.8% of fields (Table 1a), while at a higher level of stringency (Table 1b) 89.8% of predictions were correct (50% = no better than random).

	Predicted present	**Predicted absent**	**% predicted correctly**
Observed present	46	17	73.0
Observed absent	16	47	74.6

Table 1a Classification table for Lapwing binary logistic model with least stringent cut-off ($p < 0.5$ = absent, $p \geq 0.5$ = present).

	Predicted present	**Predicted absent**	**% predicted correctly**
Observed present	23	2	92.0%
Observed absent	4	30	88.2%

Table 1b Classification table for Lapwing binary logistic model with more stringent cut-off ($p \leq 0.2$ = absent, $p \geq 0.8$ = present, $0.2<p<0.8$ = not classified).

So, while is regrettable that a satisfactory model to predict numbers of Lapwings could not be developed, the binary logistic model is reasonably accurate at predicting in which fields Lapwings attempt to breed in the West Midlands. As this model was developed in a mixed landscape it is to be hoped that altering the baseline to resemble a pastoral or arable landscape will not be extrapolating the model beyond the point at which it can be expected to provide acceptably accurate results.

Predictions were made on a baseline constructed from the West Midlands data, with changes in crop type and field size made to fit each of the three baseline landscapes. Therefore predictions took the form of number of fields (out of 126) where Lapwings attempted to breed, with the cut-off set at 0.5 ($p < 0.5$ = absent, $p \geq 0.5$ = present).

Stock Dove *Columba oenas*

No model was available for this species.

Woodpigeon *Columba palumbas*

Woodpigeon was one of the many species whose abundance in field boundaries was modelled over a number of summers and winters in two areas of East Anglia by Parish, Lakhani and Sparks (1992, 1994, 1995). The model produced to describe summer distribution in their Huntingdon study area showed a high R^2 value (0.76) indicating a good fit of model to data. Models from the Huntingdon study area were considered by the authors to better explain the relationship between the bird variables and explanatory transect attributes than those developed in the Swavesey study area, as study sites in the Huntingdon region included a wider range of hedgerow types and sizes.

The most important explanatory variable was transect location, defined as being in pasture, small arable fields or large arable fields. The analysis produced separate models for each farmland type:

- Pasture: "Woodpigeons" = (0.0133 × THTN) + (0.01199 × HLHH) - (1.994 × CW) - 0.179
- Small arable: "Woodpigeons" = (0.06551 × THTN) - (0.00024 × HLHH) + (0.671 × CW) - 0.51
- Large arable: "Woodpigeons" = (0.00646 × THTN) + (0.00559 × HLHH) - (0.392 × CW) + 0.095

Where "Woodpigeons" = mean number of birds recorded by five summer visits to a 200 m transect along field boundary, THTN = average tree height × number of trees, HLHH = hedge length × mean hedge height and CW = mean hedge crown width. All distance measurements were in metres.

In the following manipulations one of these three models has been applied to each baseline landscape: pasture to the pastoral landscape; small arable to mixed; large arable to arable. Predictions are of the number of individual Woodpigeons present in a 200 m field boundary transect.

Turtle Dove *Streptopelia turtur*

No habitat-association model is available for Turtle Dove, and although studies have been made on the use of various habitats by Turtle Doves in the breeding season (Browne & Aebischer 2001) no published study has produced habitat-specific estimates of density.

Skylark *Alauda arvensis*

Wilson *et al.* (1997, 2000) developed a model predicting Skylark territories within single fields, using data collected from nine Oxfordshire farms over three summers. Models were developed for each month through the breeding season, as Skylark distribution tends to change through the spring and summer as some crop types grow and become less suitable for breeding. For this study the model for May was used, as this is approximately the mid-point of the breeding season. Farming system (conventional or organic) was an important predictor in this model, and determined not only the value of the crop type variable (i.e. different values were used for the same crop type in different farming systems) but also the value of the area and crop height predictors.

Parameter Variable	**Farming system**	**Estimate**
Factors:		
Crop 1 (winter cereal)	Organic	13.4
Crop 2 (spring cereal)	Organic	14.15
Crop 3 (oil-seed rape)	Organic	-
Crop 4 (field beans)	Organic	14.07
Crop 5 (maize)	Organic	13.27
Crop 6 (set-aside)	Organic	13.9
Crop 7 (silage)	Organic	13.39
Crop 8 (grazed pasture)	Organic	11.8
Crop 1 (winter cereal)	Conventional	11.4
Crop 2 (spring cereal)	Conventional	12.15
Crop 3 (oil-seed rape)	Conventional	11.23
Crop 4 (field beans)	Conventional	12.07
Crop 5 (maize)	Conventional	11.27
Crop 6 (set-aside)	Conventional	11.9
Crop 7 (silage)	Conventional	11.39
Crop 8 (grazed pasture)	Conventional	9.8
Covariates:		
Field area	Organic	0.2428
Field area	Conventional	0.3381
Field area^2	All	-0.007118
Boundary index	All	-0.8710
Crop height	Organic	0.01460
Crop height	Conventional	0.02783
Crop height^2	All	-0.0003182
Slope	All	0.09543
Slope^2	All	-0.007116
Field shape	All	-18.95
Field shape^2	All	6.956

Table 2 Parameter estimates for model predicting Skylark densities in May.

For example, the model to predict the number of territories in a field of conventionally farmed winter cereal:

Log_e(number of Skylark territories) = 11.44 + (0.3381 × field area) - (0.007118 × field area^2) - (0.871 × boundary index) + (0.02783 × crop height) - (0.003182 × crop height^2) + (0.09543 × slope) – (0.007116 × slope^2) - (18.95 × field shape) + (6.956 × field shape^2)

Details of methods for measurement of habitat variables are given in Wilson *et al.* (1997). Modelling was based on a baseline landscape derived from the 300 fields used in the original analysis. Percentage changes in crop type were based on the area of land covered, not the number of fields. The variables for field shape and slope were fixed as for the Oxfordshire study landscape for each of the three baselines, while crop height was linked to crop type so that manipulations of crop type resulted in an appropriate change in crop height. As the model contained non-linear predictor variables (quadratics of area, crop height and slope) the predictions made could vary, depending on which individual fields were altered. Therefore for each manipulation five replications were made, each time randomly choosing fields until the desired % (±0.1%) of area had been manipulated.

The most significant effect of the non-linear relationships is of that with field area. Larger fields are predicted to contain disproportionately more Skylark territories – a few large fields will contain more Skylarks than a greater number of smaller fields of the same total area. This is illustrated by Figure 1, which shows the impact of a beneficial landscape change (moving cereal sowing season from autumn to spring). As can be seen, if only large fields (above average size) are altered, the increase in Skylarks will be 6.7% greater than if fields are selected at random. As stated above, random selection of fields was used in model manipulations, so it should be borne in mind when reading the results below that the predicted increases in the number of Skylark territories could be 6.7% greater if larger fields were preferentially targeted.

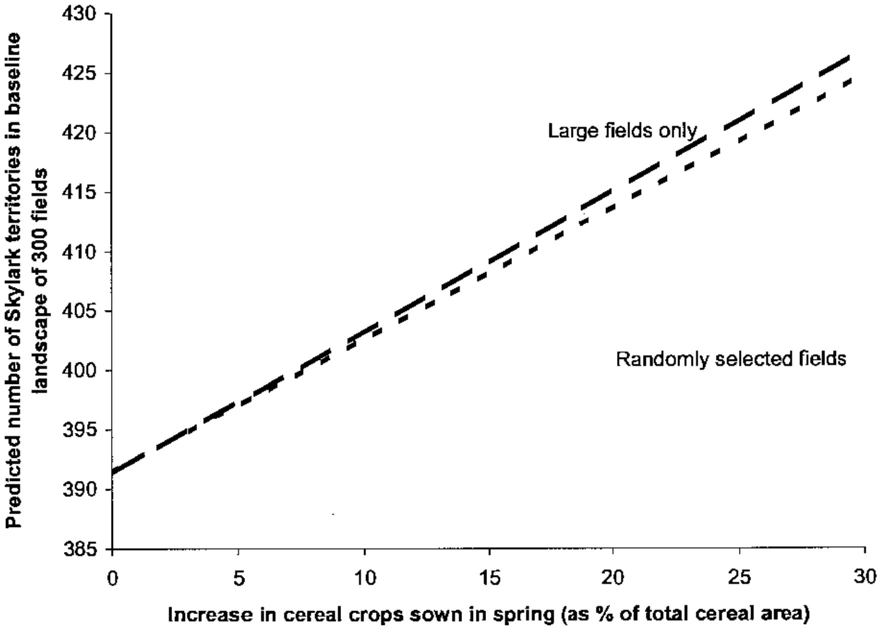

Figure 1 Increase in predicted number of Skylark territories if sowing season changed, demonstrating impact of selecting larger fields for manipulation.

Because this Skylark model was developed in a mixed farmland landscape, it is felt that extrapolating from the model to make predictions for pastoral and arable landscapes is still within the bounds at which the model should be expected to function accurately.

Yellow Wagtail *Motacilla flava*

As this species has been comparatively-poorly studied in this country there is no available habitat-association model for Yellow Wagtails. However, Yellow Wagtail was one of the species surveyed in agricultural habitats in Essex by Mason & McDonald (2000), work which resulted in estimates of Yellow Wagtail densities in 17 habitat categories. These density estimates allow estimates of total Yellow Wagtail density to be made for the three baseline landscapes used in this study, and for predictions of how changes in cropping will affect numbers. Caveats must be stated regarding the generality of this model, as density estimates were measured in a region that has relatively high densities of Yellow Wagtail (Gibbons *et al.* 1993). This approach would not produce accurate estimates for overall densities in regions such as South-west England where Yellow Wagtail numbers are much lower.

Predictions were made for the number of Yellow Wagtail territories in 1 km^2 square with the proportion of crop types in that square matching baseline estimates and being manipulated as desired. Although predictions were made for landscapes not representative of the Essex region were data was collected (the pastoral landscape, in particular) the predictions for these landscapes should be viewed with caution.

Whitethroat *Sylvia communis*

Binary logistic regression models predicting Whitethroat presence/absence along 100 m lengths of hedgerow have been developed in two regions, Leicestershire (Stoate & Szczur 2001) and Wiltshire (Stoate, unpubl.). The two models are strikingly similar, both predicting presence/absence as a function of hedgerow height and uncropped herbaceous margin width. To test the generality of these two models, each was used to predict Whitethroat occurrence in the other region. The model developed in Wiltshire predicted 83.9% of presence/absences in the Leicestershire landscape correctly, while the Leicestershire model predicted 84.4% correctly.

Therefore the two models were amalgamated, producing a mean model for the purpose of this report:

$$\text{Log}_e(p/1\text{-}p) = (\text{herb width} \times 11.31) - (\text{hedge height} \times 1.609) - 2.97$$

Where p = the probability of Whitethroats being present in any given 100 m length of hedgerow, herb width = mean width of uncropped herbaceous margins on both sides of hedge combined (m) and hedge height = mean height of hedge measured at five points along 100 m length (m).

Predictions were based on the original modelling landscape of 73 hedgerow lengths, giving a prediction of the number of occupied sections at a cut-off level of 0.5 ($p < 0.5$ = absent, $p \geq 0.5$ = present). As similar models were produced in two distinct regions, one of mixed farmland and the other arable, it is felt that the final model has excellent geographical generality and the predictions should be for all three baseline landscapes.

Rook ***Corvus frugilegus***

Griffin (1999) used linear multivariate regression analysis to develop a habitat-association model that described the distribution of Rooks in County Durham. This model predicted the number of breeding pairs of Rook in 5 × 5 km squares (25 km^2) using DEFRA agricultural statistics, and is given below:

$$\text{Rooks} = ((0.236 \times a) + (3.595 \times b) + (0.02373 \times c) + 10.473)^2$$

Where a = square root of area (Ha) of grassland over five years old, b = $\log_{10}$ of the area (Ha) of rough grassland and c = area of grassland under five years old.

The R^2 of this model was 0.41, indicating that it described 41% of the variation in Rook numbers. Hence while this model describes a significant proportion of variation in Rook numbers, a larger proportion must be determined by habitat variables that are not included, and which very probably cannot be measured at such a large spatial scale. Predictions were converted to nesting $pairs.km^{-2}$. The predictions for Rook density given by the models for the three baseline landscapes fall within the ranges given by Gibbons *et al.* (1993). Therefore it is felt that this model should adequately predict density for all three landscapes.

Starling ***Sturnus vulgaris***

Parish, Lakhani & Sparks (1995) (see entry for Woodpigeon above) found a significant relationship between the abundance of Starlings in field boundaries and boundary characteristics (hedge height, length and crown width) in their Huntingdon study area (R^2 = 0.67). The relationships varied depending on the type of farmland these boundaries were upon, and can be described by the following three equations:

- Pasture: "Starling" = (0.00982 × HLHH) - (1.627 × CW) + 0.179
- Small arable: "Starling" = (-0.00158 × HLHH) + (0.280 × CW) + 0.353
- Large arable: "Starling" = (0.00076 × HLHH) - (0.187 × CW) + 0.181

Where "Starling" = mean number of birds recorded by five summer visits to a 200 m transect along field boundary, HLHH = hedge length × mean hedge height and CW = mean hedge crown width. All distance measurements were in metres.

However closely these models describe the abundance of Starlings in hedgerows, the response variable – mean count – may not be appropriate for measuring the importance of a measurement unit for breeding Starlings. The presence of Starlings in hedgerows may not be reflective of the importance of these boundaries for breeding. Additionally, the availability of nest sites in buildings and the presence of sufficient suitable pasture, the preferred foraging habitat, may be more important determinants of population size. For these reasons the models given above were not used to predict the effect of habitat manipulations on Starling numbers.

Tree Sparrow ***Passer montanus***

No habitat-association model is available for Tree Sparrow.

Linnet *Carduelis cannabina*

A model predicting the probability of occurrence of one or more Linnet nests in a field boundary section has previously been developed from field data collected in Oxfordshire (Wilson *et al.* 2000). This model predicted Linnet presence as a function of boundary type and length, margin width and whether the boundary was adjacent to a road/farm track:

$\text{Log}_e(p/1\text{-}p) = \text{farm} + \text{boundary type} + (0.003484 \times \text{boundary length}) + (0.9637 \times \text{road} + (1.666 \times \text{margin width})$

However, this model performed poorly when tested using classification tables. Although the model predicted absence of Linnet nests well (95.2% of cases where no nests were present, using a 0.5 "cut-off") it predicted presence of Linnet nests poorly (only 22.9% of presences were predicted correctly). This poor performance may be due to the traditional occupation of nesting locations by Linnets, meaning that birds may remain breeding even when local habitat conditions have become less suitable. In addition, foraging Linnets roam up to 3 km from the nest, so landscape variables may have been instrumental in determining distribution on a much larger scale than considered by the Oxfordshire Linnet study. This model was not used in the following analyses.

Goldfinch *Cardeulis carduelis*

No habitat-association model is available for Goldfinch. Although this species was one of those modelled in relation to boundary characteristics by Parish, Lakhani & Sparks (1994) their models had a very poor goodness-of-fit (R^2 values of 0.13, 0.53 and 0.19 for the three models describing summer distribution) and so have not been used in this study.

Greenfinch *Carduelis chloris*

No habitat-association model is available for Greenfinch. As for Goldfinch, the models produced by Parish, Lakhani & Sparks (1994) failed to explain a significant proportion of variation in Greenfinch abundance.

Reed Bunting *Emberiza schoeniclus*

No habitat-association model is available for Reed Bunting. None of the models produced by Parish, Lakhani & Sparks (1994) succeeded in explaining a significant proportion of variation in Reed Bunting abundance.

Yellowhammer *Emberiza citrinella*

Bradbury *et al.* (2000) used multiple log-linear regression (Crawley 1993) to produce a model predicting the number of Yellowhammer territories on field boundary sections as a function of boundary characteristics and the crop types neighbouring the boundary. This model was slightly altered for use in this study and is given below.

$\text{Log}_e(\text{Yellowhammer territories}) = (0.0882 \times \text{margin width}) + (0.2805 \times \text{hedgerow}) + (0.3469 \times \text{ditch}) + (0.5064 \times \ln(\text{boundary length})) - (0.3864 \times \text{grass-only}) - 3.022$

Where margin width = maximum width of uncropped margin on one side of the boundary (m). Hedgerow, ditch and grass-only are binary presence/absence variables (absence = 0, presence = 1), with grass-only being the presence of grassland (permanent pasture or temporary ley) on both sides of a boundary.

Predictions were made of the number of Yellowhammer territories present on the 387 field boundary sections that the original modelling dataset consisted of, with variables altered to fit the three baseline landscapes (see below). However, these boundaries were contiguous, so individual Yellowhammer territories were often recorded on multiple boundaries. To correct for this over-prediction the relationship between the total number of boundary territory counts was plotted against the known number of territories (calculated by mapping Yellowhammer sightings) for each study site in each summer of recording. There was a significant linear relationship between the number of boundary territory counts and number of territories, with the slope of this line (2.67) being used to "correct" boundary territory counts – for every 2.67 boundary counts there was deemed to be one actual territory.

As the model for Yellowhammer was developed in mixed farmland it was felt it was not unreasonable to extrapolate predictions for both pastoral and arable landscapes.

Corn Bunting *Miliaria calandra*

Although a considerable amount of research has been concentrated on Corn Buntings, none of this has produced a habitat-association model suitable for use in this study. Some studies have studied the use of different agricultural habitats by Corn Buntings by correlating bird density to the abundance of habitat types (Aebischer & Ward 1997). Although such data are important indicators of habitat use they are not suitable for manipulation in the same manner as for (for example) Yellow Wagtail (above). Habitat-specific density estimates are available from two studies. However, density estimates for Corn Buntings in Sussex were felt to be unsuitable due to the unusual nature of the study area, in that the area was unusually good for Corn Buntings, sensitively managed and therefore unrepresentative of the agricultural landscape as a whole (N. Brickle, pers. comm.). The estimates produced by Aebischer & Ward (1997) were more generally applicable as they were derived from a larger (Sussex) study area, but unfortunately only calculated for three very general habitat types (mixed rotational, intensive arable and non-arable). This would only have allowed for the most basic of manipulations.

3.3 Baselines for Modelling

In order to calculate the predicted changes in bird abundance caused by changes in habitat parameters it was first necessary to construct "baseline" landscapes, representative of different agricultural landscapes. Three landscapes were developed to represent arable, mixed and pastoral farmland.

A number of different data sources were used to generate these baselines. In some cases it was possible to obtain at least some of the required habitat variables from the data used to generate the models themselves. However, this data had to be treated with some caution, as the regions used for data collection were sometimes selected to represent a wide cross-section of farmland habitat types rather than an accurate census of the frequency of occurrence of these habitats.

Variable	Model(s)	Source
Crop type	Lapwing, Skylark, Yellow Wagtail	DEFRA June Agricultural statistics 2000
Field size	Lapwing, Skylark	Donald[1]
Field shape	Skylark	Wilson[2]
Boundary index	Skylark	Wilson[2]
Grass-only	Yellowhammer	DEFRA June Agricultural statistics[3]
Ditch presence	Yellowhammer	Bradbury[4]
Hedge presence	Woodpigeon, Whitethroat, Yellowhammer	CS1990[5]
Margin width	Whitethroat, Yellowhammer	Bradbury[4]
Trees – abundance in field boundaries	Woodpigeon	Proffitt[6]
Mean tree height (field boundaries)	Woodpigeon	Proffitt[6]

Table 3 Sources of baseline data.

[1] = Unpublished data from research conducted on Skylarks by P. Donald. Data from Dorset study site used for pastoral landscape, from Oxfordshire for mixed landscape and from East Anglia for arable landscape.
[2] = Unpublished data collected from nine mixed farms in Oxfordshire and used to construct models predicting Skylark abundance in Wilson *et al.* (1997) and the Skylark model used in this report.
[3] = The probability of a boundary having grassland on both sides was calculated using the formula (grassland as a proportion of total agricultural land)^2. This method makes the assumption that grassland is randomly distributed amongst arable land, which is not the case.
[4] = Data collected by Bradbury *et al.* (2000)
[5] = Data from the Countryside Survey 1990 (CEH).
[6] = Unpublished data collected by F. Proffitt in Oxfordshire and Warwickshire.

For a number of habitat variables it was not possible to obtain estimates for baselines other than that from original modelling data. In all cases – field shape, field slope, boundary index – data was collected from mixed farmland. As this landscape is intermediate between the other two categories it was decided that retaining the same estimates for these variables for all three landscapes was unlikely to lead to large errors. In all cases these variables were not manipulated in the models.

3.4 Habitat Manipulations

The models detailed in the previous section can provide predictions of the impact of manipulating any of the parameters that they contain. Therefore it is possible to investigate, for example, the impact on Skylark numbers of varying the shape of fields within an agricultural region, or how Lapwings would respond to a decrease in the presence of woodlots on farmland. However, this report has concentrated on modelling habitat changes that are likely to be achievable through agri-environment schemes, and within "sensible" margins; it is conceivable, for example, that 5, 10 or 15% of all cereal crops could be spring-sown rather than autumn-sown, but not that all cereal crops could be spring-sown.

Table 4 below shows which manipulations are tested in this report, and for which species.

Habitat parameter manipulated	**Lapwing**	**Woodpigeon**	**Skylark**	**Yellow Wagtail**	**Whitethroat**	**Rook**	**Yellowhammer**
Cereal sowing season (autumn → spring)	✓		✓	✓			
% of arable land as set-aside	✓		✓	✓			
% of arable land as rape	✓		✓	✓			
% of all farmland as organic			✓				
Ratio of arable:pastoral	✓		✓	✓		✓	✓
% of field boundaries with hedgerow		✓	✓		✓		✓
% of field boundaries with ditches							✓
% of field boundaries with uncropped margins					✓		✓

Table 4 Habitat manipulations modelled.

For some models, predictions were generated for one unit – for example, one field boundary length for Woodpigeons and one 5 km square for Rooks. However, for a number of models the datasets used to develop the models were available, allowing landscapes of numerous units to be developed. For example, the model for Yellowhammers was used to produce predictions for each of 387 boundaries in a landscape based upon the Oxfordshire farmland where data was collected, with the appropriate variation in variables to approximate the three baseline landscapes but other variables (i.e. ditch presence, boundary length) staying the same. With these models there was a random chance element involved in the manipulations, as slightly different predictions could be generated depending on exactly which units (boundary sections or fields) were selected for manipulation. Therefore at each level of manipulation five replicates were calculated, with a different random selection of units being changed each time, and a mean of these predictions calculated.

Manipulation 1: Changing sowing season of cereal crops

Lapwing

	Pastoral		Mixed		Arable	
% of all cereals sown in spring	**Birds**^{2}	**% change**	**Birds**^{2}	**% change**	**Birds**^{2}	**% change**
Baseline[1]	46.03	-	54.46	-	61.33	-
+5%	46.13	+0.25	54.71	+0.25	61.63	+0.50
+10%	46.23	+0.50	54.97	+0.51	61.94	+0.99
+15%	46.33	+0.76	55.23	+0.76	62.24	+1.51
+20%	46.42	+1.01	55.48	+1.02	62.54	+1.97

Table 5 Changes in the predicted numbers of fields occupied by Lapwings in response to increases in the proportion of cereal crops sown in spring.

[1] = based on barley data from DEFRA June agricultural statistics and an estimate of 2% of all wheat as spring-sown. Baseline values are pastoral = 14.86%, mixed = 10.41% and arable = 11.11% spring-sown.

[2] = number of fields in which breeding attempts were made, based on a baseline landscape of 126 fields.

It should be noted that manipulations above are presented as percentages of the total area of cereals, not of the total agricultural area. One percent of the total area of cereals equates to 0.188% of the total agricultural area in the pastoral baseline landscape, 0.352% in the mixed landscape and 0.503% in the arable landscape. Predicted increases in Lapwings are generally low for a change in sowing date, as the intercept value for spring-sown cereals was only marginally higher than that for autumn-sown cereals.

The geographical generality of the lapwing model is likely to be very poor, as the 1998 national Lapwing survey showed very significant differences in the occupation of different habitat types in different regions (Wilson *et al.* 2001). There has been a marked range contraction from some southern and western parts of the range, despite the remaining presence of apparently suitable habitat – the model developed in the West Midlands would very likely massively overpredict the presence of lapwings in these regions.

Skylark

% of all cereals sown in spring	Pastoral		Mixed		Arable	
	Birds[2]	% change	Birds[2]	% change	Birds[2]	% change
Baseline[1]	246.42	-	393.33	-	468.13	-
+5%	247.42	+0.41	398.41	+1.29	480.20	+2.58
+10%	248.43	+0.82	403.48	+2.58	492.28	+5.12
+15%	249.44	+1.22	408.55	+3.87	504.35	+7.74
+20%	250.46	+1.65	413.63	+5.16	516.43	+10.32

Table 6 Changes in the predicted numbers of Skylark territories in response to increases in the proportion of cereal crops sown in spring.

[1] = as for Lapwing in Table 5.
[2] = number of territories predicted in baseline landscape consisting of 300 fields.

The conversion of autumn-sown cereals to spring-sown is predicted to result in a considerable increase in Skylark territories, particularly in the arable landscape where there is proportionately far more cereal cropping.

Yellow Wagtail

% of all cereals sown in spring	Pastoral		Mixed		Arable	
	Birds[2]	% change	Birds[2]	% change	Birds[2]	% change
Baseline[1]	0.4726	-	0.6799	-	0.8632	-
+5%	0.4734	+0.22	0.6816	+0.35	0.8654	+0.39
+10%	0.4745	+0.44	0.6842	+0.70	0.8692	+0.78
+15%	0.4757	+0.67	0.6869	+1.05	0.8729	+1.17
+20%	0.4769	+0.89	0.6895	+1.41	0.8767	+1.56

Table 7 Changes in the predicted density of Yellow Wagtails in response to increases in the proportion of cereal crops sown in spring.

[1] = as for Lapwing in Table 5.
[2] = birds per km^2.

Changes in the season of cereal sowing has little effect on densities of Yellow Wagtail, as densities in cereal crops are low regardless of sowing season.

Manipulation 2: Increasing the proportion of arable land as set-aside

Lapwing

	Pastoral		Mixed		Arable	
% of arable as set-aside	Birds^2	% change	Birds^2	% change	Birds^2	% change
Baseline[1]	46.03	-	54.46	-	61.33	-
+10%	47.08	+2.28	57.64	+5.83	65.64	+7.03
+20%	48.13	+4.56	60.32	+11.68	69.96	+14.07
+30%	49.18	+6.84	63.99	+17.52	74.27	+21.10

Table 8 Changes in the predicted numbers of fields occupied by Lapwings in response to increases in the proportion of arable land converted to set-aside.

[1] = baseline values are pastoral = 10.215%, mixed = 10.686% and arable = 12.177% of all arable land as set-aside.
[2] = based on a baseline landscape of 126 fields.

As with manipulation 1, the area of land available for conversion varies between the three baseline landscapes. One percent of arable land equates to 0.338% of the total agricultural area in the pastoral baseline landscape, 0.616% in the mixed landscape and 0.876% in the arable landscape.

The lapwing habitat association model predicts very considerable increases in the percentage of fields used by breeding Lapwings as a result of the increase in fields put into the set-aside scheme. Again, this is particularly noticeable in the arable baseline landscape, where the area of land available for conversion to set-aside is considerably greater.

Skylark

	Pastoral		Mixed		Arable	
% of arable as set-aside	Birds^2	% change	Birds^2	% change	Birds^2	% change
Baseline[1]	246.42	-	393.33	-	468.13	-
+10%	251.95	+2.24	402.09	+2.23	487.17	+4.07
+20%	257.49	+4.49	410.86	+4.46	506.22	+8.14
+30%	263.02	+6.74	419.62	+6.68	525.26	+12.20

Table 9 Changes in the predicted numbers of Skylark territories in response to increases in the proportion of arable land converted to set-aside.

[1] = as for Lapwing in Table 8.
[2] = number of territories predicted in baseline landscape consisting of 300 fields.

The Skylark model suggests that set-aside holds breeding Skylarks at higher densities than cereals and other crops, and hence predicts that conversion to set-aside will result in increased population levels.

Yellow Wagtail

% of arable as set-aside	Pastoral		Mixed		Arable	
	Birds[2]	**% change**	**Birds**[2]	**% change**	**Birds**[2]	**% change**
Baseline[1]	0.4726	-	0.6799	-	0.8632	-
+10%	0.4728	+0.027	0.6735	-0.0094	0.8513	-1.38
+20%	0.4730	+0.055	0.6670	-0.0188	0.8392	-2.76
+30%	0.4731	+0.082	0.6605	-0.028	0.8274	-4.14

Table 10 Changes in the predicted density of Yellow Wagtails in response to increases in the proportion of arable land converted to set-aside.

[1] = as for Lapwing in Table 8.
[2] = birds per km^2.

Interestingly, although Yellow Wagtail density is predicted to show a slight change only in response to an increase in set-aside, the direction of response is different in the pastoral landscape (where conversion of arable land to set-aside would be beneficial to Yellow Wagtail) from the response in mixed and arable land (where increased set-aside would cause a slight decline in overall Wagtail density). This can be attributed to the mosaic of habitats being converted to set-aside being generally poor for Yellow Wagtails in pastoral landscapes while in the other two baselines (arable landscapes in particular) arable land contains greater proportions of the potato, bean and pea crops that hold higher Wagtail densities.

Manipulation 3: Changes in the proportion of arable land growing oil-seed rape

Skylark

	Pastoral		Mixed		Arable	
% of arable as oil-seed rape	**Birds^2**	**% change**	**Birds^2**	**% change**	**Birds^2**	**% change**
-10%	252.04	+2.28	399.70	+1.62	497.29	+6.23
-5%	249.23	+1.14	396.52	+0.81	482.71	+3.12
Baseline[1]	246.42	-	393.33	-	468.13	-
+5%	243.61	-1.14	390.14	-0.81	453.55	-3.12
+10%	240.80	-2.28	386.96	-1.62	438.97	-6.23

Table 11 Changes in the predicted numbers of Skylark territories in response to changes in the proportion of arable land growing oil-seed rape.

[1] = baseline values are pastoral = 4.98%, mixed = 7.68% and arable = 5.68% of total arable land.
[2] = number of territories predicted in baseline landscape consisting of 300 fields.

Skylarks show a modest increase in density in response to conversion of oilseed rape area to other arable crop types, and of course a corresponding decrease in response to an increase in rape cropping. This predicted change is considerably lower than for other cropping changes, such as increasing the proportion of spring-sown cereal (above).

Manipulation 4: Increasing the proportion of all farmland managed organically

Only one habitat-association model included a predictor variable related to farming system, that for Skylarks developed in Oxfordshire.

	Pastoral		Mixed		Arable	
% of organic	**Birds[2]**	**% change**	**Birds[2]**	**% change**	**Birds[2]**	**% change**
Baseline[1]	246.42	-	393.33	-	468.13	-
+10%	261.73	+6.21	409.98	+4.23	479.98	+2.53
+20%	277.04	+12.42	426.64	+8.46	491.83	+5.06
+30%	292.36	+18.64	443.29	+12.70	503.69	+7.59

Table 12 Changes in the predicted number of Skylark territories in response to increased proportions of farmland under organic management.

[1] = baseline value set to 3% for all three landscape types.
[2] = number of territories predicted in baseline landscape consisting of 300 fields.

This model predicted a large increase in Skylark territory numbers under organic farming, most noticeably in pastoral rather than arable landscapes.

Manipulation 5: Changing the ratio of arable:pastoral land

The same three species used for manipulations 1-3, Lapwing, Skylark and Yellow Wagtail, could be used to predict the impact of increasing or decreasing the ratio between arable land and pasture. In addition, the model predicting Rook abundance in 5 km squares was solely dependant on grassland abundance (in three different categories), and the presence of only grassland had a negative effect in the model predicting Yellowhammer territories on field boundaries.

Lapwing

	Pastoral		Mixed		Arable	
% of farmland as grassland	**Birds[2]**	**% change**	**Birds[2]**	**% change**	**Birds[2]**	**% change**
-10%	48.14	+4.48	56.98	+4.63	63.47	+3.49
-5%	47.08	+2.29	55.72	+2.31	62.40	+1.74
Baseline[1]	46.03	-	54.46	-	61.33	-
+5%	44.97	-2.29	53.19	-2.31	60.26	-1.74
+10%	43.92	-4.48	51.93	-4.63	59.19	-3.49

Table 13 Changes in the predicted numbers of fields occupied by Lapwings with manipulations of the amount of grassland and tillage in the agricultural landscape.

[1] = baseline values are pastoral = 68.79%, mixed = 41.91% and arable = 15.13% grassland.
[2] = Based on a baseline landscape of 126 fields.

The Lapwing model predicted that fields were more likely to be occupied if they were not grassland. However, it should be mentioned that this model fails to describe the needs of breeding Lapwings: Lapwings may prefer arable land for nesting, but then prefer to chick-rear in pasture (Galbraith 1988). Therefore, a mosaic of both arable land and pasture is possibly better for Lapwings, a fact not reflected by the predictions of this model.

Skylark

	Pastoral		Mixed		Arable	
% of farmland as grassland	Birds[2]	% change	Birds[2]	% change	Birds[2]	% change
-10%	289.50	+17.38	438.47	+11.48	511.34	+9.23
-5%	268.06	+8.69	415.90	+5.74	489.73	+4.61
Baseline[1]	246.62	-	393.33	-	468.13	-
+5%	225.18	-8.69	370.76	-5.74	446.52	-4.61
+10%	203.74	-17.38	348.19	-11.48	424.92	-9.23

Table 14 Changes in the predicted number of Skylark territories in response to manipulations of the amount of grassland and tillage in the agricultural landscape.

[1] = as for Lapwing, Table 14.
[2] = number of territories predicted in baseline landscape consisting of 300 fields.

Predicted numbers of Skylark territories increase with an increased proportion of arable land in the agricultural landscape, and decrease with an increased proportion of grassland. This is particularly marked in pastoral landscapes, although this is due to the increases being on a sparser original population (and so a greater percentage increase) rather than any difference in the empirical size of the increases.

Yellow Wagtail

	Pastoral		Mixed		Arable	
% of farmland as grassland	Birds[2]	% change	Birds[2]	% change	Birds[2]	% change
-10%	0.5606	+18.62	0.7607	+11.88	0.9409	+9.0
-5%	0.5166	+9.31	0.7204	+5.94	0.9020	+4.5
Baseline[1]	0.4726	-	0.6799	-	0.8632	-
+5%	0.4286	-9.31	0.6394	-5.94	0.8243	-4.5
+10%	0.3846	-18.62	0.5989	-11.88	0.7855	-9.0

Table 15 Changes in the predicted density of Yellow Wagtails in response to manipulations of the amount of grassland and tillage in the agricultural landscape.

[1] = as for Lapwing, Table 14.
[2] = birds.km^{-2}.

As with Skylarks and Lapwing, Yellow Wagtails are predicted to decrease with an increase in the proportion of grassland and increase with a greater proportion of arable land, as they are found at higher densities in arable crops than in grassland.

Rook

	Pastoral		Mixed		Arable	
% of farmland as grassland	Birds[2]	% change	Birds[2]	% change	Birds[2]	% change
-10%	8.08	-13.39	5.45	-15.5	2.87	-12.23
-5%	8.70	-6.70	5.95	-7.75	3.07	-6.12
Baseline[1]	9.33	-	6.45	-	3.27	-
+5%	9.95	+6.70	6.95	+7.75	3.47	+6.12
+10%	10.58	+13.39	7.45	+15.5	3.67	+12.23

Table 16 Changes in the predicted density of Rooks with manipulations of the amount of grassland and tillage in the agricultural landscape.

[1] = as for Lapwing, Table 14.
[2] = pairs.km^{-2}

In all three baseline landscapes increasing the area of grassland by converting arable land results in predicted increases in the Rook population of between 1.2-1.6% per 1% of agricultural land converted. Actual increases are far greater in pastoral farmland, as this is predicted to have a starting density nearly three times greater than that in arable land. For each 1% of agricultural land converted the model predicts an increase of 0.125 birds.km^{-2} in pastoral farmland, 0.1 birds.km^{-2} in mixed farmland and 0.4 birds.km^{-2} in arable farmland.

Yellowhammer

An increase in the proportion of farmland as grassland would result in an increase in the occurrence of "grass-only" in the Yellowhammer habitat-association model.

	Pastoral		Mixed		Arable	
% of farmland as grassland	Birds[2]	% change	Birds[2]	% change	Birds[2]	% change
-10%	111.92	-5.43	133.08	+2.36	142.14	-1.56
-5%	115.32	-2.71	135.82	-1.18	143.24	-0.78
Baseline[1]	119.4	-	138.1	-	144.4	-
+5%	122.88	+2.71	139.94	+1.18	145.52	+0.78
+10%	125.89	+5.43	141.36	+2.36	146.66	+1.56

Table 17 Changes in the predicted number of Yellowhammer territories with manipulations of the amount of grassland and tillage in the agricultural landscape.

[1] = as for Lapwing, Table 14.
[2] = number of territories predicted in baseline landscape consisting of 387 field boundaries.

The Yellowhammer model predicts that increasing the proportion of grassland (and hence increasing the occurrence of "grass-only" adjoining field boundaries) will produce relatively small decreases in Yellowhammer territory numbers.

Manipulation 6: Changing the proportion of field boundaries with hedgerows

Whereas previous manipulations have relied on models for field-centre nesters such as Lapwing and Skylark, changes to field boundaries are likely to have a greater impact on those species that use boundaries for nesting, such as Yellowhammer and Whitethroat, and it is models for these species that contain hedgerow or hedgerow characteristics as a predicting variable.

Woodpigeon

	Pastural		**Mixed**		**Arable**	
% of total boundary length hedgerow added to	**Birds²**	**% change**	**Birds²**	**% change**	**Birds²**	**% change**
Baseline¹	0.518	-	7.828	-	1.678	-
+5%	0.9308	+79.69	7.820	-0.1	1.873	+11.62
+10%	1.344	+159.4	7.812	-0.2	2.068	+23.24
+15%	1.7572	+239.2	7.804	-0.3	2.263	+34.86

Table 18 Changes in the predicted number of Woodpigeons in field boundaries with the addition of hedges to field boundaries.

[1] = baseline values are pastoral = 43.37% of boundaries have hedgerow, mixed = 62.0% and arable = 51.11%.
[2] = number of birds along 200m transect of field boundary.

The results predicted for the Woodpigeon model and given above clearly illustrate the problems with extrapolating a model onto landscapes dissimilar from that used to generate it. The baseline estimates are wildly disparate, and while the addition of hedgerow in the pastoral landscape causes a massive increase in the occurrence of Woodpigeons, the same manipulations in mixed farmland produce a prediction of a slight decline.

Skylark

It is hard to relate boundary index to actual changes in hedge presence, as boundary index is calculated as a mean of categorical scores. However, as an indication of the negative impact of field boundaries on Skylark territory numbers, Table 19 gives the predicted change in Skylark numbers following the increase of boundary scores by 1. This increase was only applied to boundary originally scored at under 0.5 (0 being no boundary, 1 being a low hedgerow).

	Pastoral		**Mixed**		**Arable**	
% of all boundaries increase added to	**Birds[2]**	**% change**	**Birds[2]**	**% change**	**Birds[2]**	**% change**
Baseline[1]	246.62	-	393.33	-	468.13	-
+5%	243.04	-1.45	389.14	-1.42	460.55	-1.62
+10%	239.47	-2.90	384.95	-2.84	452.96	-3.24
+15%	235.89	-4.35	376.57	-4.26	445.38	-4.86

Table 19 Changes in the predicted number of Skylark territories with increases in boundary index values to simulate the addition of hedgerows.

[1] = as for Woodpigeon, Table 18.
[2] = number of territories predicted in baseline landscape consisting of 300 fields.

Yellowhammer

	Pastural		**Mixed**		**Arable**	
% of total boundary length hedgerow added to	**Birds[2]**	**% change**	**Birds[2]**	**% change**	**Birds[2]**	**% change**
Baseline[1]	119.4	-	138.1	-	144.4	-
+5%	122.41	+2.52	142.43	+3.13	147.72	+2.30
+10%	125.36	+4.99	145.28	+5.19	151.58	+4.97
+15%	128.48	+7.60	149.06	+7.94	155.40	+7.62

Table 20 Changes in the predicted number of Yellowhammer territories with the addition of hedges to field boundaries.

All three baseline landscape types give similar predictions for increases in Yellowhammer territories (approximately 0.5% increase for every 1% of boundary hedge row is added to).

Manipulation 7: Changing the proportion of field boundaries with ditches

Yellowhammer

	Pastural		Mixed		Arable	
% of boundaries ditches added to	**Birds**[2]	**% change**	**Birds**[2]	**% change**	**Birds**[2]	**% change**
Baseline[1]	119.4	-	138.1	-	144.4	-
+5%	120.3	+0.73	139.0	+0.65	145.3	+0.72
+10%	121.1	+1.46	139.9	+1.30	146.2	+1.45
+15%	122.0	+2.19	140.8	+1.95	147.1	+2.17

Table 21 Changes in the predicted number of Yellowhammer territories with the addition of ditches to field boundaries.

[1] = the baseline value was the same for each of the three landscapes, as data was only available for one (mixed, from the dataset used to generate the Yellowhammer model). Baseline = ditches present on 46.23% of field boundaries.
[2] = number of territories predicted in baseline landscape consisting of 387 field boundaries.

The addition of ditches to field boundaries results in a slight increase in the predicted number of Yellowhammer territories.

Manipulation 8: Changing the proportion of field boundaries with uncropped field margins

Field margin width occurs as a predictor variable in models for two species, Whitethroat and Yellowhammer. The model for Whitethroat uses width of margins of herbaceous vegetation as a parameter, not simply margins of any type; for the manipulation of this model it must be assumed that the margins are not simple sown grass mixtures but contain 'weedy' herbaceous vegetation; this vegetation is a major component of what makes field margins important for breeding Whitethroats.

	Pastoral		Mixed		Arable	
	Birds	**% change**	**Birds**	**% change**	**Birds**	**% change**
Baseline[1]	119.4	-	138.1	-	144.4	-
+10%	121.6	+1.98	140.8	+2.07	146.9	+1.94
+20%	124.1	+3.96	143.8	+4.15	149.9	+3.88
+30%	126.5	+5.95	146.7	+6.23	152.8	+5.82

Table 22a Changes in the predicted number of Yellowhammer territories with the addition of 2 m uncropped margins to field boundaries.

	Pastoral		Mixed		Arable	
	Birds	**% change**	**Birds**	**% change**	**Birds**	**% change**
Baseline[1]	119.4	-	138.1	-	144.4	-
+10%	127.3	+7.10	147.9	+7.47	154.0	+7.20
+20%	136.3	+14.20	158.74	+14.94	165.3	+14.40
+30%	144.9	+21.30	169.06	+22.42	175.6	+21.61

Table 22b Changes in the predicted number of Yellowhammer territories with the addition of 6 m uncropped margins to field boundaries.

Increasing field margins causes substantial (and similar) increases in the predicted number of Yellowhammer territories in all three baseline landscapes. The effect of increasing the width of the added margins is not linear, as there is a slightly higher return per metre for wider margins. However, as the modelling dataset included no margins greater than 6 m, it is not known (and unsafe to predict) how Yellowhammer populations would react to even greater widths of margin being added.

Whitethroat

The baseline for the Whitethroat model is the same for all three landscape types; there was no evidence that hedge height varied between landscape types and uncropped field margin was set at the mean value for the Leicestershire and Wiltshire study areas combined, due to the lack of reliable data from other regions.

	2m uncropped margins added		6m uncropped margins added	
% of boundaries with margin added	Sections with birds[2]	% change	Sections with birds[2]	% change
Baseline[1]	29.69	-	29.69	-
10%	36.58	+23.06	42.67	+47.26
20%	43.48	+46.12	54.73	+84.51
30%	50.24	+69.18	67.35	+126.77

Table 23 Changes in the predicted number of hedgerows occupied by Whitethroats with the addition of 2 m and 6 m uncropped margins to field boundaries.

[1] = mean margin (both sides of the hedge combined) 0.8 m.
[2] = number of 100 m hedgerow sections with Whitethroats present in the breeding season out of 94.

This model predicted very significant increases in the presence of Whitethroats in hedgerows for the addition of uncropped margins (2 m, or to even a greater extent 6 m) to a comparatively low proportion of hedgerows.

3.5 Species Summaries

In the section below, the approach taken previously (reporting the predicted increase in response variable (some measure of bird abundance) caused by a given change in habitat variable(s)) has been changed to that of reporting the habitat changes required to achieve a given increase in the response variable. In each case the manipulations quoted (always in the order pastoral/mixed/arable landscape) are those that would produce a 1% increase in the predicted response, be that numbers (Yellowhammer, Skylark) or % presence (Lapwing, Whitethroat). All changes in habitat variables are presented as % of the total agricultural area or % of the total field boundaries that manipulations were performed upon.

Lapwing

The lapwing model used in this study allows for three ways of increasing the probability of a given field being used by breeding Lapwings. However, as the relationship with area is linear, while increasing field size will increase the likelihood of Lapwing occupation, the resulting decrease in the number of fields for Lapwings to breed in will negate this increase. Secondly, decreasing the number of fields neighboured by woodland is not a sensible management option given the disbenefits that this would offer to other bird species, although the model does predict that this would result in an increased occupancy by Lapwings. It should be noted, however, that given the negative impact of adjacency to woodland on

Lapwings, management action aimed at increasing Lapwing numbers would be best aimed at fields not bordered by woodland.

The model did show that crop type does also have a significant impact on the likelihood of a field being occupied by lapwings. Option 1b in the Arable Stewardship was greatly preferred to other field types, followed by set-aside and stubbles. Grassland was the least preferred field type, followed by autumn-sown cereals.

An increase of 1% in the predicted number of fields that Lapwings attempted to breed in could be achieved by a 1.00/1.05/1.24% increase in set-aside at the expense of arable land, while a similar increase could be gained by a conversion of 2.23/2.16/2.86% from grassland to arable (a decrease of the same magnitude would result from the conversion the other way). The greatest return would be from an increase in Option 1b of the pilot Arable Stewardship Scheme, which is predicted to produce a 1% increase for every 0.43/0.63/0.50% of the total agricultural area given to this treatment.

It should be mentioned here that this model does not truly represent the requirements of breeding Lapwings, as there is no requirement in the model for grassland. As will be discussed in the review section of this report, Lapwings prefer to use pasture for chick rearing, even though tillage is preferred for nesting. Attempts to build this consideration into the model were unsuccessful.

Woodpigeon

The model taken from Parish, Lakhani & Sparks (1995) was not successful at predicting the impact of increasing the proportion of field boundaries with hedgerows, producing wildly different (and unbelievable) predictions for the three different baseline landscapes. It has been retained in this report as an example of how models that provide a good fit to the data used to generate them can perform poorly when used on new datasets, especially those from other regions or landscape types.

Skylark

The Skylark model used in this report was the most heavily parameterised, using 10 predictor variables including a crop-type covariate with eight states and differing values for three parameters depending on the farming system. As reported previously (Wilson *et al.* 1997, 2000) this model has proved excellent at predicting Skylark abundance. The manipulation of a number of variables proved to have significant effects on predicted territory numbers. Converting arable to set-aside would provide 1% more Skylark territories for every 1.54/2.77/2.15% of farmland converted, while the same increase could be achieved by replacing rape with other crops on 1.48/3.80/1.41% of all land. A similar level of return would be obtained by converting conventional farmed land to organic – 1% more Skylarks for every 1.61/2.36/3.96%, while 2.75/2.38/1.69% of farmland would have to be converted from autumn-sown to spring sown for the same predicted increase. The most efficient (in terms of area manipulated, not necessarily effort) means of attaining this increase is converting grassland to tillage, with 0.57/0.87/1.08% required.

Yellow Wagtail

The use of habitat-specific density estimates meant that only changes in cropping could be considered. Of the manipulations, converting grassland to arable was the one that gave the best return, with 0.54/0.84/1.11% needing converting to produce a 1% increase in predicted density. Other manipulations were less successful, with the sowing season of cereals needing to be moved from autumn to spring for 5.11/5.47/7.08% of all farmland to produce a 1% increase, while increasing set-aside was predicted to result in a very slight increase in pastoral landscapes but actually a decrease (again slight) in mixed and arable land.

Whitethroat

The binary logistic model for Whitethroats allows for two ways of increasing the number of hedgerow sections occupied by Whitethroats: decreasing the height of hedges and increasing the width of uncropped field margins. Although the former would also have benefits for other farmland bird species (field centre nesters such as Skylark and Lapwing) this would be massively outweighed by negative impacts on a greater suite of hedge-nesting species and so is not practicably available as a management option for increasing Whitethroat numbers. However, increasing margin width gave a huge return, with very large increases in Whitethroat occupation of hedgerows predicted for the addition of margins to comparatively low proportions of hedges. A 1% increase in Whitethroat present was predicted for every 0.43% of hedges that 2 m wide margins were added to, while 6 m margins would only have to be added to 0.24% for the same effect. It should be noted that there is a non-linear effect with the width of margin being added – although 6 m margins produce a greater increase in Whitethroat presence than 2 m ones, this increase is just under twofold, not threefold as would be expected if the relationship was linear.

Rook

The model for Rooks uses only large-scale habitat variables relating to the abundance and type of grassland. Although this model described a significant proportion of variation in Rook numbers in the region in which it was developed (County Durham), it failed to describe the majority of this variation. Therefore it must be emphasised that although the availability of different grassland types is important to Rooks, and it may be possible to increase Rook numbers (if this was desired) by increasing the availability of rough grassland, for example, there are obviously other habitat features that determine Rook abundance. It is likely that such features should be measured on a smaller spatial scale than the 5 km squares used in the model manipulated in this report.

The model predicted that converting arable land to grassland would produce 1% additional Rook pairs for every 0.74/0.64/0.82% of farmland converted.

Yellowhammer

The Yellowhammer habitat-association model allows four manipulations that would result in a predicted increase in Yellowhammer territories: increasing the proportion of field boundaries with hedge and ditches, increasing the width of uncropped field margins and decreasing the number of fields containing grassland.

Adding ditches was predicted to increase Yellowhammer numbers for every 6.84/7.69/6.91% of boundaries they were added to, a comparably low return compared with the addition of hedges to boundaries, which provided the same increase for every 1.97/1.89/1.96%. The effect of adding uncropped field margins very much depended on the width of those margins – an increase of 1% required 5.04/4.81/5.15% of new 2 m margins but only 1.41/1.34/1.39% of 6 m margins. Lastly converting grassland to arable is predicted to produce an increase of 1% for every 1.84/4.24/6.41% of total farmland area converted.

3.6 Discussion

This study has attempted to use habitat association models to predict the effect of changes in the agricultural landscape on as many species in the farmland bird index as possible. That only seven species have been covered, and with different degrees of confidence and success, suggests that there is much scope for further collection of data and analysis to refine the existing models and produce models for species that have not so far been studied in this way.

There are a number of constraints and pitfalls that should be borne in mind when employing the methods used in this report to predict the impact of changes in the agricultural landscape. Habitat association models are empirical, developed using mathematical relationships between predictor variables and the response variable – they are not theoretical models, but simplified (sometimes grossly) approximations of reality. They can prove to be excellent predictive tools if they can fulfil two criteria, (a) that they have sufficiently high goodness-of-fit and (b) are reasonably successful at explaining bird distribution in other regions and/or habitats. Examples of models failing to meet these criteria are to be found in the preceding pages. The model developed to describe Linnet distribution in Oxfordshire simply failed to predict the occurrence of nest sites accurately – the associations between variables entered into the modelling process and nest distribution were too weak. Other models, such as that for Lapwing, provided a good approximation for the data used to develop it, and indeed is likely to be good at predicting the impacts of agricultural changes in the West Midlands. However, as BTO survey data shows (Wilson *et al.* 2001), it is unlikely to provide accurate predictions for different regions, as Lapwings show a very significant skew in their distribution, with low densities in the south and west but much higher densities in the north. Models such as those for Skylark and Yellowhammer were developed in mixed farmland, with intermediate levels of grassland and tillage. The values for the baselines for pastoral and arable landscapes lie outside the range found in mixed farmland, and so this extrapolation of the models beyond the range of landscape conditions from which they were generated mean that the resulting predictions should be trusted less. That said, research conducted to describe Whitethroat distribution in two distinct regions with different landscapes produced remarkably similar models, indicating the generality of the factors controlling distribution.

Simple, correlative habitat association models are the least sophisticated of several types of models that could be used to predict the effect of projected changes in land management, being as they are static and demonstrating correlation rather than causation. However, there are many grounds to believe that the relationships found in these models are linked to known ecological processes. For example, the strong correlation between crop type, vegetation height and Skylark distribution is known to reflect the inhibiting effect of dense, tall vegetation on both nest building and successful foraging by adults (Odderskaer *et al.* 1997, Wilson *et al.* 1997, Donald 1999). Ideally habitat association models should be combined with information on species demographic rates, as this report attempts to do. Demographic models can inform, for example, whether constraints on population size do actually occur in the summer. All the models included above predict breeding distribution and make predictions of increases in numbers following habitat manipulations based on the assumption that the manipulated habitat variables were constraining population levels. However, it is distinctly feasible (and for some species likely) that it is factors outside the breeding season that determine the population levels of some farmland bird species.

This study has shown that it is possible to use habitat-association models to make predictions on the impact of farmland landscape changes on birds. It will allow the comparison of these impacts to aid in the decision-making process as to how increases in bird populations can be best achieved. The results show, for example, that uncropped margins are good for Whitethroats and Yellowhammer, that Lapwings benefit most from fallow areas and that Rooks will increase if the area of grassland was increased at the expense of arable land but that Skylarks and Yellowhammers would decrease. It also allows for the comparison of the relative returns of manipulations – quantitative rather than merely qualitative results. For example, predictions generated with the Yellowhammer model suggest that although adding both ditches and hedges to field boundaries will increase numbers, the return from hedges is over three times as high as for adding ditches. However, it can give no clear indication on the reliability and accuracy of these predictions. That can only be clarified by further investigation into the accuracy and geographical generality of habitat association models and their relationship with demographic models.

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CHAPTER 4 PREDICTING THE RESPONSE OF FARMLAND BIRD POPULATIONS TO AGRICULTURAL CHANGE: REVIEW OF SPECIES-SPECIFIC AND GENERIC RESOURCE REQUIREMENTS

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4.1 Summary

This review shows that an extremely wide range of resources is used throughout the year by the 20 species on the farmland bird index. Some of these species can be loosely classed as "generalists", indicating that they are found in a wide variety of habitats and feed upon a wide range of food types, whereas other species have a narrower range of requirements (specialists). However, it must be borne in mind that some species that might be regarded as generalists may have specific resource-requirements during at least one stage in their life history. The Starling is an example of one such species; Starlings feed upon a wide variety of food resources in many habitats, but have a specific requirement for holes in trees or buildings for nesting, and the lack of suitable nest-sites may limit them in some regions.

Conversely, species that might be regarded as specialists may demonstrate some plasticity in their requirements. Linnets, for example, have a diet restricted chiefly to weed seeds throughout the year, whereas other granivorous birds take grain and invertebrates at different times. However, with the decline in the abundance of many important weed species in recent decades, oilseed rape now provides a large proportion of the diet of Linnets in the breeding season, thereby reducing the impact of the loss of more "natural" food resources.

Tables 1 and 2 summarise the requirements (habitats and food) of each species in summer and winter. Categories have been kept broad deliberately, and for the specific details of resource requirements readers should refer back to the relevant text sections. These tables serve to highlight that many of the 20 species have common resource requirements, and the provision of some resource categories would in fact aid many species.

Nesting sites

Trees, shrubs and hedgerows are clearly important to breeding farmland birds, with 15 of the 20 index species using them rather than (or as well as) cropped areas. However, this is one example of a conflict between the interests of different species; although many birds need boundary features for nesting and/or foraging, a few (Grey Partridge, Lapwing, Skylark) show a marked aversion to at least some boundary features. In these cases, it may be necessary to consider targeting different areas for different species. Thankfully such direct clashes between the resource requirements of different species are rare.

Foraging habitat

As for nesting sites, it is clear that certain habitats are important for a significant proportion of birds on the farmland bird index. As with nest sites, the habitats used by the greatest number of species are uncropped ones, in particular uncropped margins, rank grass and set-aside in the breeding season and margins, set-aside and stubble in the winter.

Food

The two most commonly taken food categories in the breeding season are weed seeds (taken by 11 of the 20 species) and non-soil dwelling invertebrates (15 of 20). Weed seeds remain important in the winter, with 13 of the 17 species still present in the winter taking them.

4.2 Introduction

The resources required by farmland bird species can be loosely placed into three categories: food in the breeding season, nesting sites, and food in the winter. Food and habitat are inexorably entwined, and although choice of habitat may be determined by the food within that habitat (hence it is the food choice that determines habitat use) there may be other considerations such as proximity to nest site and the risk of predation while foraging.

There may be additional requirements, such as winter roost sites, and other seasonally specific requirements, such as needs of birds during the post-fledging period. For three species (Turtle Dove, Yellow Wagtail and Whitethroat) there are considerations of their migration to and wintering in sub-Saharan Africa, but these resource-requirements are considered to be outside the remit of this report, as they are unrelated to the UK agricultural environment (although the chance of a bird surviving autumn migration may be influenced by body condition at the start of migration, which may be influenced by its breeding environment; see Turtle Dove, below).

Although resource requirements fit into these three categories, disentangling these needs is not simple. The need for one resource has to be measured against the requirements for another. For example, nesting sites will be chosen not just for their suitability for nesting per se but also for their proximity to food resources. In the following species accounts resources requirements have been separated into "breeding" and "non-breeding" season requirements. Even these two categories are not independent of each other, as the needs of a species in the winter may influence its distribution in the summer, or vice versa.

4.3 Species Accounts

Kestrel *Falco tinnunculus*

The commonest raptor in the country, the Kestrel has an extensive distribution and can be found in a wide range of habitats, including agricultural landscapes. Densities are particularly high in the south-east, following recovery from the direct effects of pesticide mortality (Village 1990). However, there have been declines in western regions in recent years (Gibbons *et al.* 1993).

Breeding season requirements

Kestrels usually select territories with a number of alternative nesting sites, a suitable roost site and open country for hunting. They defend an exclusive territory around the nest at the beginning of the breeding season, but later in the season the territory may expand and overlap with that of neighbouring pairs. Most hunting is done within 2 km of the nest site, but the range is often much smaller, especially in years when prey is abundant. Although Kestrels can be found in most open countryside types, some habitats are able to support higher densities than others. The highest densities of nesting Kestrels are reached on grassland, with mixed farmland holding intermediate densities and arable farmland generally low densities. These densities may be related to the abundance of voles *Microtus* (Village 1990). Kestrels prefer to hunt in areas of long grass and rank vegetation, which holds the highest densities of small mammals. Therefore farmland with uncultivated areas or grassland with low grazing intensity is preferred (Village 1990).

Kestrels are adaptable in their use of nest sites, but do not build their own nests. Old or disused nests of Carrion Crows *Corvus corone*, Magpies, *Pica pica*, Grey Herons *Ardea cinerea* and Sparrowhawks *Accipiter nisus* are used, as are ledges on buildings and rock-faces. They are also regular hole-nesters, using natural tree-holes and also readily accepting nest-boxes. It is possible that in intensively-managed arable farmland the lack of suitable nesting sites (due to the loss of hedgerow trees) may depress Kestrel densities.

The diet of Kestrels consists largely of voles, with the most important being the Short-tailed (Field) Vole *Microtus agrestis*, which can account for up to 80% of Kestrel diet (Village 1982). Mice *Mus* and shrews *Sorex* are also preyed upon (especially the common shrew, *S. araneus*), and other mammals taken include young rabbits *Oryctolagus cuniculus*, squirrels *Sciurus* spp., moles *Talpa europaea* and rats *Rattus norvegicus*. Other vertebrates taken include small birds and lizards. The commonest bird species taken are open country nesters such as Meadow Pipits *Anthus pratensis*, Skylark *Alauda arvensis* and Starling *Sturnus vulgaris*. Fledglings of these species may be important food sources while brood-feeding, especially in urban areas (where House Sparrow *Passer domesticus* may be a very important component of diet, Yalden 1980).

Invertebrates are also taken, including caterpillars (mostly Noctuidae: Lepidoptera), ground beetles (Carabidae), dor beetles (Geotrupidae) and grasshoppers (Acrididae). In some habitats, or in years of low vole abundance, insects may be a very important component of the diet, and some authors (e.g. Simms 1961, Yalden & Warburton 1979) believe that this may have been underestimated by other studies.

Wintering requirements

There is little change in the habitat requirement of Kestrels in the winter, although birds breeding in upland regions may migrate to lowland or coastal areas (Snow 1968). Densities increase in arable areas; it has been suggested that this may be due to many first-year birds migrating away from the breeding area and settling on arable land as they can obtain earthworms (Lumbricidae) from freshly tilled fields (Shrubb 1980, Pettifor 1983). Earthworms form a large proportion of Kestrel diet in the winter, especially for first-year birds, and birds will follow the plough in the autumn. Kestrels will also forage for ground beetles and other invertebrates in cereal stubbles after harvest. Voles and, to a lesser extent, small birds, remain important.

Grey Partridge *Perdix perdix*

The Grey Partridge has shown massive declines in population and range since the early part of the twentieth century. Most obviously it has retreated from the western part of its range; it is now absent or scarce in Cornwall, Devon, Wales, NW England and West Scotland, for example. CBC results indicate that the British population declined by approximately 75% between the two Breeding Bird Atlases (Gibbons *et al.* 1993) and Potts (1980) estimated a decline of 80% between 1952 and 1986. The principal driving force behind this decline is a decline in chick survival rates from 45% to below 30% (Potts 1980). This decline has been linked to herbicide use reducing the availability of invertebrate food for chicks (Potts 1986). Similar declines have been recorded in Grey Partridge populations throughout Europe and North America. Grey Partridge is on the red list of Birds of Conservation Concern (Gibbons *et al.* 1996) and a UK BAP Priority Species.

Breeding season requirements

Grey Partridges are birds of open ground, having evolved on steppe grassland and subsequently adapted to the agricultural landscape. In Britain they are particularly linked to areas of cereal growing. They are sedentary, so to be suitable an area has to provide habitat suitable for both breeding and wintering.

Grey Partridges make shallow nest scrapes in dense grassy cover. They prefer nest sites that are above level ground, i.e. on raised ground such as a bank or hummock (Rands 1986). They also tend to avoid banks or hedges with more than 10 trees per kilometre of linear cover (Rands 1982). The availability of such sites – densely and permanently grassed, raised but away from treelines – explains most of the difference in Partridge abundance between estates (Potts 1980). Grey Partridges tend not to nest within the crop itself, but will do if more suitable sites are not available.

Predation of Partridge nests (either of the eggs or the sitting hen), particularly by foxes *Vulpes vulpe,* stoats *Mustela erminea,* weasels *Mustela nivalis,* hedgehogs *Erinaceus erinaceus,* rats *Rattus norvegicus* and Corvids (Carrion Crows *Corvus corone* and Magpies *Pica pica*) may depress population levels (Potts 1986, Dowell 1988). There is some evidence to suggest that predator control measures may result in raised Partridge numbers (Potts 1986, Tapper *et al.* 1996). A study in France by Bro *et al.* (2001) found that predation (by foxes *Vulpes vulpes*, mustelids and Hen and Marsh Harriers *Circus cyaneus* and *C. aeruginosus*) determined female survival rate.

In the early part of the breeding season the diet of (adult) Grey Partridges is mainly plant in origin, with grass and cereal leaves dominating, but clover leaves and weed seeds being preferred if available (Potts 1986). More invertebrates are taken later in the season when the chicks are being fed, but even then only about 12% of the diet is insects, mainly ants (Formacidae).

In contrast to the parent diet, chick diet is mainly animal. Ninety-five percent of food items eaten in the first week of life and 80% in the first two weeks are insects (Vickerman & O'Bryan 1979). A wide variety of species are taken, including spiders (Araneae), crane-flies (Tipulidae), ground beetles (Carabidae), weevils (Curculionidae), rove beetles (Stapylinidae), leaf beetles (Chrysomelidae), grasshopper nymphs (Acrididae), ant pupae (Formicidae), caterpillars (Lepidoptera), sawfly larvae (Symphyta), ichneumon wasps (Ichneumonidae, Braconidae), plant bugs (e.g. Delphacidae, Cicadellidae) and aphids (Aphididae). Although chicks will eat plant matter, even high protein plant food such as clover leaves are not sufficient for chick growth, so insect food is vital in the first few days of life. Annual variation in the abundance of favoured chick food insects explains 48% of variation in chick mortality (Green 1984). There is a strong body of evidence to suggest that chick starvation because of reduced food resources caused by insecticide and herbicide use is the main cause of recent declines in Grey Partridge populations (Rands 1985, Potts 1986). Measures such as conservation headlands and beetle banks are being advocated to reverse this decline (Sotherton 1991, Game Conservancy Trust 2001).

Wintering requirements

As mentioned previously, Grey Partridges remaining in the breeding area over the winter, gathering together in coveys consisting of a few adult birds and their young.

Polygonaceae, Caryophyllaceae, Leguminosae, Labiatae and Gramineae are the most important plant components of Grey Partridge diet in the winter (Wilson *et al.* 1997a). Within these families the most favoured foods are the seeds of bistorts *Polygonum* (especially redshank *P. persicaria*, knotgrass *P. aviculare* and black bindweed *P. convolvulus*), corn spurrey *Spergula arvensis* and chickweeds *Stellaria*, meadow grass *Poa*, cereal grain and leaves (wheat, barley, oats) and the leaves and pods of vetches *Vicia* and clovers *Trifolium*. Also taken are the seeds of dead-nettles *Lamium*, hemp-nettles *Galeopsis*, gromwells *Lithospermum* and cornflowers *Centaurea*. The availability of such seeds has declined massively in recent years, with for example black bindweed present in only 4% of cereal fields in 1982 (Chancellor & Froude-Williams 1984), and Polygonum has declined from contributing 31% of the total amount of food in Partridge crops examined in the 1930s to 2% of those examined in 1977 (Potts 1984). The same foods remain important throughout the winter, with the addition of the foliage of autumn-sown cereals, which becomes more important as grain and weed seed stocks are depleted.

Lapwing *Vanellus vanellus*

Lapwings are widely spread throughout Britain, with 83% of British 10-km squares holding Lapwings during the 1988-92 Breeding Bird Atlas survey (Gibbons *et al.* 1993). However, although widespread, Lapwing populations have declined since the mid-1950s. Nicholson (1938) estimated the Lapwing population in England and Wales as 175,000 pairs. This had declined to 123,000 pairs by 1987 (Shrubb & Lack 1991) and then a further decline of 49% occurred between 1987 and 1998 when the population was estimated as 62,923 pairs (Wilson *et al.* 2001). As a consequence of this decline Lapwing is on the red list of Birds of Conservation Concern (Gibbons *et al.* 1996). The majority of English Lapwings breed on lowland farmland, but a small proportion (though sometimes at very high local densities) breeds in the uplands (O'Brien 2002).

Breeding season requirements

Both pastoral and arable farmland is important for Lapwings, with few birds breeding on other habitats; 95% of the English and Welsh population breed on farmland. The importance of damp lowland grassland has declined in recent decades (O'Brien & Smith 1992). The 1998 BTO/RSPB national Lapwing survey found 56% of the population breeding on grassland and 39% on arable farmland. When the selection of habitat types was compared to the availability of these habitats, there was a clear preference for spring tillage, followed by rough grazing. Set-aside was also favoured in some regions. Permanent grass was used as expected according to its availability, while ley grassland and in particular autumn tillage were avoided (Wilson *et al.* 2001). Autumn tillage is generally unsuitable for Lapwings as growth is too high and dense by the start of the Lapwing breeding season in April; Lapwings leave a site when cereal reaches 30 cm tall (Lister 1964).

The relationship between Lapwing distribution and habitat is not simple, as although Lapwings prefer to breed on tillage, the chicks are reared, if possible, on grassland. Grassland provides the invertebrate food required for chick rearing (see below) (Redfern 1982, Galbraith 1988). Wilson *et al.* (2001) found that while 17.8% of fields abutting grassland held Lapwings, only 5.2% of fields without adjacent grassland did. Therefore lapwings prefer a mosaic of different agricultural landtypes – mixed farmland. This is increasingly rare due to the move away from crop rotation in modern agriculture.

Grazing density may affect Lapwing clutch success, as high grazing pressure may result in nest trampling by livestock. In addition, the low sward structure diversity that results from high grazing pressure may result in higher predation rates of chicks, as refuges from predators are scarcer. Sward structure may have a significant effect on Lapwing breeding success, as the dense sward structure of modern improved grassland may both reduce the availability of nest sites and the accessibility of invertebrates within or below the sward (Shrubb 1990).

In the uplands, Lapwings favour short, damp swards, rushy pastures and rough grazing. They avoid leys, improved grass and unenclosed heath/moorland (Galbraith *et al.* 1984; Baines 1988). These preferences reflect two mechanisms. First, short swards enhance the ability of the birds to see and capture prey, regardless of prey density. Secondly, damp habitats increase availability of earthworms. Indeed, Lapwings are associated with fields with small flooded areas (O'Brien 2002). However, a fine balance needs to be achieved between a field that is damp enough that worms approach the soil surface, and a field that is so damp that earthworms are effectively 'drowned' (O'Brien 2002). The apparent selection of rushes may simply reflect the association of rushes with dampness, although the habitat heterogeneity that this affords may be important for hiding nests from predators.

The diet of Lapwings consists primarily of Mollusca, Annelida, Arachnida, Orthoptera, Hymenoptera, Diptera and Coleoptera. The most important prey groups are slugs (Mollusca), earthworms (Annelida), spiders (Araneae), leatherjackets (Tipulidae larvae), ants (Formicidae), ground beetles (Carabidae), weevils (Curculionidae), click beetles (Elateridae), dung beetles and chafers (Scarabeidae), grasshoppers (Acrididae), Buprestidae and Tenebrionidae. Soil invertebrates tend to be more important in the early part of the breeding season, but as the ground becomes harder lapwings tend to switch to feeding upon surface invertebrates (Baines 1990). Chick diet is varied, including most of the invertebrate classes listed above. Sheldon (in prep.) found that the body condition of Lapwing chicks was correlated with the number of earthworm setae per gram of faecal sample, suggesting that earthworms were a desirable food for chicks.

Recent research (Sheldon in prep.) has indicated that the "option 1B" of the pilot Arable Stewardship Scheme (an overwintered cereal or linseed stubble followed by a spring/summer fallow) may be beneficial for breeding Lapwings. Breeding success in this habitat was higher than in other habitat types, (although still below that thought necessary to maintain the population at present levels) and Lapwings demonstrated positive selection for nesting in option 1B fields.

Wintering requirements

Immigrants from the continent swell the British wintering population of lapwings in late autumn, with an estimate of at least 1,000,000 made by the 1981-84 Winter Atlas (Lack

1986). This number can vary markedly, as Lapwings make hard weather movements and large numbers may leave the country to avoid severe frost and snow (although local movements to southwest England are more usual).

Although they are found wintering throughout lowland Britain, Lapwings show a clear preference for grassland over arable land in the winter, so are found in high densities in the midlands rather than eastern England (Lack 1986). Plough and stubbles are used more often in the early autumn, with more birds using grassland and autumn-sown cereals in the winter, grassland becoming more favoured later in the winter. Lapwings prefer to feed in fields where crop growth is below 75 mm high (Village & Westwood 1994), so the fast growth of autumn-sown cereal fields may mean that by mid-winter they are too high for Lapwings to use.

Invertebrate remain the most important food resource throughout the winter, with earthworms providing a large proportion of diet. The transitory use of arable land in the autumn may be partly due to the exploitation of earthworms temporarily exposed by tilling, before birds move to feed on higher earthworm densities found in grassland (Village & Westwood 1994). Other soil-dwelling invertebrates, such as the larvae of Tipulidae, Carabidae and Curculionidae, remain important also (Cramp & Simmons 1983).

Lapwings prefer to both feed and roost in large fields, presumably as this allows for improved detection of predators (Shrubb 1988, Mason & Macdonald 1999). Ploughed fields are preferred for roosting (Lack 1986).

Stock Dove *Columba oenas*

Stock Doves are widespread and resident throughout lowland Britain (absent only from parts of Wales and north and west Scotland) (Gibbons *et al.* 1993). Stock Doves spread from a much smaller range in southern England, expanding the range north and west from the mid-19th century onwards with the expansion of arable farming (O'Connor & Mead 1984). Although there have been some recent declines and a slight range contraction the Stock Dove remains a common farmland bird.

Breeding season requirements

As a hole-nesting species, the breeding distribution of the Stock Dove is linked to the availability of trees with suitable nest sites, although if other habitat conditions are desirable then Stock Doves will nest in rabbit burrows and rock crevices, as well as in buildings. Nest boxes are readily accepted if provided. The preferred breeding habitats are woodland edges, parkland and wooded farmland, which provide a combination of mature trees likely to provide nesting sites and suitable open foraging habitats. The loss of hedgerows and trees due to recent agricultural intensification may have resulted in a limiting shortage of nest sites in agricultural habitats (Gibbons *et al.* 1993)

Diet of both nestlings and adult is almost entirely plant-based, with buds, leaves and fruits of a wide variety of plants taken at different stages of the breeding season. Amongst the 22 plant families recorded in the diet (Wilson *et al.* 1997a) bistort (Polygonaceae), goosefoot (Chenopodiaceae), chickweed (Caryophyllaceae), buttercup *Ranunculus* (Ranunculaceae) charlock *Sinapis arvensis* and cultivated *Brassica* (Cruciferae), vetch and clover (Leguminosae) are possibly the most important, although a wide range of other crop and weed species are also taken. Both seeds and leaves are taken, and the vast majority of foraging is terrestrial; Stock Doves do not spend as much time foraging arboreally as other Columbidae. Chicks are fed entirely on crop milk when very young (Cramp 1985).

Wintering requirements

There is little change in the distribution of Stock Doves between summer and winter, although as with many farmland bird species there is some withdrawal from upland areas (Lack 1986). Stock Doves gather in small flocks, and generally concentrate on arable areas. Weed seeds are the most important dietary component, with much the same species taken as listed above. Cereal grain can be the major constituent of diet in the autumn (Cramp 1985). *Brassica* seeds contribute up to 45% of the total diet in September-October and remain important throughout the winter, as do weed seeds such as bistort (Polygonaceae) and goosefoot (Chenopodiaceae). Although crop foliage is taken (Murton 1965) this is a less important part of diet than for Woodpigeons (see below).

Woodpigeon *Columba palumbas*

With a British breeding population estimated at 2,550,000 pairs (Gibbons *et al.* 1993), the Woodpigeon is the commonest of the species considered in this report. It is abundant throughout many habitats, but particularly in lowland farmland, where it can be a pest species.

Breeding season requirements

Woodpigeons are found in many habitats, but generally prefer some combination of trees and open spaces. When nesting in dense woodland or plantations (deciduous, coniferous and mixed are all used) they often feed in surrounding farmland rather than in the wood themselves. Woodpigeons are only absent from open upland, aquatic and coastal habitats, having adapted to living in urban areas (Simms 1975).

The breeding season of Woodpigeons can be extremely long, with egg-laying having been recorded in every month (Murton 1965). However most breeding activity is concentrated into summer, when Woodpigeons nest in woods, copses, treelines, hedges, scrubland and gardens. Nests are placed in tree branches or outer edges of shrubs, although they will nest on or near the ground under thick cover or on buildings (Cramp 1985).

Little or no animal food is taken by Woodpigeons, although earthworms and small insects are occasionally taken (Colquhoun 1951); instead a huge range of plant matter is taken. Depending on the season, buds, leaves, fruits and seeds are eaten. Some trees are important food species, including beech *Fagus sylvatica* (buds, flowers and seeds) (Fagaceae), ash *Fraxinus excelsior* (Oleaceae) (buds and flowers) and hawthorn *Crataegus* (Rosaceae) (buds, flowers and berries). Other plant food is taken by foraging on the ground. Leaves of many species are eaten, especially charlock *Sinapis arvensis* and cultivated *Brassica* such as oil-seed rape (Criciferae) and vetches and clovers (Leguminosae). Other plants taken include spurry *Spergula*, chickweeds *Stellaria*, mouse-ears *Cerastium* (Caryophyllaceae), buttercup and celandine *Ranunculus* (Ranunculaceae), speedwell *Veronica* (Scrophulariaceae), plantain *Plantago* (Plantaginaceae) dandelion *Taraxacum* (Compositae), nettle *Urtica* (Labiatae) and mallow *Malva* (Malvaceae). Nestling diet is much as for adults, with the addition of crop milk.

Wintering requirements

Although British Woodpigeons are largely resident, there are small-scale movements in the autumn as some birds from the north and west move south (Murton & Ridpath 1962). The largest winter concentrations occur in the large arable farming regions of south and east England, where flocks of thousands gather. There is a relatively small amount of immigration from the continent (Lack 1986).

The most important food for Woodpigeons over the winter is the foliage of crops. In the 1950s and 1960s this consisted principally of clover grown on leys. With the rapid decline of this agricultural practice in the late 1960s and 1970s Woodpigeon numbers declined. However, numbers returned to previous levels from the late 1970s onwards following the introduction of oilseed rape. The foliage of autumn-sown oilseed rape is now the most important food resource for Woodpigeons, and the area sown with this crop is the limiting factor to Woodpigeon population size (Inglis *et al.* 1990). This crop is required in the late winter, when other food resources used by Woodpigeons in the winter have been exhausted (Murton 1965). Stubbles are foraged upon in the autumn, with birds moving onto winter cereals after spilt grain and weed seeds are consumed. In addition to crops (either foliage or seeds) a large number of weed species are taken of the species listed under breeding season requirements.

Aside from feeding on crops, Woodpigeons forage upon tree seeds (oak *Quercus* and beech *Fagus sylvatica* (Fagaceae) in particular) and fruit such as hawthorn *Crataegus* (Rosacae) and elder *Sambucus* (Caprifoliaceae), although these latter food resources are usually depleted by early winter.

Interestingly, a number of studies have shown that the large part of Woodpigeon food intake is in the late afternoon (Gibb & Hartley 1957, Murton *et al.* 1963). Woodpigeons roost in large concentrations in woods and copses, as well as in smaller groups in hedgerows and isolated trees. They may travel long distances – possibly up to 65 km – to roost sites (Cramp 1985).

Turtle Dove *Streptopelia turtur*

Turtle Doves have a restricted range in Britain, being found in the south and east of the country, with very few birds in Wales, the south-west or north of the Humber. This range has contracted in recent decades (Gibbons *et al.* 1993), accompanied by a dramatic fall in numbers: there was a 70% decline in abundance in the UK between 1970 and 1998 (Gregory *et al.* 2001). This recent decline follows a long and gradual increase in both numbers and range since the mid-19th century, if not earlier (Holloway 1997). The most recent population estimate is 30,000 pairs in 2001 (Browne & Aebischer 2001), and the Turtle Dove is red-listed as a Bird of Conservation Concern (Gibbons *et al.* 1996) and a UK BAP Priority Species.

As a summer visitor, arriving in late April/early May and departing in September, a large portion of the life history of Turtle Doves lies outside the remit of this review. It is possible that past population declines, and future trends, may be in part or whole caused by factors either during migration or on the wintering grounds (the Sahel region of Africa, Cramp 1985). However, as well as affecting breeding productivity, factors on the breeding grounds may affect the body condition of birds in the pre-migration period, and hence their likelihood of reaching the wintering grounds successfully.

Breeding season requirements

Little serious study has been devoted to Turtle Dove in Britain until the publication of a recent study (Browne & Aebischer 2001), apart from the work of Murton *et al.* (1964). This review depends heavily on these two sources of information.

Turtle Dove distribution appears to be linked to the presence of suitable nesting habitat - scrub, large hedges, small coniferous plantations and woodland edge. Intensively managed (annually cut) hedges are rarely used as nest sites. Shrubs, especially those overgrown and thorny (hawthorn, *Crataegus monogyna*, is particularly preferred), are preferred to trees as nest sites. Shrubs with climbers such as Traveller's Joy *Clematis vitalba* (Ranunculaceae), Bramble *Rubus fruticosus* (Rosaceae) and Honeysuckle *Lonicera periclymenum* (Caprifoliaceae) are preferred: 53% of nests found by Browne & Aebischer (2001) were within 0.1 m of climbers. Territory size ranges from 1.91 to 3.08 ha.

The immediate vicinity of nest sites is not necessarily important as feeding sites for Turtle Doves, as they will travel up to 10 km (on average 0.5 to 1.5 km) to forage (Browne & Aebischer 2001). In the 1960s Turtle Doves made extensive use of "natural" feeding sites. Before they largely disappeared from the agricultural landscape, fields of clover leys and hay were important feeding areas in the early part of the breeding season. Similarly, the stoked wheat and weedy pea fields used for foraging around harvest time are no longer available. Previously, Turtle Dove diet primarily consisted of weed seeds such as goosefoots *Chenopodium* (Chenopodiceae), fumitories *Fumaria* (Fumarizceae), charlock *Sinapis alba* (Cruciferae), sunflowers *Helianthus* Compositae), fescues *Festuca* (Gramineae), bistorts *Polygonum* (Polygonaceae) and medicks *Medicago* (Leguminosae) (Murton *et al.* 1964, Wilson *et al.* 1997a), with little or no exploitation of crop seeds. However, a decline in the abundance of these agricultural weeds has resulted in a shift in foraging behaviour and diet (at least in some regions, such as East Anglia). Browne & Aebischer (2001) found that Turtle Dove diet consists primarily of crop seeds, especially wheat and oil-seed rape, which Turtle Doves foraged for at "artificial" sites – spilt and stored grain and animal feed, with little foraging on "natural" habitats. When Turtle Doves did forage in natural habitats they preferred weedy areas, particularly set-aside and rough ground which receive no herbicide applications in the breeding season and were particularly rich in field pansy *Viola arvensis* (Violaceae), fumitory and chickweeds *Stellaria* (Caryophyllaceae). This current predominance of crop seeds was found for both adult (60% crop seeds, compared to 23% in Murton's 1963 study (Murton *et al.* 1964)) and chick diet (69%, compared with just 5% in 1963).

Turtle Doves feed on sites with short (about 12 cm or less) and sparse (40% or less) vegetative cover. When feeding in crops they are usually restricted to weedy strips round the perimeter, or stubbles after harvest.

The current decline in Turtle Dove numbers may be linked to the shortening of the breeding season and consequent reductions in the number of breeding attempts. In the 1960s 24% of breeding attempts were started in August, compared with 5% nowadays (Browne & Aebischer 2001). This reduction in the length of the breeding season has resulted in overall productivity nearly halving. This decline may be associated with reduced food availability and the increased distance which birds have to forage over, with the subsequent effect on adult body condition resulting in birds ceasing breeding earlier.

Barn Owl *Tyto alba*

As a relatively scarce, widespread and nocturnal species, the Barn Owl is difficult to survey accurately. In addition, it is well established that the population undergoes cyclical fluctuations related to cycles of vole populations, making comparisons between single-year surveys unreliable. Therefore there is little accurate information on population trends, although it is clear that the British Barn Owl population has declined. The national survey conducted from 1995 to 1997 produced estimates between 3,480 and 3,951 pairs for the three years in which it was conducted (Toms *et al.* 2001). These estimates are slightly down on the 4,400 pairs estimated by a 1982-85 census (Shawyer 1987) and massively down on the figure of 12,000 pairs in England and Wales derived from a survey in 1932 (Blaker 1933). Although still widespread, the Barn Owl is now rare or absent from many areas of Britain (Gibbons *et al.* 1993), and is on the amber list of Birds of Conservation Concern (Gibbons *et al.* 1996). There is evidence that organochlorine poisoning may have played a role in the decrease of Barn Owls; both breeding success and survival rates have increased since the mid-1970s, indicating a recovery from those deleterious effects (Percival 1991).

Breeding season requirements

As the English name suggests, Barn Owls commonly breed in buildings, particularly unoccupied agricultural buildings such as barns. Large tree cavities and (to a lesser extent) rock fissures and caves are also used. Just over two-thirds of all nests are in buildings (Shawyer 1987) although there are considerable regional variations with, for example, 95% of all nests in Devon being in buildings (Ramsden 1995) but birds in eastern England preferring to nest in trees (Shawyer 1987). Barns with haylofts are preferred for breeding (Cayford 1992). The loss of barns through conversion to dwellings may have had a significant effect on the availability of nest sites for Barn Owls. Ramsden (1998) found that the loss of a breeding site through barn conversion often resulted in Barn Owls abandoning an area, even though alternative breeding sites were available, thus having a disproportionately large negative effect on Barn Owls. Large, isolated and often dead trees the favoured natural nesting sites; in Suffolk dead pollarded elms are favoured, many of which are removed by farmers for safety reasons (Cayford 1992). Barn Owls will take readily to nest-boxes (Petty *et al.* 1994); the density of a population in Norfolk was raised from 15 to 27 pairs.km^{-2} by the provision of nest-boxes (Johnson 1994), indicating that the lack of suitable nesting sites was previously limiting the population.

During the summer Barn Owls generally forage within 2 km of the nest site (Cayford 1992). Territories may overlap, probably due to the clumping of suitable nest sites. Birds forage over open areas of permanent rank grassland, field margins, set-aside, drainage ditches and semi-natural and unmanaged grassland – all habitats with thick sward, deep litter layer and high densities of small mammals (see below). Habitats with very tall and dense vegetation are avoided (Tome & Valkarna 2001), as are arable crops (Cramp 1985, Cayford 1992). Shawyer (1987) declared wet grassland edges to be of "supreme importance", with Barn Owls being particularly abundant on farmland dissected by river valleys, preferring traditional water meadows or other wise foraging along dikes, ditches and over marshes and damp moorland edges.

Isolated patches of (drier) rough grassland are also favoured, including graveyards, disused railway embankments, building plots, poultry runs, disused airfields and roadside verges. The latter habitat is responsible for a high level of mortality from road-traffic collisions; Massemin & Zorn (1998) found that most Barn Owl deaths from traffic collisions in north-east France occurred along embanked highways crossing open fields without hedges. In Britain road deaths were estimated to contribute 15% of all mortality between 1955-1969 (Glue 1971), by 1982-85 this figure had reached 51.9% (Shawyer 1987).

Highest Barn Owl densities are found in mixed farmland, as a high degree of habitat heterogeneity provides a mosaic of foraging habitats and a wide variety of prey (Tome & Valkama 2001). Barn Owls are rare in areas where farming is predominantly of a single type (Shawyer 1987).

Barn Owl diet comprises mainly of small mammals, although other prey items are taken when available. A wide range of mammals are taken (see Cramp 1985) but in Britain (short-tailed) field voles *Microtus agrestis* are taken selectively (Glue 1967). Also taken are bank voles *Clethrionomys glareolus*, yellow-necked and wood mice *Apodemus flavicollis* and *A. sylvaticus* and common and pigmy shrews *Sorex araneus* and S. *minutus*. There has been a shift in Barn Owl diet in the last three decades, with a widespread decrease in the importance of common shrew, with instead more pygmy shrews, bank voles and *Apodemus* spp. being taken (Love *et al.* 2000). Other mammals recorded in Barn Owl diet by Love *et al.* (2000) included mole *Talpa europaea*, water shrew *Neomys fodiens*, harvest mouse *Micromys minutus*, house mouse *Mus domesticus* and brown rat *Rattus norvegicus*. Foraging habitat strongly influences the prey taken with, for example, birds foraging around farmsteads taking a higher than average proportion of house mice and those foraging over potato crops taking more brown rats (Cooke *et al.* 1996).

Other prey groups include birds, reptiles, frogs and even fish, all of which typically provide a negligible contribution to total diet (Cramp 1985).

Wintering requirements

Most adult Barn Owls remain on the breeding territory throughout the winter, often remaining with their breeding partner. Sites suitable for nesting in the summer are also used as winter roosts, although trees are often used as roosts even in areas where buildings are used for nesting. There may be some movement, particularly in severe weather when birds in hilly regions may move to lower altitudes (Lack 1986). There is more movement from first-year birds dispersing from the natal site, although few birds disperse more than a few km. The hunting range may increase in the winter, with birds foraging up to 4 km from the roost site.

As fat reserves in healthy Barn Owls are as low as 5.4-5.6% of total body weight (Honer 1963) they are susceptible to starvation during periods of low food availability. There are growing concerns regarding the effects of second generation anticoagulant mobilised from stored fat during periods of food stress (Burn *et al.* 2002). Britain is at the northern limit of the world range of Barn Owl, and they are absent from areas of the country that typically experience harsh winter conditions. Snow for extended periods can result in prey being unavailable, leading to starvation. During such periods Barn Owls may hunt closer to farmsteads or move to coastal regions (Lack 1986). Shawyer (1987) found a close relationship between the abundance of Barn Owls in Britain and the snowcover in the preceding winter. Winters when snowcover reached or exceeded 20 days duration (using a mean from eight meteorological stations around Britain) were followed by an immediate crash in the Barn Owl population. However, this decline was less due to direct mortality (which only increases when snowcover exceeds 35 days) but to declines in vole populations caused by the severe weather.

As during the breeding season, small mammals provide the large part of Barn Owl diet in the winter. There is some variation in the diet, both between birds in different regions and between individual birds (Barn Owls show marked individual preferences in both foraging habitat and prey type, e.g. Cayford 1992) Generally field voles are taken more frequently during the winter, reflecting their greater availability in this season (Love *et al.* 2000).

Skylark *Alauda arvensis*

Skylarks breed in most open habitats, and hence are found in farmland throughout Britain. However, there has been a very rapid decrease in Skylark density since the mid-1970s (population levels declined by 60% between 1972 and 1996, Crick *et al.* 1997), although this decline has not been accompanied by a decrease in range (Gibbons *et al.* 1993). A decline in populations has also been recorded in mainland Europe (Tucker & Heath 1994). Because of this decline Skylark is on the red list of Birds of Conservation Concern (Gibbons *et al.* 1996) and a UK BAP Priority Species.

Breeding season requirements

Upland and semi-natural grassland habitats hold the highest densities of Skylarks in the breeding season (see below), but within the intensively farmed agricultural landscape densities are greatest in arable farmland, intermediate in mixed and lowest in pastoral (Chamberlain & Gregory 1999). Skylark density is positively correlated with diversity of field types, which suggests that simplification of farm rotations may be one cause of the decrease in Skylark density. Skylark reproductive performance per nesting attempt has increased in recent years (Chamberlain & Crick 1999), so this can be disregarded as the cause of declines. However, a decrease in the number of nesting attempts through the breeding season (Skylarks usually attempt to breed two or three times, Cramp 1988) could be a major cause of the decline. Chamberlain & Gregory (1999) found that densities of Skylarks on arable land decline through the breeding season. Although winter cereals may provide suitable nesting habitat for Skylarks in the early breeding season, their rapid growth means that they are too dense and high later in the season (Evans *et al.* 1995, Wilson *et al.* 1997b).

In the uplands, Skylarks favour short graminoid cover over moorland, and so would actually seem to benefit from the historical increases in chronic grazing pressure (Pearce-Higgins & Grant 2002). However, at greater resolution, Skylark numbers increase with increased spatial heterogeneity in grass biomass (Pearce-Higgins & Grant 2002). Therefore, as with Lapwings, variation in sward structure may be beneficial, presumably because of the ability to provide both nest sites and feeding sites, and the increased protection from predators afforded by increased habitat complexity.

Skylark diet in the breeding season consists of a wide range of invertebrate taxa. Important amongst these are spiders (Araneae), bugs (Heteroptera), aphids (Aphididae), craneflies (Tipulidae) (both adults and larvae), soldier-flies (Stratiomyidae), hover-flies (Syrphidae), hymenopteran larvae (Symphyta), ground beetles (Carabidae), weevils (Curculionidae), leaf beetles (Chrysomelidae) and click beetles (Elateriidae). Nestling diet consists of soft-bodied invertebrates such as spiders, aphids and larvae in the first few days, with chitinous prey becoming progressively more important as nestlings grow (Poulsen 1993, Wilson *et al.* 1997a).

A study of foraging behaviour in the breeding season showed that Skylarks foraged in set-aside, grass tracks, field margins and organically-managed silage while avoiding arable crops, grazed pasture and conventionally managed silage. Of the winter cereals, wheat was preferred over barley (Wilson 2001). Preference for particular habitats for foraging is probably due to a combination of invertebrate abundance and patchy vegetation cover that allows ease of access. The dense sward structure of winter cereals prohibits easy access to food. Odderskaer *et al.* (1997) found that Skylarks nesting in cereal crops spent over 50% of foraging time in tracks despite this habitat covering a small proportion of the total agricultural landscape.

Wintering requirements

Skylarks winter mainly on farmland and coastal habitats, with the highest densities being recorded on saltmarsh (Gillings & Fuller 2001). The majority of the wintering population is found on lowland farmland. Studies have shown that Skylarks either avoid winter cereals (Gillings & Fuller 2001) or use them in proportion to their occurrence (Donald *et al.* 2001), although this usage is greater than that by other granivorous passerines (Wilson *et al.* 1996). Grassland is also avoided (Donald *et al.* 2001, Gillings & Fuller 2001). Like other farmland granivorous passerines Skylarks prefer to feed upon winter stubbles, with set-aside and broad-leaved crops also selected (Donald *et al.* 2001). Barley stubbles (which have a higher proportion of weed species) are preferred to wheat stubbles, whilst among set-aside; rotational set-aside fields hold more Skylarks than non-rotational (Buckingham *et al.* 1999, Donald *et al.* 2001). Skylark numbers are positively correlated with soil seed density (Robinson & Sutherland 1997, Wakeham-Dawson & Aebischer 1998). Declines in soil seed density mean that Skylarks have to forage for a longer period of the day (Robinson 1997), which may have an effect on their susceptibility to predation (Robinson & Sutherland 1997).

As in the breeding season, wintering Skylarks prefer large fields with a low boundary to area ratio and without tall hedgerows or woodland bordering the fields (Robinson 1997, Donald *et al.* 2001, Gillings & Fuller 2001). This dislike for boundaries is probably as a precaution against predation.

The diet of Skylarks in the winter varies significantly between the different foraging habitats utilised. Birds on cereal stubbles and cereals feed largely on cereal products, either grain or growing leaves. Birds in grassland feed on cereal leaves, those in broad-leaved crops on broad-leaved leaves (both crop and weeds) and those in bare tillage and set-aside have more mixed diets not dominated by any one food type (Donald *et al.* 2001). Despite higher food availability on organic farms Skylark densities are not higher on such farms (Chamberlain *et al.* 1999); this may be due to the smaller field sizes and higher field boundaries found on such farms. Skylarks take few invertebrates in any habitat. The selection of winter stubbles as the preferred foraging habitat is probably linked to the availability of cereal grain, the most energetically profitable winter food for Skylarks (Green 1978).

Yellow Wagtail ***Motacilla flava***

The Yellow Wagtail is a summer visitor to Britain, wintering in sub-Saharan Africa (Cramp 1988) and returning to Britain in April. It is widely distributed as far north as southern Scotland, with few birds in south-west England or Wales, and concentrations along the coastal fringe of south-east England and in the east Midlands. The range had contracted from western areas by 1968-72 (Sharrock 1976) and this contraction has continued, with birds disappearing from parts of coastal south England, farmland in Dorset, Hampshire and Wiltshire.

Breeding season requirements

Yellow Wagtails are associated with water, but do not require running or open water in their breeding habitat. Preferred habitats are along river valleys, especially in the lower reaches, in water meadows and damp fields. Fifty-two percent of nests are close to water, and 67% in grassland (Mason & Lyczynski 1980). Nearly all Yellow Wagtails nest on the ground, and avoid dense and rank vegetation. Mason & Lyczynski found that only 12% of Yellow Wagtail nests recorded by the BTO nest record card scheme were on farmland. However, there has been a recent trend towards breeding in farmland in drier habitats, possibly due to the loss of damp riverside habitats. In farmland, Yellow Wagtails nest in field centres, preferring large fields (Mason & Macdonald 2000). Seventy-three percent of Yellow Wagtails found, by a study in Essex were in spring-sown crops, with potatoes being particularly favoured (and, to a lesser extent, legumes) and autumn-sown crops and grassland avoided (Mason & Macdonald 2000, Nelson 2001). Lack (1992) also noted a preference for root crops by Yellow Wagtails nesting on farmland. Nelson (2001) found that nests in crops were located in areas of low crop height and density, and that the few birds that bred in winter cereals nested near tramlines. Yellow Wagtails prefer to feed in open habitats with low vegetation and so nest in or near these habitats, although they will travel up to 1 km to forage.

The date of grass cutting may be important for birds nesting in grass grown for hay or silage. Wilson (1991) found that up to 33% of nests in a study site in the Yorkshire Dales failed due to grass cutting before nestlings had fledged, and recommended that delaying cutting by one or two weeks would have enabled this problem to have been avoided. Similarly, ADAS (1995) found that 25% of 83 nests studied in the Pennine Dales ESA were lost to cutting operations in June & July.

The diet of both adults and chicks consists almost entirely of small invertebrates (Davies 1977). In the early part of the breeding season (shortly after arrival from Africa) Yellow Wagtails in Oxfordshire took almost all Diptera, and 86% of the diet consisted of midges (Chironomidae). By May this proportion had dropped to 35%, with a further 44% of the diet consisting of Drosophilidae (Davies 1977). Diptera and Nematocera are important prey throughout the breeding season, as are spiders (Araneae) (Gibbons *et al.* 1993).

Nestlings are fed mainly on aphids (Homoptera) and a variety of Diptera, including blowflies (Calliphora), robber-flies (Asilidae), crane-flies (Tipulidae), dung flies (*Scathophaga stercoraria*, amongst others) and house-flies (Muscidae) and weevils (Curculionidae). Nelson (2001) found that damselflies (Zygoptera), Diptera and Coleoptera were the main food for nestlings in arable land in Lincolnshire.

Whitethroat *Sylvia communis*

As a summer visitor to this country, a large proportion of the Whitethroat life history lies without Britain. It may well be that factors on the wintering grounds (or during migration) determine British population levels. This was dramatically illustrated in 1969 when fully three-quarters of the breeding population failed to return from the sub-Saharan wintering grounds due to drought there (Winstanley *et al.* 1974). Since that decline numbers have gradually increased, but still remain well below pre-1969 levels.

Whitethroats are present throughout most of England and Wales, although scarce or absent in upland areas. In Scotland most are present in the south and east, with birds further north concentrated in the lowland coast fringes.

Breeding season requirements

Whitethroats are particular in their choice of breeding habitat, preferring scrub habitats at a particular stage of succession (Fuller 1982). Therefore they breed in young woodlands, recently coppiced woodland, open scrubland and hedgerows. A study of BTO nest record cards (Mason 1976) showed that 56% of Whitethroat nests reported were in scrubland, with only 18% in hedgerows. However, hedgerows are considered to be an important nesting habitat for Whitethroats and the loss of hedgerows due to agricultural intensification may have been a contributing factor to declines, or rather may have slowed the recovery from crashes caused by conditions on the wintering grounds. Evidence that this may be the case is provided by the slower recovery of breeding numbers on farmland than in riparian habitats where there has been less habitat change (Crick *et al.* 1997).

Stoate & Szczur (2001) found that the presence of Whitethroats on farmland hedges could be predicted by the height of the hedge and the width of the uncropped margin adjoining the hedge, with low hedges and wide margins increasing the likelihood of Whitethroat presence. Whitethroats usually nest in herbaceous vegetation alongside hedges, rather than in the hedges themselves, so the presence of such uncropped vegetation may be as important as the presence of a hedge. Whitethroat densities are higher on arable land, and it may be that this is due to the effect of grazing depressing herbaceous growth adjacent to hedges. Over half of Whitethroat nests recorded by the BTO nest record card scheme were in bramble *Rubus* (Rosaceae) or nettle *Urtica* (Urticaceae); grass is also used freely (Mason 1976). Nests are usually below 1 m from the ground (Persson 1971).

The diet of Whitethroats consists primarily of insects, although in late summer both adults and first-year birds may take berries e.g. bramble *Rubus* (Rosaceae), elder *Sambucus* (Caprifoliaceae), buckthorn *Rhamnus catharticus* (Rhamnaceae). The range of invertebrates taken during the breeding season is wide, but several groups predominate, notably Lepidoptera, Coleoptera, Diptera, Hymenoptera and Hemiptera. More than half the diet is Lepidopteron, both adults and caterpillars, most of which are gathered within 30 m of the nest (Stoate *et al.* 1998). Other major prey items include sawfly larvae (Symphyta), weevils (Curculionidae), ground beetles (Carabidae) crane-flies (Tipulidae) and spiders (Araneae). Diet may change through the breeding season with, for example, carabids being important in the early part of the breeding season and Hymenoptera such as ants (Formacidae) more so later in the season (Cramp 1992). Nestling diet contains a higher proportion of soft-bodied prey, in particular caterpillars and spiders (Macdonald 1979, Moreby & Stoate 2001).

Jackdaw *Corvus monedula*

Jackdaws are present throughout most of Britain, being absent only from north-west Scotland. This absence is related to a preference for lowland areas; Sitters (1988) found that few nested above 350 m (and none above 450 m) in Devon. Although there have been declines and range losses in a few areas (Gibbons *et al.* 1993), there has been a steady population decline over the second half of the 20th century, with Common Bird Census data indicating an increase of 80% between 1970 and 1998 (Gregory *et al.* 2000). The most recent estimate of the British population was 390,000 territories (Gibbons *et al.* 1993); numbers may have increased since that estimate.

Breeding season requirements

The vast majority of Jackdaw nests are in cavities, either in tree holes, rock crevices or in buildings (commonly in chimneys), although domed nests built of stick have been recorded (Gibbons *et al.* 1993). Due to this requirement Jackdaws are often found clustered in loose colonies at sites where multiple suitable nest-sites are available; a study in Oxfordshire found no preference for clumped or solitary nest-boxes (Heeb 1991). Jackdaws territories consist of the nest and immediate vicinity only. Often colonies are found in free-standing or grouped mature trees, small copses and avenues rather than in woodland. As Jackdaws will travel considerable distances to forage (Cramp & Perrins 1994) the habitat in the immediate vicinity of the nest is not of crucial importance, hence jackdaws can be found breeding in a wide variety of habitats where suitable nesting sites are available. However, they prefer semi-open habitats and show a marked preference for foraging on grazed grasslands during the breeding season, although swards above 15-20 cm high are avoided (Strebel 1991). Therefore Jackdaws are commonest in areas of pasture or mixed farmland – a decline in East Anglia in the 1960s may have been due to the conversion of grass leys to permanent arable (Tapper 1981).

Diet in the breeding season comprises of a wide range of invertebrates, taken mainly on the ground. Unlike Rooks, Jackdaws tend not to probe for soil-dwelling invertebrates but pick up surface- and sward-dwellers, as well as foraging in and underneath dung (Lockie 1956). Prey includes weevils (Curculionidae), ground beetles (Carabidae), ants (Formacidae), spiders (Araneae), grasshoppers (Orthoptera) and many more (see Cramp & Perrins 1995). Caterpillars (Lepidoptera) may be foraged for in tree foliage (Coombs 1978). In addition, plant matter may be important, with cereal grain (from feeding troughs if not available in fields), weed seeds, tree seeds and fruit are all taken if available. Jackdaws have been recorded feeding on a long and varied list of other food resources ranging from the pirated contents of milk-bottles to bats, eggs and both chicks and adult birds (Cramp & Perrins 1994).

Nestling diet is predominantly invertebrate; a study in Wales found that Lepidoptera larvae, Coleoptera and Diptera together contributed nearly 70% of total nestling diet (Richford 1978).

Wintering requirements

There is some withdrawal from upland regions, but otherwise the distribution of Jackdaw is the same in the winter as for summer (Lack 1986). As in the summer, grassland is favoured habitat, but Jackdaws are also commonly found in stubbles and freshly ploughed ground.

Fifty-eight percent of observations of foraging Jackdaws made by Waite (1984) were on grassland and stubble. Jackdaws will also feed on rubbish tips more readily than other Corvids and are a familiar site in towns, parks and gardens scavenging on any available food (Lack 1986).

Although surface dwelling invertebrates such as weevils remain important throughout the winter, the proportion of plant matter in the diet increases during the winter. Grain can be very important, providing the main part of the diet in some regions (Holyoak 1968). Other crops such as peas, beans and root crops are also foraged upon when available. Fruits are important in the autumn; weed seeds may be so later in the winter (Cramp & Perrins 1994). As in the breeding season, Jackdaws are opportunistic feeders and will take a wide variety of food types.

Jackdaws generally gather in central roosts in the winter (usually in woods or copses) that they may travel considerable distances to attend, often in the company of Rooks.

Rook *Corvus frugilegus*

Rooks are found in all lowland farmland regions of Britain, being only absent from upland areas such as north and west Scotland, parts of Wales and the Pennines. They breed up to 450 m in altitude in England (Yapp 1962). They are resident, with little evidence of any movements between summer and winter. The most recent Rook survey estimated there were 1,270,000 pairs of Rooks in the United Kingdom in 1996 (Marchant & Gregory 1999), which was an increase of 40% from 1975-77. This 1975-77 level was 43% less than that in 1943-46 (Sage & Vernon 1978).

Breeding season requirements

Rooks nest colonially in rookeries, averaging 30 pairs per rookery. Rookeries are in the tops of fairly tall trees, either on woodland edges or preferably in clumps or lines – treelines and farmland copses are particularly favoured. In lowland areas deciduous trees are preferred (70% of rookeries in England are in deciduous trees, Marchant & Gregory 1999), with oak *Quercus* (Fagaceae), sycamore *Acer pseudoplatanus* (Aceraceae) and ash *Fraxinus excelsior* (Oleaceae) being favoured species.

There is some disagreement between studies in the relationship between Rook numbers and agricultural land management. All studies agree that numbers of Rooks increase with an increasing proportion of grassland in the landscape: Griffin (1999) found that Rook breeding density with 5 km grid squares in County Durham was related to the area of grassland. Lomas (1968) found that declines in Rooks numbers in the 1960s were greatest in areas of cereal growing. However, Brenchley (1984) found that Rook numbers increased as the proportion of grassland increased up to 55% of the total agricultural area, but decreased at levels thereafter: the optimum balance of land-use for Rooks was 55% grass and 45% arable. This suggests that there could be some requirement for arable as well as grassland, and that mixed farmland would provide the best habitat for Rooks. Both Chater (1996) and Griffin (1999) found that Rook density was positively correlated with (sheep) stocking density, probably because shorter swards allowed greater access to soil invertebrates for Rook feeding on grassland.

The diet of Rooks during the breeding season is varied, but earthworms (*Lumbricus* and *Allolobophora* spp., Lumbricidae) and leatherjackets (Tipulid larvae, Tipulidae) are the most important prey (hence grassland is the most important habitat for foraging during the breeding season, Feare *et al.* 1974). Rooks breed earlier than other lowland Corvids, and it is possible that this is so nestlings can be fed earthworms before drying ground results in earthworms becoming less accessible. A wide variety of other invertebrates are taken, including weevils (Curculionidae), dung beetles (Scarabaeidae and Hydrophilidae), ground beetles (Carabidae), Lepidoptera (both adults and caterpillars), spiders (Araneae), rove beetles (Staphylinidae) and ants (Formacidae) (Holyoak 1972, Cramp & Perrins 1994) with surface dwelling invertebrates becoming more important as the breeding season progresses. Nestling and fledgling Rooks feed almost entirely on invertebrates, but adults do feed on grass leaves, potatoes and other root crops, legumes and weed seeds (such as bistorts *Polygonum* (Polygonaceae), violets *Viola* (Violaceae) and buttercups *Ranunculus* (Ranunculaceae)). In addition Rooks will forage on carrion, human waste, spilt food at piggeries and other ephemeral food resources. Rooks characteristically glean recently-harvested silage fields, presumably taking damaged or exposed invertebrates.

Unlike most resident farmland species, Rooks have greater difficulty obtaining sufficient food in the summer than in the winter, and starvation is a real threat later in the summer (Dunnet & Patterson 1968). This mortality is particularly marked on juvenile birds, and can be higher in hot dry years when soil invertebrates are less accessible.

Wintering requirements

It is thought that Rooks rarely suffer hardship in the winter (Feare *et al.* 1974). The winter diet, as in the summer, is varied, but still contains a high proportion of invertebrates as listed above. Earthworms are particularly important prey. House-flies (Muscidae) are a surprisingly common winter food, probably as Rooks forage on rubbish dumps and manure heaps where house-flies are present throughout the year. In addition to invertebrates, plant food is much more important to Rooks outside of the breeding season. Cereal grain is taken from standing crops at the end of the summer, and then from stubbles and finally from autumn sowings. Root crops are also commonly foraged upon.

Carrion can be an important component of diet, particularly in winter, and the increase in animal road casualties due to more road traffic has been suggested as one factor contributing to the recent increase in Rook numbers (Marchant & Gregory 1999).

A notable feature is their use of traditional roost sites, which thousands of birds may gather at, along with other Corvids. Rooks arrive at these gatherings along set flight-routes, often with pre-roost gatherings on the way, and may travel upwards of 20 km to roost. Roost-gatherings usually consist of birds from all the rookeries within the "catchment area" of the roost. Somewhat strangely, birds from a particular rookery may travel long distances to a communal roost site despite one being present considerably nearer the feeding area (Griffin 1999).

Starling *Sturna vulgaris*

One of the most widespread and numerous of all British bird species, Starlings are present in almost all habitats apart from moorland and mountains. Therefore they are found throughout Britain apart from areas in north-west Scotland and smaller gaps in Wales and northern England (Gibbons *et al.* 1993). However, populations have been in steep decline recently

(Gibbons *et al.* 1993) and the Starling is to be redesignated to the red list of Birds of Conservation Concern (it was formerly amber-listed, Gibbons *et al.* 1996).

Breeding season requirements

Starlings are hole-nesters, and so require suitable holes or crevices for breeding. Such nesting sites may be found in buildings, trees or nestboxes. Birds will nest in other situations, such as inside old nests of Carrion Crows *Corvus corone* (amongst others), but those in holes lay larger clutches and have higher breeding success (Feare 1984). The large-scale loss of hedgerows and hedgerow trees could mean that the availability of nesting sites limits the abundance of Starlings on farmland.

Starlings show a very strong preference for foraging on grassland, taking invertebrates from the ground surface and probing in the first few centimetres of soil. They prefer short grass, presumably because it allows easy access to the ground surface and soil (Brownsmith 1977). They also show a liking for foraging on freshly cut grass (Tinbergen 1981).

Starlings take an extremely wide variety of foods and are able to change diet to take advantage of ephemeral resources, such as waste food material from humans. However, they are primarily invertebrate feeders, taking larvae of Lepidoptera, Diptera, Hymenoptera and Coleoptera, particularly the larvae and pupae of craneflies, especially *Tipula paludosa* (Tipulidae), march-flies (Bibionidae), ants (Formicidae), ground beetles (Carabidae) and snipe-flies (Rhagionidae). Adult insects are also taken, including ground beetles, weevils (Curculionidae), rove beetles (Stapylinidae), spiders (Araneae) and earwigs (Dermaptera). Little plant matter is taken in the summer, although in the late summer birds begin to feed on fruit. This is particularly true of juveniles, which in general are more arboreal than adults after fledging (Taitt 1973).

Nestling diet tends to be far more restricted than that taken by adults, and in Britain is dominated by cranefly larvae (leatherjackets) (Whitehead 1994).

Wintering requirements

The British population of Starlings is swelled in the winter by a huge number of immigrants from the continent. These birds gather in huge (and often well-known) roosts in conifer plantations, evergreen shrubberies and reedbeds as well as in urban areas roosting on buildings, bridges and other man-made structures, in order to minimise predation risk and to gain energetic advantages. Birds may travel up to 38 km to attend roost sites (Feare 1984).

Outside of the breeding season the reduced availability of invertebrate means that Starlings take a wider range of plant food, although they still forage for buried larvae (especially tipulids) and earthworms, with short turf remaining the preferred foraging habitat. Fruit eaten include yew *Taxas baccata*, ivy *Hedera helix*, hawthorn *Crataegus* spp., elder *Sambucus* spp. and especially dogwood *Thelycrania sanguinea* (Snow & Snow 1988). Fallen fruit waste in orchards is also taken. Cereal grain may also be important if invertebrates are scarce, as are weed seeds (from the ground). As mentioned before, the opportunistic nature of Starlings means that they frequently forage in gardens, both rural and urban, and on rubbish tips and other places where human waste is accessible. Starlings can adapt physiologically rapidly in response to digestive challenges of new food resources (Al-Joborne 1979).

Tree Sparrow *Passer montanus*

Tree Sparrows are a tree nesting species found in hedgerows, parkland and open woodland, although not in dense forests. Despite the scientific name it is a lowland species, commonest in a band across mid-England from East Anglia into east Wales. It is scarce along the south coast away from Kent, absent from the south-west, and in the north confined mainly to the east (Gibbons *et al.* 1993). Numbers of Tree Sparrows in Britain have fluctuated since at least 1860 (Summers-Smith 1989, Holloway 1997), with a high population from the 1880s to the 1930s followed by a decline, and then another peak during 1960-1978. Since 1978 numbers have again declined, leading to the Tree Sparrow being red-listed as a Bird of Conservation Concern (Gibbons *et al.* 1996) and a UK BAP Priority Species.

Breeding season requirements

Tree Sparrows are associated with trees, but not with dense woodland – they are found in farmland with tree-lines or hedges with isolated trees as well as in parkland and open woodland. Tree sparrows nest in loose colonies of up to 50 pairs, spread over between 1 and 30 ha – the proximity of nests is determined by the availability of nest sites. Although colonies are usually loosely clustered rather than densely packed, if suitable nesting sites are available in close proximity they will be used as Tree Sparrows show no territoriality outside of the nest (Summers-Smith 1995). As hole nesters, Tree Sparrows may be limited by the availability of nesting holes. Tree holes are the most frequently used location, but buildings are also used, as are stone walls. Small holes of approximately 30 mm are preferred (Löhrl 1978). In the absence of holes, Tree Sparrows may nest within large stick nests i.e. of Grey Herons *Ardea cinerea* and Carrion Crows *Corvus corone* (Harrison *et al.* 1982).

Adult Tree Sparrows take a mixed diet in the breeding season, foraging on the ground for both invertebrates and weed seeds. Fat hen *Chenopodium album* (Chenopodiaceae) and knotgrass *Polygonum aviculare* (Polygonaceae) are favoured weed species, along with other Polygonum spp., common amaranth *Amaranthus retroflexus* (Amaranthaceae), chickweed *Stellaria* (Caryophyllaceae), forget-me-not *Myosotis* (Boraginaceae) and grasses (Gramineae) (Grün 1975). Cereal grain becomes more important later in the breeding season, with oats especially preferred (Keil 1973).

Invertebrate prey types include Lepidoptera, Hemiptera, Diptera, Coleoptera and Orthoptera. Within these orders favourites include spiders (Araneae), aphids (Aphididiae), grasshoppers (Acrididae), crickets (Gryllidae), bush crickets (Tettigoniidae) and weevils (Curculionidae) (Wilson *et al.* 1997a). Recent research at Rutland Water (G. Anderson pers. comm.) suggests that the edges of aquatic habitats are important foraging sites for insect food (especially for the larvae of aquatic insects).

Nestling diet is almost completely invertebrate. Younger birds are fed spiders, aphids and small larvae, while older birds get larger chitinous adult insects.

Wintering requirements

Tree Sparrows remain near the breeding colony in the winter, rarely straying more than a few kilometres, although they may join with birds from neighbouring colonies in feeding flocks. Most birds feed in arable farmland, taking cereals (principally in the autumn) and weed seeds (see above for details), which can contribute up to 90% of the diet in the late winter (Keil 1973).

Linnet *Cardeulis cannabina*

Linnets are widespread throughout England and Wales, although scarcer in upland areas. The range extends into southern Scotland, but further north Linnets are restricted to the coastal fringe (Gibbons *et al.* 1993). This predominance around the coastline is also exhibited in east and south England. Linnets are commonly associated with Gorse *Ulex* (Leguminosae), but are found in other habitats that provide dense habitat. Seventy percent of Linnets in the UK nest on farmland (Gregory & Baillie 1998).

Linnets declined by 41% in the UK between 1968 and 1995 (Siriwardena *et al.* 1998). This decline was actually concentrated between the mid-1970s and 1987, and there has been a slight increase since (Moorcroft & Wilson 2000). As a subsequence of this decline Linnet is on the red list of Birds of Conservation Concern (Gibbons *et al.* 1996) and a UK BAP Priority Species.

Breeding season requirements

Linnets breed semi-colonially, forming the largest nesting groups of any of the *Carduelis* species (Tast 1970), although they will breed in isolation (it is postulated that this may be caused by low levels of available nesting habitat, Frey 1989). Nests are located in hedgerows, dense shrub or isolated bushes. As Linnets nest colonially they do not defend territories but rather defend their mate to insure paternity (Drachmann, Komdeur & Boomsma 2000). Breeding birds feed communally in areas of abundant food, travelling up to 2 km from the nest site to feed. This wide-ranging foraging behaviour means that habitats in the immediate vicinity do not necessarily determine the location of nest sites.

Linnets are unusual amongst finches in that they feed almost entirely upon seeds – even nestling diet only contains a small proportion of aphids and a few small caterpillars (Wilson *et al.* 1997a). They also tend to avoid berries and fruits and rarely take tree seeds. Traditionally the main food items of Linnets have been weed seeds, including those of bistorts and docks (Polygonaceae), chickweeds and mouse-ears (Caryophyllaceae), Brassicas such as charlock *Sinapis arvensis* and oilseed rape, a wide variety of composites (Compositae) such as dandelion *Taraxacum*, thistles *Cirsium*, groundsel *Senecio* and sow-thistles *Sonchus* and some grasses (Gramineae) (Wilson *et al.* 1997a). Oilseed rape was absent from Linnet diet in 1962-64 (Newton 1967) but by 1996 was dominant, with only dandelions and sow-thistles more important (Moorcroft *et al.* 1997). Many other weed species have declined in recent decades: the increase of oilseed rape as a crop has probably softened the impact of these declines on the Linnet population. Weed species on grassland have not declined to the extent of those in arable farming systems, indeed some species such as dandelions *Taraxacum* (Compositae) have increased in abundance, as they are tolerant of both grassland "improvement" and heavy grazing pressure (Grime *et al.* 1988).

Wintering requirements

The winter distribution of Linnets in Britain is rather similar to that in the summer, although altitudinal migration means that they are largely absent from upland regions (Lack 1986). A proportion of the population migrates south to continental Europe to winter; although this has not been precisely quantified, Lack (1986) gives an estimate of half of the population emigrating. Most Linnets concentrate on arable farmland, although they are also to be found on waste ground and on coastal habitats such as saltmarshes. They gather in small flocks, although these can number up to 200 or even higher, and large roost gatherings are common (Newton 1972). Thick scrub such as hawthorn *Crataegus* (Rosaceae) or gorse *Ulex* (Leguminosae) is selected for roosting. Diet consists entirely of seeds during the non-breeding season, with grasses, thistles, dandelions and bistorts and docks being the most important components.

Wilson *et al.* (1996) found that Linnets showed a very strong preference for foraging in stubble fields in the winter, while Moorcroft *et al.* (in press) demonstrated that they select for stubble fields with high weed seed density. Linnets were rarely found in fields where the density of weed seeds important in their diet fell below $250m^{-2}$. In addition, within selected fields Linnets prefer to forage in areas of high seed density and with a greater proportion of bare earth (Moorcroft *et al.* in press).

Goldfinch *Cardeulis carduelis*

The Goldfinch is a widespread bird, found throughout England and Wales and most of Scotland. The range has spread northward for several decades (Gibbons *et al.* 1993). Although Goldfinches are present in Britain throughout the year, the majority (maybe as high as 80%, Newton 1972) migrates south in September and October to winter in Belgium, France and Iberia.

Breeding season requirements

Goldfinches nest in a variety of lowland habitats, mainly those that are open with scattered shrubs or trees. Woodland edges, parkland and gardens are used, but the large part of the population is found breeding on farmland. Shrubs and trees are used for nesting; as nests are usually 4-10 m above the ground short, intensively managed hedges are not suitable for breeding in (Gibbons *et al.* 1993). Like other Cardueline finches, Goldfinches will nest in loose colonies.

The diet of adult Goldfinches is dominated by Compositae (see below), with few wild grasses or cultivated cereals. Therefore Goldfinches require sufficient weedy areas, either within crops, along margins or in other habitats such as rough ground or set-aside. These do not have to be in the immediate vicinity of the nest, as Goldfinches will range widely to forage (Newton 1972). Little is known of the diet of Goldfinch nestlings, but it is thought to contain invertebrates such as caterpillars, aphids (Aphidididae), small flies (e.g. Muscidae), ants (Formicidae), small beetles and their larvae (e.g. Curculionidae, Chrysomelidae, Coccinellidae, Elateridae) and parasitic wasps (e.g. Braconidae) as well as regurgitated seeds, presumably similar to those eaten by the parent birds (Wilson *et al.* 1997a).

Wintering requirements

As stated previously, most Goldfinches migrate south from Britain in the autumn, returning in April. However, birds remain throughout the range through the winter, with some withdrawal from higher ground (Lack 1986). As in the breeding season, Goldfinches are reliant on a diet of weed seeds, with a marked preference for Compositae when available. These include thistles *Cardius* and *Cirsium*, dandelions *Taraxacum*, groundsels and ragworts

Senecio, Knapweeds *Centaurea* and burdocks *Arctium*. Thistles provide a third of the annual diet (Newton 1967). Later in the winter Goldfinches will also feed on the seeds of birches *Betula*, alders *Alnus* and Teasel *Dipsacus fullonum* as resources such as dandelions become depleted. This change in diet in the late winter, and the departure of the large part of the population, suggests that winter food resources may be limiting for Goldfinches in Britain (Lack 1986).

Greenfinch *Carduelis chloris*

Greenfinches are widespread and common, being absent only from treeless upland regions of Britain. Unlike the other finches described above, Greenfinches are chiefly resident in Britain, with a December population estimated to be as high as 5-6 million (Lack 1986). Numbers are believed to have been relatively stable since the mid 1960s (Gibbons *et al.* 1993).

Breeding season requirements

Although nesting in trees and in part reliant on their seeds as food, Greenfinches do not nest in densely wooded areas but instead prefer a mosaic of habitats including tall trees. Therefore they are found in a wide variety of habitats but usually those with a mix of trees and shrubs and open areas; farmland, parks and suburban gardens are favoured habitats. Greenfinches have adapted well to the influence of man both on agricultural land and in suburban and urban areas (Newton 1972).

Nests are placed in small trees or large shrubs. Unlike some other Cardueline finches, Greenfinches do not forage in flocks in the breeding season, and usually do not forage long distances away from the nest.

The large bill of the Greenfinch means it is able to take a wide variety of seed types. The most important dietary items are probably tree seeds such as from spruce *Picea* (Pinaceae) and elm *Ulnus* (Ulmaceae). Also taken are the seeds of weeds such as Chenopodiaceae, dicks and bistorts (Polygonaceae), chickweeds Stellaria (Caryophyllaceae), wild and cultivated Brassicas (Cruciferae), groundsel *Senecio*, burdocks *Arctium* and Dandelions *Taraxacum* (Compositae).

Nestling diet is similar to that of parents, but contains a slightly higher incidence of invertebrates, especially aphids and caterpillars (Wilson *et al.* 1997). There is some evidence that oilseed rape may have become more important for brood rearing, possibly compensating for declines in weed seeds are invertebrates (R. Bradbury, pers. comm.).

Wintering requirements

The feeding habitats of Greenfinches become more generalised in the winter, with birds staying in the breeding habitats but also dispersing to rough and open ground along the coast, and on arable farmland (Lack 1986). There is a slight retreat from upland regions, and a low level of emigration.

In wood and parkland yew *Taxas buccata* (Taxaceae), hornbeam *Carpinus betulus* (Corylaceae), whitebeam *Sorbus*, rose *Rosa* and bramble *Rubus* (Rosaceae). In arable land

Greenfinches feed upon a wide variety of weed seeds, but particularly upon charlock *Sinapsis arvensis* (Cruciferae) and Persicaria *Polygonum* (Polygonaceae)

Greenfinches have shown an increasing use of bird tables (Lack 1992). This reliance on peanuts and other provided foods such as sunflower seeds are probably responsible for maintaining the population at its current level, especially in late winter and early spring when supplies of naturally occurring seeds become depleted. Greenfinches have become less abundant on farmland, where favoured weed species have declined, instead wintering around towns and villages.

Reed Bunting *Emberiza schoeniclus*

Reed Buntings are widespread and resident through out Britain although, like many farmland bird species considered in this review, they are absent from upland areas. In recent decades there has been a slight contraction of range, predominantly in the north and west (Gibbons *et al.* 1993). Reed Buntings increased in numbers between 1963 and 1975, expanding into less preferred farmland habitats as numbers increased. However, this increase was followed by a steep decline in numbers between 1975 and 1983. During this period Reed Buntings declined by 58% on farmland and by 66% along linear waterways (Peach *et al.* 1999) which led to Reed Bunting being placed on the red list of Birds of Conservation Concern (Gibbons *et al.* 1996) and a UK BAP Priority Species. Population levels have remained relatively stable since this decline.

Breeding season requirements

Reed Buntings are found within a wide variety of habitats and, although commonly associated with wet or damp areas, 50% of British Reed Buntings breed on farmland (Gregory & Baillie 1998). A further 38% nest in semi-natural grassland, heath and scrub with only 12% in the preferred wetland habitats due to the relative scarcity of these habitats. Although breeding in farmland, most Reed Buntings will have damp or aquatic habitat within the territory, along with suitable dense ground cover for nesting in and song-posts for territorial establishment and defence (Gordon 1972).

The diet during the breeding season consists primarily of invertebrates (100% in May, Prys-Jones 1977). Precisely which prey is taken varies through the breeding season. Springtails (Collembola) and midges *Chironomidae* (Diptera) are important in adult diet early in the spring. In April and May caterpillars (Lepidoptera) (foraged for in hedges and trees) are the most important component, and by June-July spiders (Araneae) and dragonflies and damselflies (Odonata). Also taken are craneflies (Tipulidae), weevils (Curculionidae), horse flies (Tabanidae), mayflies (Ephemeroptera), sawfly larvae (Symphyta: Tenthredinidae) and Orthoptera (Cramp & Perrins 1994, Wilson *et al.* 1997a). Chick diet is comprised totally of invertebrates. Adults, however, do take seeds during the breeding season, particularly those of wild grasses (see below).

Wintering requirements

There is little emigration from Britain during the winter, with less than 1% of birds leaving the country (Prys-Jones 1984). Aside from a withdrawal from more upland regions, Reed Buntings are found in most farmland areas during the winter, although they are scarce in some regions such as mid-East Anglia and the south-west (Lack 1986). They are found widely spread over farmland in small flocks, although they congregate in evening roosts in marshy areas such as reedbeds. Most birds do not move far from the breeding site in the winter (Prys-Jones 1984) although movements of 10-20 km are quite common (Fennell & Stone 1976).

Invertebrates taken in the non-breeding season include spiders (Araneae), springtails (Colembola), Hemiptera, Diptera larvae and beetles. However, animal food is relatively unimportant outside of the breeding season; only 5% of corrected volume in 108 Reed Bunting stomachs collected in November-December was animal material (Cramp & Perrins 1994). Reed Buntings feed mainly on the seeds of grasses and herbs, taken from on or near the ground. A diverse range of seeds are taken, including goosefoots (Chenopodiaceae), amaranths (Amaranthaceae), chickweeds and mouse-ears (Carophyllaceae), crucifers (Cruciferae), lupins *Lupinus* (Leguminosae) and the seeds of wild grasses such as meadow-grass *Poa*, millet *Setaria*, fescues *Festuca*, rye-grass *Lolium* and cockspur *Echinochloa*. Grass seeds are particularly important, with Prys-Jones (1977) estimated that 67.5% of all seeds taken were grass seeds. Unlike other buntings cereal seeds are relatively unimportant to Reed Buntings, although they can form a larger proportion of the diet in the late winter when other seed stocks have been diminished. Other foods are taken, for example Reed Buntings in south-west England feed on decaying potatoes left after harvest (King 1985).

There is strong evidence that first-year (and, to a lesser extent, adult) survival decreased during the late 1970s and the 1980s, the time during which the British population was declining. These declines in over-winter survival were sufficiently large to have caused the population decline and, given that nesting success was relatively high during this period, are likely to have done so (Peach *et al.* 1999). It is likely that declines in this period were due to loss of winter food resources, especially weed-rich stubbles, the preferred feeding habitat (Wilson, Taylor & Muirhead 1996) that were lost at a high rate during the period of Reed Bunting decline.

Yellowhammer *Emberiza citrinella*

Yellowhammers are widely distributed across lowland Britain, being absent only from the Pennines and north and west Scotland, as well as a few pockets in Wales. However, although well distributed, there is marked variation in abundance with greater densities being found in the Midlands and eastern Britain. A slight contraction in range in the western part of Britain has been detected in recent years (Gibbons *et al.* 1993). This contraction in range is slight compared to the fall in abundance displayed in recent years: Yellowhammers are unique amongst British farmland birds in that they did not start to decline until the late 1980s. However, since that date the CBC (Common Bird Census) population index for Yellowhammers has declined rapidly (Siriwardena *et al.* 1998). Yellowhammers are likely to be placed on the red list of Birds of Conservation Concern in the near future.

British Yellowhammers are, for the large part, sedentary. Therefore, unlike some of the other species considered in this review, they require both suitable breeding and non-breeding resources to be present in the same area.

Breeding season requirements

Yellowhammers nest along field boundaries, either on or near the ground in ditches or uncropped margins, or in hedges or isolated shrubs (Kyrkos 1997). Nest height increases through the breeding season, as birds tend to shift from breeding in margins to breeding in hedges themselves (Bradbury & Stoate 2000). Therefore they select boundaries with hedges, ditches and wide uncropped field margins (Bradbury *et al.* 2000). Even if not used for nesting location, hedges, shrubs and hedgerow trees are required as singing posts, perches for vigilance and cover from predators (Biber 1993). The adjoining crops are also important in determining Yellowhammer presence, with birds tending to avoid boundaries adjoining grassland. Kyrkos *et al.* (1998) found that yellowhammer density decreased with increasing proportion of farmland under grassland. It may be that "modern" improved grassland has neither the weed density required by adult Yellowhammer or sufficient invertebrate prey for birds feeding nestlings. The dense sward structure of highly fertilized leys may also reduce access to invertebrate prey (Perkins *et al.* 2000).

Stoate & Szczur (1997) found that Yellowhammer showed a distinct preference for foraging in some habitats, and this habitat choice changed through the breeding season. In the early part of the breeding season (early June) rape and beans were the most favoured habitat, followed by tracks/set-aside and then barley. Pasture/woodland was poorly favoured, wheat even less so. However, later in the breeding season (late June – July) barley and then wheat were the most favoured habitats, followed by field boundary, rape/beans. Morris *et al.* (2001) found that Yellowhammers showed a consistent preference between different regions for foraging habitats such as boundary features and winter barley, with stubbles and cultivated grass fields being avoided. The use of boundary features reflects the abundance of key food groups in habitats such as hedges, ditches, scrubby field corners and grass banks, verges and margins. Nearly all foraging is within about 300 m of the nest. Perkins *et al.* (in press) also found a preference for grass margins, although there was no difference in the use of cut and uncut grass.

Adult diet is principally granivorous. Cereal grain and the seeds of wild grasses are preferred. These include meadow-grass *Poa*, rye grass *Lolium*, fescues *Festuca* and couch *Elymus* (Graminaeae). Seeds of other taxa, including composites (Compositae), bistorts, docks and sorrels (Polygonaceae) and chickweeds (Caryophyllaceae) form a smaller proportion of diet (Cramp & Perrins 1994, Wilson *et al.* 1997a). Almost all of this food is gained from foraging on the ground.

Stoate *et al.* (1998) found cereal husks in a high proportion of nestling faecal samples from Leicestershire, but also found invertebrate remains in every faeces. Invertebrate prey includes spiders (Araneae), Coleoptera including ground beetles (Carabidae) and weevils (Curculinonidae), Lepidoptera (both adults and larvae) and Diptera such as craneflies (Tipulidae) and st mark's-fly (Bibionidae) (Cramp & Perrins 1994, Stoate *et al.* 1998, Moreby & Stoate 2001). The importance of cereal grain (unripe) is probably much greater in summers of cold or wet weather when insect availability may be low.

Wintering requirements

A study of winter habitat use by Yellowhammers in Leicestershire by Stoate & Szczur (1997) found that both oil-seed rape and cereal crops were avoided in the winter. Cereal-based "wild-bird cover", planted with wheat and triticale, was the most-preferred habitat in the early winter, but less so in the late winter. Gamebird feeding sites were used throughout the winter. Another study, in Oxfordshire (Wilson *et al.* 1996), found that grass was used less than expected, as were a number of crops such as broad-leaved crops, intensive winter wheat fields and bare tillage. The only habitat type that a clear preference was displayed for was winter stubble. The winter diet of Yellowhammer consists of weed seeds as listed above, and cereal grain.

Corn Bunting *Miliaria calandra*

Corn Buntings are largely restricted to farmland in Britain, with a strong association with cereal farming. They also have strongholds in other habitats such as downland pasture and Hebridean machair. Declines in the British population were noted as long ago as 1920s, but have gathered pace in more recent decades (Gibbons *et al.* 1993). However, following a decline during the agricultural depression between the two world wars, populations and range in the West Midlands increased with the area of spring barley in the 1960s and 1970s (Harrison *et al.* 1982), but has since declined with losses in spring barley and rotational, short term leys (West Midland Bird Reports). The current population is estimated at 20,000 territories (Corn Buntings are polygynous, (Ryves & Ryves 1934)) (Donald & Evans 1995). The range has contracted massively, particularly in the west and north of the range, with some additional enlarging "gaps" in the east.

Breeding season requirements

Corn Buntings breed in open farmland and grassland, where they nest on or near the ground. It was previously thought that they greatly preferred to breed in barley (and that the decline in barley acreage could have been a cause of the recent decrease in Corn Buntings nationally) but this has been disputed (Donald 1997). Ward & Aebischer (1994) found a positive correlation between Corn Bunting density and spring barley on the South Downs, and Gillings & Watts (1997) found that Corn Bunting density was positively correlated with winter barley in the Fens, but the best predictor of density was crop diversity. Eighty percent of all nests were in standing crops. However, association with barley is likely to be due to the fact that barley is the most extensively sown spring cereal, harvested late and characteristic of rotational agricultural systems, which have other benefits for Corn Buntings (see below). Crick (1997) found no evidence that nests in barley were more productive than those in wheat.

The date of harvesting of crops may be important, as Corn Buntings are comparatively late-breeders (Yom-Tov 1992). This late breeding may make Corn Buntings susceptible to nest loss or fledgling death during harvesting; nearly 50% of nests are still active at the end of July, approximately the mid-point of winter barley harvesting (Crick *et al.* 1991). Less than 20% of nests are still active at the onset of harvesting of spring-sown crops in mid-August. The trend towards autumn-sown crops may have contributed to the higher rate of nest loss due to agricultural operations recorded post-1970 (Crick *et al.* 1994).

Floristic diversity may provide more suitable nesting sites; Hartley & Shepherd (1997) found that most (80%) Corn Bunting nests in the Outer Hebrides were under the cover of Hogweed *Heracleum sphondylium* (Umbelliferae). Birds also showed a preference for uncultivated land, where Hogweed was commoner. Pasture was avoided, probably because it provided no nesting cover.

Males use elevated perches for singing, as do all birds for loafing, preening and as vantage points for predators (Watson & Rae 1997). These perches are usually telegraph poles and wires, but stone walls, trees, bushes and small herbs are also used.

The diet of Corn Bunting chicks is mainly insects, although unripe grain is fed to them (Watson 1992, Gillings & Watts 1997, Brickle & Harper 1999).

Brickle and Harper (1999) found that sawfly (Symphyta) and moth (Lepidoptera) larvae, grasshoppers (Orthoptera), spider (Araneae) and carabid beetles (Carabidae) were the most important food items for nestling Corn Buntings. Flies (Diptera) may be important in areas with damp or aquatic habitats (Hartley & Quicke 1994), although Corn Buntings are traditionally associated with free-draining soils. Adult diet in the breeding season is similar to that fed to chicks (Wilson *et al.* 1997a). Corn Buntings do most of their foraging within crop fields, although field margins and road verges are also used, and at all seasons grain may be foraged upon at stock feeding trough, spread slurry, dung, bales and stacks and on seeds spread for stock-feeding (Watson & Rae 1997). Brickle *et al.* 2000 found that Corn Buntings in West Sussex preferentially foraged in grassy field margins, spring-sown barley, unintensified grass and set-aside, where invertebrate chick food was more abundant.

Wintering requirements

Historically, Corn Buntings were abundant around stack yards and threshing yards during the winter, so it appears that spilt grain was once an important winter food (Donald *et al.* 1994). With cleaner and more efficient farming practices such food sources are no longer available.

As with other buntings, seeds form the vast part of the winter diet of Corn Buntings. They differ from other buntings in that the grain of cultivated cereals is by far the most important component of diet, with no other plant family being important (Wilson *et al.* 1997a). Winter stubbles are the source of this preferred food. Donald & Evans (1994) found that 60% of Corn Buntings fed on winter stubbles, which were the only field type for which a consistent preference was detected. Weedy stubbles were preferred to clean ones. Winter cereals were avoided in all land-use types, and while grassland was used in some cases (unimproved and semi-improved grassland are used roughly in proportion to their availability), improved grassland was always avoided.

Watson & Rae (1997) found that favoured stubble fields in north-east Scotland were those with patches of tall vegetation remaining. Grass pasture was sometimes used, autumn-sown cereals seldom so. As with many small passerines, Corn Buntings feed close to cover (Robinson & Sutherland 1997), so hedgerow removal may have resulted in higher predation risk for birds during the winter period.

Corn Buntings gather in winter roosts, using reed-beds, gorse patches and scrub. In southern England roosts of 300-500 are not uncommon, which Lack (1986) suggests may be evidence of a lack of suitable roost sites, possibly due to recent habitat loss.

		Kestrel	Grey Partridge	Lapwing	Stock Dove	Woodpigeon	Turtle Dove	Barn Owl	Skylark	Yellow Wagtail	Whitethroat	Jackdaw	Rook	Starling	Tree Sparrow	Linnet	Goldfinch	Greenfinch	Reed Bunting	Yellowhammer	Corn Bunting
Nesting requirement	Buildings	▒			▒			■				▒		▒	▒						
	Tree holes	▒			■			▒				■		▒	■						
	Trees	▒				▒	▒						■				▒	▒			
	Shrubs					▒	■				▒					▒	▒	▒	▒	▒	
	Hedges					▒	▒				▒					▒		▒	▒	▒	
	Margins/rank grass and herbs		▒	▒						▒	▒								▒	▒	▒
	Cereal crops		▒	▒					▒	▒											▒
	Broad-leaved crops			▒					▒	▒											▒
	Grassland		▒	▒					▒	▒											
Foraging habitat	Woodland				▒	▒															
	Scrubland				▒	▒					▒				▒			▒	▒		
	Parks/gardens				▒	▒						▒		▒	▒			▒			
	Hedges										▒				▒			▒	▒		
	Margins/rank grass	■	▒		▒		▒	■	▒		▒	▒		▒	▒	▒	▒	▒	▒	▒	▒
	Damp/aquatic habitats			▒				▒		▒					■				■	▒	
	Cereal		▒	▒	▒	▒	▒		▒	▒			▒		▒	▒				▒	▒
	Broad-leaved crops				▒	▒			▒	▒			▒			▒				▒	
	Set-aside	■	▒	▒	▒		▒	▒	▒	▒		▒	▒	▒	▒	▒	▒	▒	▒	▒	▒
	Grassland	▒	▒	▒				▒	▒	▒		■	■	■		▒	▒				
Food	Tree seeds/fruit					▒						▒						▒			
	Grain						▒					▒	▒		▒					▒	▒
	Weed seeds		▒		▒	▒	▒					▒	▒		▒	▒	▒	▒		▒	
	Rape				▒	▒	▒									▒		▒			
	Crop foliage		▒		▒	▒															
	Soil invertebrates	▒	▒	▒								▒	■	■	▒						
	Other invertebrates	▒	▒	▒					■	■	■	▒	▒	▒	■		▒	▒	■	▒	■
	Vertebrates	■						■				▒									

Table 1 Main nesting, foraging habitat and food requirements of the species in the Farmland Bird Index during the breeding season. Black squares indicate resources considered to of major iinportance, grey squares those of secondary importance.

		Kestrel	**Grey Partridge**	**Lapwing**	**Stock Dove**	**Woodpigeon**	**Turtle Dove**	**Barn Owl**	**Skylark**	**Yellow Wagtail**	**Whitethroat**	**Jackdaw**	**Rook**	**Starling**	**Tree Sparrow**	**Linnet**	**Goldfinch**	**Greenfinch**	**Reed Bunting**	**Yellowhammer**	**Corn Bunting**
Foraging habitat	Woodland																	▨			
	Scrubland																	▨			
	Parks/gardens																	▨			
	Hedges																	▨	▨		
	Margins/rank grass	▨	▨		▨			▨	▨			▨			▨	▨	▨	▨	▨	▨	▨
	Damp/aquatic habitats							▨											▨		
	Cereal		▨	▨	▨	▨			▨				▨	▨							
	Broad-leaved crops				▨	▨							▨	▨							
	Set-aside	▨	▨		▨	▨		▨	▨			▨		▨	▨	▨	▨	▨	▨	▨	
	Stubble	▨	▨		▨	▨			■			▨	▨	▨	▨	■	▨	▨	▨	▨	■
	Grassland	▨	▨	▨				▨	▨			■	■	■		▨	▨				▨
Food	Tree seeds/fruit					▨						▨		▨				▨			
	Grain		▨		▨	▨			▨			■	▨	▨						▨	■
	Weed seeds		▨		▨	▨			▨			▨	▨	▨	▨	■	▨	▨	■	▨	
	Rape				▨	■									▨						
	Crop foliage		▨		▨	▨			▨												
	Soil invertebrates	▨										▨	■	■							
	Other invertebrates											▨	▨	▨					▨		
	Vertebrates	▨						■													

Table 2 Main foraging habitat and food requirements of the species in the Farmland Bird Index during the non-breeding season. Black squares indicate resources considered to of major importance, grey squares those of secondary importance.

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CHAPTER 5 THE PROVISION OF HABITATS FOR FARMLAND BIRDS UNDER AGRI-ENVIRONMENT SCHEMES AND OTHER MANAGEMENT PRACTICES

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5.1 Introduction

The previous chapter reviews the resources (food in summer and winter and nest sites) required by the 20 bird species within the farmland bird index. Put simply, the populations of many of these species are declining because the modern agricultural landscape fails to meet some or all of these needs. There is a whole suite of mechanisms by which food and nest site requirements could, once again, be provided within current farming systems. Broadly speaking these fall within two categories; (i) agri-environment schemes such as Environmentally Sensitive Areas ESAs and Countryside Stewardship Scheme CSS (e.g. Ovenden *et al.* 1998) and (ii) general agricultural practices such as reduced chemical inputs resulting from systems such as organic or so called Integrated ("precision") Farming or set-aside. In this chapter we briefly review the suite of different approaches available and the extent to which they provide the resource requirements of the 20 farmland bird index species. In doing so we highlight gaps in coverage with respect to species and habitats and identify priorities for the future action designed to reverse the downward trend of the farmland bird index.

In the review presented below we focus only on major schemes or practices relevant to the bird indicators that are (or could be made) available largely on intensively managed farmland at large geographical scales. We do this because reversing the farmland bird index requires meeting the resource needs of common widespread bird species. This will only be possible through national or regional action.

Within the major schemes or practices we consider only options that will provide benefits for farmland birds. Our aim is not to provide a detailed analysis of every scheme and option but to focus very specifically on the question 'what does this scheme or agricultural practice provide for farmland bird index species'?

In England DEFRA operate several agri-environment schemes, but the two largest are Environmentally Sensitive Areas (ESAs) and Countryside Stewardship Scheme (CSS) (e.g. Ovenden *et al.* 1998). In Wales, NAWAD and CCW operate Tir Gofal, which has succeeded earlier schemes (ESA and Habitat Scheme). In terms of general agricultural practices, the major management approaches that may benefit birds are often applied within organic farming, precision farming (or integrated farming systems) or set-aside and include practices such as reduced/targeted use of chemical inputs, minimal tillage and the incorporation of rotations.

Very few of the farmland bird index species occur in significant numbers (in terms of percent of the population in Britain [or England and Wales]) in non-farmland habitats. An exception to this is Skylark for which 34% of the breeding population in Britain is found on upland and marginal upland habitats (Browne *et al.* 2000). Management options for lowland heath within CSS include those designed to maintain, enhance or recreate heathland. Management options for lowland and upland heaths within, for example the Breckland and Dartmoor ESAs include those designed to maintain or recreate heathland. Management options including grazing restrictions on upland and lowland heaths lowland within Tir Gofal could affect Skylarks. These options could affect Skylarks but knowledge of this species requirements in such habitats is insufficient to assess the degree or indeed the direction (positive or negative) of such management.

In the section below we consider options within CSS, English ESAs and Tir Gofal, and practices within organic farming, integrated farming and set-aide. For each of these in turn we provide a (i) brief background to the scheme and an outline of the general aims and coverage; (ii) list the main management options that will benefit farmland bird index species; (iii) consider the latter in more detail in several categories: (a) management of the cropped area - whole fields and field margin in grassland and arable systems and (b) management of the non-cropped areas – boundaries (hedges and ditches), woodland and scrub.

In the final section of this chapter we match the resource requirements of the birds with the resources offered by the agri-environment and agricultural practices considered.

5.2 Agri-Environment Schemes

5.2.1 Countryside Stewardship

General outline and coverage of scheme

The Countryside Stewardship Scheme (CSS) was launched in England by the then Countryside Commission in 1991 as the Government's premier agri-environmental scheme for the wider countryside, outside of Environmentally Sensitive Areas. This voluntary scheme pays farmers to manage their land under prescribed management protocols through 10-year agreements with additional grants for capital works such as hedge laying and planting. Like all agri-environment schemes, agreement holders are also expected to adhere to specified "good farming practice". The main agricultural habitats targeted by CSS are: chalk and limestone grasslands, waterside land, old meadows and pastures, coastal areas, lowland heath, upland areas, old orchards, field boundaries, field margins and, from 2002, targeted arable options (Table 2). These habitats generally reflect those listed as Biodiversity Action Plan Priority Habitats (Carey *et al.* 2001). On farmland, Priority Habitats include Purple Moor Grass and Rush Pastures, Upland and Lowland Calcareous Grassland, Lowland Dry Acid Grassland, Upland Hay and Lowland Meadows, Coastal and Floodplain Grazing Marsh, Upland and Lowland Heathland, Cereal Field Margins and Ancient and/or Species-Rich Hedgerows (Ovenden *et al.* 1998).

A survey of a sample (427 agreements; 7%) carried out by CEH (Carey *et al.* 2001) suggests a broad distribution of agreement land across England, with relative concentrations in central and western pastoral areas but with lower coverage in East Anglian fenland, Nottinghamshire and East Yorkshire (Figure 1). This distribution tends to reflect the emphasis within the CSS on grassland ecosystems. Until 1998, the 'Field Margins' options were the only options to directly manipulate habitats within cereal rotations. During 1998-2001, a suite of arable options was trialed in two Arable Stewardship Pilot Scheme (ASPS) areas in East Anglia and the West Midlands. The most successful of these were introduced into CSS in 2002 (DEFRA 2001b), targetted initially at prime areas of residual farmland biodiversity, including Grey Partridge, Lapwing, Turtle Dove, Tree Sparrow and Corn Bunting. The ASPS included whole field options (overwintered stubbles followed by spring crops or fallow) and a range of field margin options (conservation headlands, grass margins (naturally regenerated or sown and beetle banks) and wildlife seed mixtures).

Farmers' applications to participate are selected on merit, depending on the quality of submitted protocols i.e. entry into the scheme is not automatic. This was also true for the ASPS. By 2001, the CSS had ca.12,000 participants (with a target of 30,000 by 2006) representing 150,000 ha of land under agreement (DEFRA 2001a). Although a high proportion of this area is upland habitat such as moorland, over 84,000 ha is grassland in one form or another (pasture, meadow or grass-heath (Roza (DEFRA) pers comm.) (Table 2), plus around 12,000 km of established grass margins and 10,000 km of restored hedgerow under the CSS.

Main management options that will benefit birds

CSS management options that are likely to benefit birds are outlined in the table below.

Management	CSS Code(s)	Option(s)
Adjustment of grazing regimes	P1 P4 UP1 UP2, UP3 UP4	Managing lowland pasture Managing chalk and limestone grassland Managing upland in-bye pasture Managing upland rough grazing Managing upland limestone grassland
Adjustment of cutting regimes e.g. promote hay cutting	H1, H3 UH1	Lowland hay meadow Upland hay meadow
Re-creation of grass or heath on arable land	R1, RR1 LH3, LRH3	Re-creating grassland on cultivated land Re-creating heath
Encouraging stubbles and fallow arable land	OS1-3 R3 SPR	Overwinter stubble followed by fallow Six-metre arable margin (cultivated) Cirl bunting special project
Encouraging spring crops and rotations	OS1 OS2	Spring crop following stubble Low input spring cereal following stubble
Creation and management of field margins/grass strips	R3 R4, R7 R4, R8 (no code)	Six metre arable margin Two-metre grass margin Beetle banks One-metre uncultivated margin along maintained field boundaries
Restoration and management of hedges	HR (no code)	Hedge restoration (capital item) Hedge maintenance (unpaid)

Nature of benefits to bird species included in the Farmland Bird Index

Management of cropped area – grassland

There is a general lack of bird data available that are specific to grassland CSS agreements and so the account below is largely inferred from other areas of grassland research (Vickery *et al* 2001).

Agricultural management that involves increased use of inorganic nitrogen fertilisers, regular re-seeding, intensive cutting and grazing promotes swards of fast growing and competitive grass species. Such swards tend to have reduced species and structural diversity and provide impoverished food resources (seeds and invertebrates) as well as reduced nesting opportunities for many farmland birds (for review see Vickery *et al.* 2001). Thus agriculturally improved and intensively managed (i.e. cut or grazed) grassland is likely to be a relatively poor habitat for breeding and wintering birds in Britain.

Management requirements under CSS seek to reduce the level of fertilizer inputs and the intensity of grazing and cutting regimes. As such, grassland CSS agreements may enhance breeding success of ground nesting species such as Skylark *Alauda arvensis* and Lapwing *Vanellus vanellus* by reducing nest and chick mortality through trampling or mechanical cutting. The prescribed management changes should also promote more species-rich and structurally-diverse swards (e.g. Carey *et al.* 2001) and so provide enhanced food resources (seed heads and invertebrates) for birds in winter or summer as well as nesting opportunities (Wilson *et al.* 1996a), but only where soil nutrient and/or moisture conditions are conducive.

The grassland managed under CSS agreement represents approximately 2% of the agricultural grassland and rough grazing in England (DETR 2001c). At a national scale the contribution of this relatively small area of land to national bird trends will depend on the quality (for birds) of re-created/restored habitats and the implementation of specific management procedures on the largest scale possible. Over half of the land under CSS agreement (44,125 ha out of 84,000 ha) is improved or semi-improved lowland and Culm grassland. Currently the value of such grassland to birds for feeding and nesting is probably relatively poor (Vickery *et al.* 2001). In relation to the reversal of the farmland bird index, there may be potential for change through the restoration of semi-improved grassland (perhaps rather than through the maintenance of Priority Habitats such as Lowland Meadows), but this may depend on the availability of low-available-phosphate soils that would be capable of enabling relatively species-rich swards to establish by 2020.

Management of cropped area – arable crops

In terms of the area covered, options to manage arable land under CSS agreements have, to-date, been under represented within the CSS. Arable options tested under ASPS were designed to provide the flexibility to manage both whole fields and margins (cereal or grass margins and wildlife seed mixtures for birds in winter and summer, DEFRA 2001b). These options have been developed in close collaboration with (among others) ornithological researchers and are designed as mitigation against the major management practices in arable systems that have been detrimental for birds (for review see Robinson & Sutherland in press).

Under ASPS, options that introduce spring crops and overwinter stubbles to rotations provide habitats for both Skylark and Lapwing populations on fields in summer (Wilson *et al.* 1997; Sheldon *pers. comm.*) or Skylarks, finches and bunting numbers in winter (Evans 1997a). In one of a small number of studies of the ASPS, the re-introduction of spring crops was investigated in relation to Lapwings in the West Midlands, where higher breeding densities were recorded on agreement land (under option 1B; stubble followed by summer fallows) compared to non agreement land. High variability in the use of spring habitats between years was, however, an important additional result that showed a response to the location and composition of the agreement fields; for example, Lapwings avoided normally preferred crops when sited near woods or when the sward was too tall in spring (Sheldon *pers comm.*).

For other birds, a second evaluation of the ASAP showed, from 100 farms (in total, 50 ASAP farms plus 50 non-agreement control farms) from eastern and western England, that at the whole-farm level, breeding Greenfinch *Carduelis chloris* and Reed Bunting *Emberiza schoeniclus* were on average, and wintering "granivores" were commoner on ASPS farms than on non agreement farms but for other species, such as Grey Partridge *Perdix perdix*, the results over the short term, were equivocal (Bradbury *pers comm.*).

Elsewhere, a winter study recorded very positive responses of birds to the provision of seed crops, although their response was highly dependent on the seed-crop composition. A greater number of birds was supported by crops containing kale, quinoa, cereal and linseed. The lowest number of birds was supported by crops containing sunflowers and buckwheat (Henderson *et al.* 2001b). Whether local responses of birds to options at the habitat level can be detected at the population level depends on the relative influence of the habitat at the farm or landscape scale. Nevertheless, the future national arable options are a subset of the options within ASPS, that is winter stubbles, spring cereals, summer fallows and wild bird cover. As such, they are expected to provide an exceptionally valuable year-round selection of resources for birds on arable land.

Management of non-cropped area – field margins

Arable field margin agreements currently occupy around 10,000 ha [by September 2001] of CSS agreement land and exceed a length of 22,000 km (DEFRA 2001a). The BAP target is for 15,000 ha by 2010 which equates to 25,000 km of 6m margins, although the CSS has substantial lengths of 2 m margins.

Combinations of grass margins and open field habitats provide a good example of successful CSS application. The Cirl Bunting *Emberiza cirlus* in Devon (Peach *et al.* 2001) was at one time close to extinction in the UK (Wotton *et al.* 2000) before intensive research revealed relatively discrete habitat requirements. These included winter stubbles (created initially under set-aside then later under a CSS Special Project) as a source of winter food (Evans & Smith 1994) and zero-input grass margins as a source of invertebrate prey for chicks in summer (Peach *et al.* 2001). Cirl Buntings increased by four fold, to around 400 pairs, during the 1990s (Wotton *et al.* 2000), assisted by the Special Project, with 83% increases on agreement land compared to a 2% increase on non-agreement land (Peach *et al.* 2001).

As is the case for Cirl Buntings, properly managed field margins also provide feeding sites for Grey Partridge (Rands 1985, 1986), Turtle Dove *Streptopelia turtur* (Browne & Aebischer 2001) and ground feeding passerines such as Tree Sparrow *Passer montanus*, Linnet *Carduelis cannabina*, Yellowhammer *Emberiza citrinella* and Corn Bunting *Miliaria calandra* (Parish *et al.* 1994, Stoate 1999, Bradbury *et al.* 2000), through supporting a diversity of arable plants (Wilson & Aebischer 1995) and/or invertebrates (Moreby *et al.* 1994, Feber, *et al.*, 1995, Barker *et al.* 1997, Haysom 1999). Some form of field margin management has been included as an option within most of the recent agri-environment schemes in the UK, including the Countryside Stewardship, ESAs (below) and Organic Farming (below). Estimates of approximately 400,000 km of cereal field edge in the UK would equate to approximately 240,000 ha of land at 6 m wide. This is, potentially, 5% of the UK's arable area. However, the suitability of field margins as foraging and nesting habitat for farmland birds will be largely determined by the way in which their management influences the abundance of food, its accessibility and by vegetation structure to provide sufficient cover for nesting (Vickery & Fuller 1998). A number of different management options exist within the CSS. These include: grass margins (grass-only strips, grass/wildflower strips and beetle banks), uncropped wildlife strips (zero-input margins), conservation headlands, wild bird seed mixtures and buffer strips (4-12 m grass strips on which the farmer abstains from pesticide or fertiliser inputs to protect potentially sensitive adjacent wildlife habitats such as hedgerows, streams or ponds).

Grass and grass/wildflower margins and CSS wildlife strips
This is the commonest form of field margin, as 2 m or 6 m widths, and is likely to occupy the majority of the length incorporated under CSS agreement. Grass seeds in grass-only strips are fed on by a range of bird species (Eaton, this report) including Starling *Sturnus vulgaris*, Tree Sparrow and Yellowhammer (Cramp *et al.* 1977-1994). Grass-only and grass/wildflower strips may also benefit small mammals (Harris & Woollard 1990, Tew *et al.* 1992) and hence increase hunting opportunities for birds of prey such as Kestrels *Falco tinninculus* and Barn Owls *Tyto alba*. Specifically on grassland, CSS "Wildlife strips", are zero-input margins designed to provide potential nesting habitat for birds such as Grey Partridge and Yellowhammer, and support the invertebrates on which these species may depend.

Grass and grass-herb strips are usually mown in August or September to prevent scrub encroachment. However, such frequent cutting may reduce the value of grass margins for ground-nesting birds such as Grey Partridges (e.g. Rands 1985, 1986) and Yellowhammers (Stoate *et al.*1998). Both species will benefit from the typical requirement to maintain taller strips near to a hedge that are mown only one year in every two or three for a more diverse structure and for insects to complete their life-cycles.

Uncropped cultivated strips
Uncropped wildlife strips are 6 m wide strips of land adjacent to the arable crop together with a 1 m wide sterile strip between the wildlife strip and the crop. The wildlife strip comprises naturally-regenerated vegetation cultivated once every year or two years to encourage annual and biennial arable flowers (Critchley 1994). The sterile strip is maintained so as to prevent aggressive arable weeds spreading into the adjacent crop. This type of field margin has, to date, been restricted to light and/or shallow soils and targeted at the conservation of rare arable weeds such as Pheasant's Eye *Adonis annua* and Cornflower *Centaurea cyanus*. If developed in the wider countryside, these may benefit a range of rare or declining arable plant species which germinate from seed banks that persist at field edges, e.g. Rough *Papaver hybridum* and Prickly Poppy *P. argemone* and Narrow-fruited Cornsalad *Valerianella dentata* (Wilson & Aebischer 1995). As well as annual weeds and their seeds, many plants that utilise this disturbed ground support phytophagous insects, especially Hemiptera, Coleoptera and Lepidoptera (Hawthorne & Hassall 1994, Hawthorne *et al.* 1999). Unlike conservation headlands, these strips are not necessarily cultivated every year, so minimising the impact on ground dwelling invertebrates (e.g. Hassall *et al.* 1992; Barker *et al.* 1997). This abundance and diversity of seed and invertebrate food, combined with the open patchy nature of the sward means that uncropped wildlife strips are likely to provide good feeding opportunities for ground-foraging farmland birds (e.g. Henderson *et al.* 2001b). In addition, results from an Arable Stewardship evaluation study indicated increases in Greenfinches and Reed Buntings in response to prescriptions that included uncropped wildlife strips.

Conservation headlands

Conservation headlands (under the new arable options) comprise either a 6 m to 24 m wide strip forming the outer margin of the crop with reduced inputs of herbicides and (in a separate option) fertiliers. This should favour arable plants, particularly broad-leaved species, and invertebrates (such as sawflies (Hymenoptera) and plant bugs (Hemiptera); Dover 1996, Hassall *et al.* 1992) which live on them. The invertebrates associated with conservation headlands have been shown to benefit Grey Partridge chick survival (e.g. Potts 1997; Rands 1986), and small mammal populations (Tew *et al.* 1992) that could benefit avian predators such as Kestrel and Barn Owl).

Wild bird seed mixtures

Wildlife seed mixtures are available to farmers as an arable option within the CSS for 2002. Many landowners in the UK grow "wild bird crops" to provide winter food and cover for gamebirds (Ring-necked Pheasants *Phasianus colchicus,* Grey Partridge and Red-legged Partridge *Alectoris rufa*). These crops are versatile but in location are generally grown along field edges. Depending on their content, they can provide a sustained abundance of seed for birds over winter (Game Conservancy Trust). Content is variable but can include seed-bearing plants such as maize, millet, cereals, kale, mustard and exotic seed-producing plants such as quinoa. Mixes are adapted to attract indigenous seed-eating passerines as well as gamebirds, and some mixes are eligible within the set-aside scheme (as "wildbird" mixtures). Higher densities of birds are found in winter bird crops than in conventional crops (e.g. winter cereals) so a positive response of birds to the provision of such crops under the CSS is expected. Experimental work has shown that the content significantly affects the use of wildbird crops by birds (Boatman *et al* 2000, Boatman *et al.* 2001, Henderson *et al* 2001b). On average the highest densities and greatest range of bird species (such as Grey Partridge, Tree Sparrow, Linnet, and Yellowhammer) were associated with crops containing brassicas (such as kale or turnips), quinoa, linseed and cereals (such as triticale for Yellowhammers and Reed Buntings). The recommended mix in the new CSS option (WM1) is now a mix of cereals (triticale), kale and quinoa, although this can be varied.

Management of non-cropped area – field boundaries

Options to manage hedgerows, ditches and banks are available through whole farm Field Boundaries agreements, but more commonly they are incorporated into CSS targeted landscapes, such as Chalk & Limestone Grassland, Waterside Landscape (Table 2). Some sections of CSS hedgerows meet the definition of the Ancient and/or Species-rich BAP Priority Habitats. Altogether, approximately 10,000 km of restored or regenerated hedgerows are under CSS control across England. These represent around 3-4% of the total length of hedgerow in England (DEFRA 2001c). Hedgerows maintained without the cover and food provided by fringing vegetation are likely to be sub-optimal for several key farmland birds, such as Whitethroats *Sylvia communis* (Green *et al.* 1994, Stoate 1999), Grey Partridge (Potts 1997), Yellowhammer (Perkins *et al.* in press). CSS Field Boundaries agreements also require adjacent 1 m strips to be left uncultivated and free from agrochemicals and where appropriate landowners should be encouraged to include grass margins, wildlife strips or buffer strips along the length of their CSS managed hedgerows. Indeed, including such margins along all hedgerows would be a predictably effective and practical way of increasing the value of both the hedgerow and field margin prescriptions for birds.

On most farmland, hedgerows provide the only real alternative to scrub. Scrub is a valuable bird-habitat (e.g. Browne & Aebischer 2001) that is usually removed on both farmland and conservation land, partly because of its invasive properties but in some cases, purely for 'aesthetic' reasons (Gough & Fuller 1998). As such, hedgerows can provide a suitable but more controlled alternative given an appropriate profile (trees, thickets, rank vegetation, such as brambles, and grass margins). Hedgerows under CSS agreement have to be cut no more than two years in five, but this does necessarily provide the full range of appropriate structural variation (e.g. Green *et al.* 1994) and the CSS may need to recognise the value of older, less frequently cut and more structurally diverse hedgerows (Smallshire pers comm.).

5.3 Environmentally Sensitive Areas

5.3.1 General outline and scheme coverage

Following the 1986 Agriculture Act and CAP reforms, in 1987 ESAs were first established (Stage I). In all, 22 English ESAs occupy downland, upland, heathland, coastal and flood plain areas (Figure 1, Table 3). ESAs generally comprise nationally important concentrations of semi-natural habitat and therefore tend to conserve land that supports significant populations or special examples of English flora and fauna. Management protocols divide objectives into tiers that are more demanding of agreement holders at higher levels. Objectives target the maintenance and/or enhancement of landscape, historic interest and wildlife and in general the lower tiers have higher uptake among landowners (Ecoscope 2001).

In England, ESAs cover approximately 890,000 ha of eligible land of which around 43% (380,000 ha) is under agreement at the lower tiers (varying between ESAs; Table 3). The Lake District and Dartmoor ESAs together occupy around one third of the eligible area. A large component of agreement land is upland heather moor or rough grazing. Grassland covers over 270,000 ha of ESA agreement land, (extensive pasture 214,000 ha, hay and extensive silage 39,000 ha and wet grassland up to 22,000 ha; Table 3). Arable options include field margins and conservation headlands (less than 400 ha of agreement land), but substantial areas of stubbles have come under agreement in four ESAs. Hedgerow prescriptions are common to virtually all ESAs (Table 3).

5.3.2 Main management options that will benefit Farmland Bird Indicator species

- Adjusting grazing regimes (extensive grazing (all ESAs) and wet grassland (11 ESAs).
- Adjusting cutting regimes e.g. promote late cuts (hay or silage grassland; 20 ESAs).
- Restoring grassland or heath on arable land (reversion to grassland is an option in 12 ESAs).
- Reducing fertiliser inputs (all ESAs).
- Retaining winter stubbles (four ESAs).
- Creating and managing field margins (arable field margins and conservation headland options are available in five ESAs, ditch management in 13 ESAs).
- Restoring and managing hedges (hedgerow options are available to 19 of the ESAs).
- Retaining mature and dead trees (tree management is an option for at least 11 ESAs).

5.3.3 Nature of the benefits to bird species included in the Farmland Bird Index

Management of cropped areas and grassland

Around 270,000 ha (71%) of ESA agreement land is grassland. This represents approximately 6% of the agricultural grassland and rough grazing in England (DEFRA 2001c), and has the potential for an effective contribution to reversals in the populations of some farmland bird species. Evidence to date suggests that by and large, bird populations on monitored ESA land have not tended to increase. Even where there have been gains in suitable habitat (such as meadows or wet grassland; ESAs 2, 4 & 18, Table 3) birds such as Lapwing have either remained stable or they have continued to decline. This suggests that either there is not enough habitat or that greater precision is needed in the type of habitat required or that farmers are flexible in the way they interpret the way grassland is managed. One study, which compares managed bird reserves with ESA land, indicated that this was so (Ausden & Hirons, in press). They suggest that relatively low numbers of breeding Lapwing on ESA agreement land (compared to those on nearby bird reserves), could be increased though uptake of the higher tier scheme objectives (where they exist) within the ESA, which improve the suitability of areas of grassland by raising water levels for this species (and potentially species such as Yellow Wagtail *Motacilla flava* too). Arable options are rarer within the ESA scheme, but there are whole-field stubble options in Breckland, the Cotswolds, South Downs and West Penwith and undersowing options in the South Downs ESA.

Throughout the ESA scheme, maintenance objectives prevent further reversion of grassland to arable land, prevent further damage to grassland by introducing fertilisers, increasing fertiliser rates or drainage activities. Meanwhile, research on the South Downs has shown that, as with non-rotational set-aside, correctly managed reversion land (to grassland) from relatively unproductive arable land can benefit Skylarks in winter and summer (Wakeham-Dawson & Aebischer 1998, Wakeham-Dawson *et al.* 1998). Further enhancement might be achieved through close control of grazing stock levels on extensive pastures (Morris 1971, Berg *et al.* 1997). As with CSS land, infrequent grazing can help improve the structure of grassland and reduce trampling mortality on grassland for breeding Lapwing or Skylarks, depending on the precise level and timing of the grazing protocol. Grazing by cattle, for example, produces a more varied sward structure than grazing by sheep (Tallowin *et al.* 1986, Berg *et al.* 1997) to produce a sward more suitable for nesting Lapwing (Galbraith 1988, Small pers. comm.). This can increase the associated invertebrate community (Morris 2000). Grazing in autumn rather than spring helps to retain sward composition, with less impact on invertebrate populations (Brown *et al.* 1990) and avoids the nests and broods of birds in spring and summer (e.g. Beintema & Muskens 1987).

On meadows and silage fields (18 ESAs affecting at least 37,000 ha; Table 3), restrictions on the timing (and frequency) of mowing to July should improve structure and conditions for ground-nesting "meadow birds" (Beintema & Muskens 1987) such as Yellow Wagtail (Mason & Lyczynski 1980), Skylark (Wakeham-Dawson *et al.* 1998) and Lapwing (Galbraith 1988). In the Pennine Dales, for example, on non-agreement land, one study observed that 25% of 87 Yellow Wagtail nests were destroyed by grass-cutting in June or early July (peak nesting time is mid-June), while on ESA agreement land, none of 135 Yellow Wagtail nest sites were destroyed when cutting was delayed until late July (ADAS 1995). Delaying cutting until July will avoid the peak nesting seasons of both Skylark and

Lapwing and provide a sward to support invertebrates (Vickery *et al.* 2001), food and cover for young birds such as Grey Partridge (Potts 1997).

Arable reversion (10,904 ha in total; Table 3) is important for the restoration of Priority Habitats (such as chalk downland or heathland) that may have been lost in the past to arable farming (Vickery *et al.* 2001). Under appropriate mowing or grazing management, reversion land is likely to increase the numbers of breeding Skylarks it can support (Wakeham-Dawson *et al.* 1998). Potts (1997), however, questions the perception that arable reversion is necessarily beneficial to biodiversity in general and to Grey Partridge in particular. This depends on the management of reversion land as grassland, which if maintained as short, grazing pasture would be unsuitable for Grey Partridge. Grey Partridge tend to avoid open short grassland, preferring taller but sparse meadow or the cover afforded *Juncus* corners or grass margins (Potts 1997). The correct conditions, however, are not easily achieved on previously fertilised grassland, without active management and by simply reducing grazing pressure. Without also removing nutrients this could simply increase the litter layer which prevents colonisation of more desirable plant species. Arable "pockets" within pastoral areas meanwhile, contribute to increased habitat diversity, which can lead to higher densities of some seed-eating birds such as Skylarks and Corn Buntings (Chamberlain & Gregory 1999, Robinson *et al.* 2001). Nevertheless, a net loss of potentially valuable permanent grassland to arable land in the order of 126,00 ha nationally between 1992 and 1997 (British Wildlife (13) 2001) suggests that, in some regions, reversion is an important option within the ESA scheme.

Restrictions on fertilisers and pesticides affects around 176,000 ha of grassland. Both measures should increase plant species richness (fertilisers favour competitive nitrogen-responsive grasses), although this is typically severely restricted by residual soil fertility. Food resources for birds would then include plants and weed-seed material (e.g. for Turtle Dove or Linnet (Wilson *et al.* 1996a)), as well as dependent arthropod populations for Grey Partridge and other birds provisioning chicks. On ESA land, herbicides are generally restricted to contact applications to thistle *Cirsium* species or docks *Rumex* species, which avoids small or prostrate grasses (such as such Fescue or *Poa* species) or herbaceous species (such as chickweeds) that can occur in extensive grassland or that can provide food for birds (Wilson *et al.* 1996a, Browne & Aebischer 2001).

Management of margins

The content, composition and use of various field margin options by birds is discussed at length under the CSS above. Under the ESA scheme, field margin options are limited, under higher tiers, to the Breckland, Cotswold Hills, South and South Wessex Downs and Clun Hills. The Broads, Somerset Levels and West Penwith, however, include "buffer strips" as options for (mainly) grassland habitats. Uptake of the field margin option is only prominent within the Breckland ESA, where uncropped wildlife strips and conservation headlands occupy 234 km of field edge (Table 3). They provide cereal seeds, weed seeds and summer invertebrates for birds such as Grey Partridge or plant food for species such as Turtle Dove (Browne & Aebischer 2001). Most other ESAs do not offer field margin options within their prescriptions, due to the predominance of grassland there. As in arable systems, however, in grassland areas, grass margins (perhaps behind electric fences), would contribute structural diversity, cover and/or food for species such as Grey Partridge and Yellowhammer, and perhaps should be considered for the wider inclusion across the ESA scheme where circumstances allow. Buffer strips provide protection to adjacent wildlife habitat from inputs onto grassland but are not fenced off, and while there may be landscape implications for using electric fences in some ESAs the effectiveness of some margins and buffer strips to protect some habitats such as hedgerow or ditch-side vegetation, might potentially be compromised by grazing.

Management of boundaries

The preferred content and composition of hedgerows for birds is discussed under the CSS above. Hedgerow options are common to almost all of ESAs (Table 1) but often the emphasis is towards maintaining traditional or stock proof boundaries (Ecoscope 2001). For birds, taller, wider hedges, with occasional trees, support larger numbers and species of birds (e.g. Parish *et al.* 1994). Structural variation and adjacent habitats, such as rank vegetation and grass margins, would further increase the habitat options for birds, and increase the suitability of boundaries to a greater variety of bird species (Parish *et al.* 1995). In the Blackdown Hills, where trees are important landscape elements, tree-tagging is a feature of the ESA. Elsewhere, restored hedges may be fenced-off to prevent stock damage to hedge regrowth and marginal vegetation but for existing hedges, hedge structure itself, as managed by the farmer or landowner, is the most important determinant of hedgerow use by birds.

5.4 Organic Farming

Regulated under the Government's UK Register of Organic Food Standards (UKROFS 1992), organic farm management is based on traditional systems, using crop rotations to supply and utilise nitrogen, by using legumes, "green manures", animal manure and composts that help maintain soil fertility. Weed control is mechanical or achieved through the choice of crop varieties and the timing of cultivations, but there is no use of synthetic pesticides, herbicides or fertilisers (Lampkin 1990). Land-based (as opposed to indoor) live stocking is integral to most organic farms. Within the rotation, reliance on nitrogen-fixing but low density clover-rich pastures means generally lower grazing stock densities are in any case, supported compared to conventionally improved clover-grass leys (Soil Association 2001). In Britain, organic management also includes sympathetic management of non-cropped habitats, such as field boundaries, woodlands and hedgerows (Chamberlain *et al.* 1999).

In 2001, organic farms occupied around 2.5% of UK farmland (3500 growers occupying around 0.5 million ha in total (0.2 million ha in England and Wales, 0.3 million ha in Scotland (DEFRA 2001c)).

5.4.1 Main management options that will benefit birds

- Increased invertebrate densities through the low use of insecticides and (indirectly) herbicides.
- Increase arable weed plant densities (low use of herbicides).
- Increase crop/field diversity (field rotation; spring crops, leys and fallows).
- Create field margins (boundary management and pest species control).
- Retain mature and dead trees and boundary structure (hedge and tree management).

5.4.2 Nature of the benefits to bird species included in the Farmland Bird Index

Management of whole-field areas

Studies have demonstrated a higher density of birds on organic farms than conventional farms (Brae *et al.* 1988, Hald & Reddersen 1990, Chamberlain *et al.* 1999). Since a principal ethos of organic farming is to restrict organic losses from the system (Soil Association, 2001), on average organic systems should support higher densities of arthropod groups that respond to organic material, especially where no pesticides are used. Most important among this group are aneceic species of earthworms (e.g. *Alolobophora* species) within the humus and mat layers of soils or grassland, Carabid ground beetles (Armstrong 1995) and some spider groups such as Lycosidae and Linyphiidae (Feber *et al.* 1998). Birds such as Lapwing (Beintema *et al.* 1990) and Grey Partridge (Green 1984) feed on these invertebrates (Eaton, this report).

On average, a higher cover and a greater complexity of broad-leaved wild plants on organic farms, such as Fabaceae, Brassicaceae & Polygonaceae, compared to nearby conventional farms (Hald & Reddersen 1990, Moreby *et al.* 1994, Brooks *et al.* 1995, Frieben & Kopke 1995, Kay & Gregory 2000), is probably due to the low use of herbicides on organic farms. Two studies specifically link broad-leaved weeds in organic systems to higher densities of key invertebrates for birds (herbivorous arthropods, sawfly and lepidoptera larvae (Moreby *et al.* 1994) and carabid ground beetles (Armstrong 1995)). This emphasises the overall potential of organic farms to provide plant and arthropod (e.g. sawfies; Barker *et al.* 1997) food for birds from crops and non-cropped areas.

In organic systems, mixed farming, through the addition of livestock, is additional to the use of field variety to combat pests and disease. Both return nutrients to the soil using legumes and clover leys as well as dung from livestock. Over-wintered cattle provided with feed and bedding, have also been show to be an important source of food for seed-eating birds such as Corn Bunting (Brickle & Harper 2001). Variable timing in the management of different crop types presents birds with a variety of foraging and/or nesting options. Corn Buntings, Skylarks and Lapwings nest at higher densities in areas of crop diversity (Ward & Aebischer 1994, Wilson *et al.* 2001). For example, Lapwings typically nest in short cover, such as fallows, stubbles or spring crops, but move their chicks to taller grassland where they can forage in cover (Wilson *et al.* 2001). On the grassland areas of the rotation, invertebrates associated with dung and higher levels of organic matter are likely to be important for the diet of adult and young Lapwings (Beintema *et al.* 1990). For Skylarks, spring cereals, fallows or legumes (such as peas) and relatively weedy first-year leys, allow birds to raise first broods in the low growing winter crops then later broods among spring crops. In this way they can raise second and possibly third broods over an extended breeding season (e.g. Wilson *et al.* 1997). Organic systems should therefore benefit both of these indicator species, but cattle grazed leys can also provide important foraging habitat for Rooks, Jackdaws and Starlings in the breeding season (Cramp *et al.* 1977-1994, Eaton this report). In addition, first year leys on organic farms, being generally more weedy and stubble-like, can provide good wintering and breeding sites for Skylarks (Gillings & Fuller 1999).

Some of the benefits provided by organic crops could be offset by the mechanical methods of weed control that are widely used on organic farms, particularly on row-crops (e.g. potatoes, and young crops of legumes; Sillet *pers comm*). For crops such as sugar beet, inter-row cultivation is a normal practice, whereas in cereals harrowing is more common. On average there are two passes per crop, typically carried out (for winter cereals) in mid/late March then early April, but later for spring cereals (Sillet *pers comm.*). Both harrowing and inter-row cultivation are potentially damaging to established nests of birds such as Lapwing and Skylark. April is a crucial time for Lapwing and Skylark to settle on nests. Lapwing will incubate and raise chicks through April and May, while Skylarks continue nesting through June and early July. Clearly the implementation and timing of mechanical weeding would require careful attention if this activity was not to seriously reduce the nesting success of these bird species and compromise the benefits provided by the organic system.

Management of margins and boundaries

A general "holistic" approach to the farm management is typical of organic systems. Organic farms tend to support larger hedgerows, more trees and more developed field margins than conventional farms (Entec 1995, Chamberlain *et al.* 1999). Around 40% more birds were recorded on organic farms compared to conventional farms in a three year study of 44 farms in England (Chamberlain *et al.* 1999) but part of this was probably explained by the quality of non-cropped habitats and boundaries associated with organic farms – rather than the farm regime itself. If this phenomenon is a consequence of organic farmers tending to belong to a more wildlife-sympathetic group, then expansion of the organic industry into the wider farming community would require firm management guidelines to ensure new farmers developed and adopted similar attitudes and practices towards non-cropped habitats. For hedgerows, this would include retention of dead wood or mature trees (for nesting Tree Sparrows and Stock Doves *Columba oenas*), shrubby vegetation (for nesting Turtle Doves) and the creation of thicket-hedgerows and grass margins (for nesting and foraging Yellowhammers and Grey Partridge (Stoate 1999, Potts 1986).

5.5 Tir Gofal

Tir Gofal is a whole farm scheme, available throughout Wales since March 1999. The scheme is delivered by the Countryside Council for Wales (CCW) together with a range of other partners on behalf of the National Assembly for Wales. Tir Gofal is designed to encourage farm practices that will protect and enhance landscapes, cultural features and wildlife as well as provide new public access opportunities. Entry for farmers is on a voluntary basis. Applications are selected provided they score above a set threshold according to a publicly available scoring system.

Agreements last for 10 years with a break clause exercisable by either party at the mid point. Annual payments are available in return for delivering basic environmental requirements on a whole farm basis as well as managing all existing wildlife habitats on the farm according to published guidelines. In addition, farmers can select from other annual payment options such as arable land management, the creation of new habitats and provision of new public access. In addition, there are a wide range of capital works projects which are funded on a one-off basis.

Tir Gofal is versatile and offers many varied and specific options for farmers to manage grassland, arable land and non-cropped areas such as scrubland, hedgerows and woodland for wildlife. In some cases specific objectives target birds, such as the restoration of improved grassland for Lapwing. At the end of July 2002 there were a total of 853 agreements covering 83,910 ha or some 5% of Welsh agricultural area and a further 260 agreements in preparation. Existing habitats, such as broadleaved woodland and scrub, heathland and wetland, accounted for 64% of this agreement land (54,000ha). In addition 3731 ha of improved grassland and arable were entered into prescriptions designed to improve the value of land for wildlife. Thus, combining these two categories, 64% of agreement land is managed as wildlife habitat although there is large variation between farms in different parts of Wales (B Pawson pers comm).

5.5.1 Main management options that will benefit birds

- Adjustment of grazing regimes and reduction in agricultural inputs (all grassland and heathland types).
- Adjustment of cutting regimes e.g. promotion of hay cutting (unimproved and semi-improved grassland).
- Encouraging weed-rich stubbles, crops and fallow arable land (arable land prescriptions).
- Creating and managing field margins (grass and fallow, game cover crops).
- Restoring and managing hedges (traditional boundary prescriptions).
- Management of scrub and woodland (scrub and woodland prescriptions).
- Management of traditional orchards and parklands (orchard and parkland prescriptions).
- Raising water levels (specific prescriptions according to habitat).
- Habitat creation (woodland, reedbeds etc).

5.5.2 Nature of the benefits to bird species included in the Farmland Bird Index

Management of whole-field areas – grassland

The grassland options under Tir Gofal cover many forms of grassland from permanent pastures and rough grazing to improved pastures, coastal grazing and wet grassland/grazing marshes.

Management guidelines limit grazing densities on permanent grassland (neutral, semi-improved and improved pasture) and on hay meadows, cutting is restricted to 1 July. These measures provide the correct sward structure and composition (through restricted inputs of fertilisers and herbicides and grazing controls) for breeding Lapwings. In meadows, they help increase plant species richness but also protect the nests and broods of birds (such as Lapwing and Skylark, potentially Grey Partridge and possibly Yellow Wagtail) from early mowing or harrowing activities. There are, however, no detailed monitoring data to indicate the success of the scheme to date. As with other agri-environment schemes, grazing pastures and improved grassland probably offer the greatest opportunity to affect change in the environment in such a way as to help reverse farmland bird declines (for grassland dependent species).

Management of whole-field areas – arable land

Arable options under Tir Gofal encourage the management of existing crops by reducing the application of pesticides, and encouraging the retention of stubbles, fallow land and grass margins. In addition, existing improved grassland can be converted to unsprayed cereal and root crops provided all stubbles are retained throughout the winter. These important provisions will allow weed populations to develop on stubbles in late autumn and fallows in summer. Weedy stubbles are especially valuable to seed-eating birds in winter and their loss has probably been significant in their declines on large areas of arable farmland in the UK (e.g. Evans 1997a). They provided weed-seeds as food for Linnets, Skylarks, sparrows and buntings, and in summer support invertebrate populations on which birds such as Grey Partridge, feed their young (e.g. Moreby *et al.* 1994). Low density grazing on fallow land, to return organic material from dung to the crop, will benefit earthworms and other invertebrates on which birds such as Lapwing feed (Bientema *et al.* 1990).

Management non-cropped areas - field margins

The management of field margins, such as grass margins, conservation strips and winter bird cover is discussed at length under the Countryside Stewardship Scheme above. The same benefits apply to habitats managed under Tir Gofal. In particular, grass or grass-herb margins provide food and nest sites for Grey Partridge (Potts 1997) and Yellowhammer (Perkins *et al.* in press) in summer and food and cover in winter. Winter bird crops containing kale (or other brassicas), quinoa, linseed and cereals provide valuable food throughout the winter for a range of species including Grey Partridge, Tree Sparrow, Linnet, and bunting species (Henderson *et al.* 2001b).

Annually cultivated (in winter), unsprayed *fallow margins* are available under Tir Gofal (cf. uncropped wildlife strips (ESA scheme) or cultivated strips with the CSS). This promotes through natural regeneration, the development of weed populations along 4 or 6 m strips that are established along crop edges. These are ideal conditions for providing summer and winter sources of plant and invertebrate food and light cover, for birds such as Turtle Dove in summer that have strongly declined throughout Wales (Gibbons *et al.* 1993; Browne & Aebischer 2001), Grey Partridge (Potts 1997), sparrows, finches and buntings (Evans 1997a, Eaton, this report), in a manner similar to high quality set-aside (Henderson & Evans 2000). Apparently, fallow margins are not an option for grassland management but perhaps could be considered in this context since this would offer an effective and practical method of both diversifying habitat options and adding food value to predominantly pastoral systems (e.g. Robinson *et al.* 2001).

Management of non-cropped areas – boundaries, scrub and heathland

Boundary habitats such as hedgerows, ditches and areas of scrub offer essential nest sites, cover and foraging options for farmland bird indicator species, given appropriate management. Under Tir Gofal, the buffering of ditches (cf. buffer strips in the CSS and some ESA schemes such as the Somerset Moors) and other water features by 6 m margins will protect the habitat from crop sprays and other activities that could damage herbs and reduce the numbers of invertebrates associated with the wet substrate. Wetter conditions are likely to benefit bird species such as Tree Sparrow (Field *pers. comm.*) and Reed Bunting (Cramp & Simmons 1977-1994).

The importance of hedgerow structure and profile for birds (Parish *et al.* 1994) and the value of scrub (Wilson *et al.* 2001) are discussed under Countryside Stewardship Scheme above. Under Tir Gofal, hedgerow guidelines emphasise a traditional system of layering or coppicing. As with the Countryside Stewardship Scheme (above) it is not clear how this form of management would necessarily benefit birds, for which hedge height, fringing vegetation and mature trees are key features (Parish *et al.* 1996).

Tir Gofal prescriptions require the protection of trees or dead wood in boundaries, since these have both landscape and cultural value as well as being of importance as nest sites for indicator bird species such as Stock Dove, Barn Owl, Tree Sparrow and Jackdaw *Corvus* *monedula*. This requires furher monitoring, however, as deterioration and loss of breeding habitat would compromise the other benefits provided by the scheme. The Tir Gofal scheme also promotes the positive management of scrub for wildlife. This important provision, will encourage the management of the structural profile of scrub for the benefit of bird species, such as Turtle Dove, Whitethroat, Tree Sparrow, Linnet, Yellowhammer and Reed Bunting.

5.6 Set-aside

The introduction of set-aside as a voluntary scheme in 1988, into arable and mixed farmland in the UK was the beginning of production control that became effectively compulsory in 1992. Through the Common Agricultural Policy (CAP), farmers were encouraged to remove between 5% and 18% of eligible arable land from production in return for Arable Area Payments (AAPs). On a field of set-aside, after harvest, typically a cereal stubble is retained until spring, when a green cover of vegetation has been established by natural regeneration, or by sowing a grass or seed mixture into the soil. The bird habitats provided by set-aside include annual winter stubbles and summer fallows (termed "rotational" set-aside) or ungrazed fallow/grassland retained over two or more years (termed "non-rotational set-aside"; Table 2). Alternative configurations include whole or part field/margin options and wildbird seed crops for the provision of bird food and cover.

In 2000, set-aside occupied 453,000 ha in England (10% of eligible arable land) and 500,000 ha across Britain, and the same rate was set for 2002. Special derogations are also available to provide, for example, cultivation for nest plots for birds such as Stone Curlews *Burhinus oedienemus*, although these may be avoided by the adoption of the management plan option of multi-annual set-aside which became available in 2001.

5.6.1 Main management options that will benefit birds

- The winter provision of seeds and grain stubbles (crop rotations/winter bird crops).
- The summer provision of invertebrate food and weedy nesting cover on fallow land (herbicide control & cutting restrictions).
- Sparse grass-herb swards (multi-annual set-aside).
- Field diversification (winter stubbles, summer fallows).
- Bare/sparsely vegetated nesting plots (through derogation or the management plan option of multi-annual set-aside).

5.6.2 Nature of the benefits to bird species included in the Farmland Bird Index

Set-aside can be incorporated into farmland as either whole or part fields or as 20 m strips (10 m alongside watercourses). Set-aside applies only to "cropped" areas (not boundaries), which we discuss here in a slightly different format to the agri-environment sections above. Principally, set-aside offers food and nesting options to birds and these two benefits are discussed below separately.

On average, set-aside typically supports higher densities of indigenous weed seeds (Wilson 1992, Draycott *et al.* 1997) and broad-leaved plants (Wilson & Aebischer 1995, Hansson & Fogelfors 1998, Critchley & Fowbert 1999) than crops or grassland. Set-aside also tends to support higher densities of invertebrates (associated with the plant community) than either cropped land or pasture (Kennedy 1992, Moreby & Aebischer 1992, Sears 1992). In summer, a wide range of bird species (seed and plant eating bird species, such as Linnet and Turtle

Dove, and species like Grey Partridge, Tree Sparrow or Yellowhammer foraging for invertebrates such as sawfly larvae,) seeking food on farmland will forage in good quality (weedy) set-aside at densities exceeding those on either crops or pasture (Donald & Evans 1994, Wilson *et al.* 1997, Henderson *et al.* 2000). In the breeding season, set-aside can offer breeding opportunities mainly for Lapwing (Wilson *et al.* 2001) and Skylark (Wilson *et al.* 1997). This is because dense winter cereals in April and May are unsuitable or sub-optimal for both species, whereas set-aside potentially provides sparse, low-growing cover in April and May that is preferred by both breeding Lapwing and Skylark (Wilson *et al.* 1997, Wilson *et al.* 2001).

In practice, the content of set-aside is extremely variable (Critchley & Fowbert 1999, Fowbert & Critchley 1999), rendering large areas relatively unsuitable for many bird species (Donald & Vickery 2000, Henderson *et al.* 2000). Two widely adopted practices contribute to this. First, weed-control is frequently carried out in late April, using broad-spectrum herbicides (glyphosate) that kill the green-cover in order to control pernicious weeds that may contaminate the following crop. This practice removes the beneficial plant community for birds, and attendant invertebrate populations (Moreby *et al.* 1994, Potts 1997) at an early stage of the breeding season for species such as Skylark and Corn Bunting. For weeds that are difficult to control, such as Blackgrass *Alopecurus myosuroides*, appropriate herbicides and timing of application, must be allowed to affect successful control. For other potential contaminants of future and adjacent crops, such as thistles or docks, weed-wiping or spot treatments, that are alluded to within the set-aside guidelines, might be encouraged more widely among farmers as affective methods of allowing other less problematic arable weeds to persist into summer. The extent to which set-aside has and can occupy the arable landscape means that emphasis on the late control of weed swards could add large areas of suitable habitat into the countryside with the potential to influence nationally significant numbers of breeding Skylarks as well as species such as Yellowhammer and Corn Bunting that breed late into the summer. Second, annual cuts of longer-term set-aside may create dense swards of dominant grass species that suppress important broad-leaved plants within the sward (Fowbert & Critchley, 1999). A dense sward prevents some bird species gaining foraging access to the ground so reducing the suitability of the composition and structure of set-aside for foraging birds (e.g. Jenny 1990, Wiebel 1998, Henderson & Evans 2000). In addition, cutting of set-aside before the end of July destroys late nesting attempts of Skylarks (Wilson *et al.* 1997). The scheme currently suggests that applications can be made by farmers to leave up to 25% of their set-side cover uncut for up to three years, to encourage structural diversity and this would undoubtedly encourage birds into set-aside throughout the year. It is unclear, however, as to how much land in practice currently benefits from this allowance.

The introduction of multi-annual set-aside to the scheme, as a five-year commitment for landowners, is a positive step that will allow farmers to manage set-aside for longer term benefits (as agreed by DEFRA), so that fallows can be specifically tailored to benefit nesting Lapwing, for example. The cutting of the green cover is banned until 31 July deliberately to protect the nests, offspring and food of species such as Lapwing, Skylark and, to a lesser extent, Grey Partridge. There is evidence that relatively isolated areas of set-aside within a landscape of cereals become a concentrated source of breeding birds that then suffer disproportionately high nest losses through predation (Donald & Vickery 2000). This is partly compensated for by high breeding success in the first place and probably the exact composition of set-aside and the level of cover provided to birds and their nests. It is likely too that larger blocks of set-aside are less likely to assist predators than easily monitored habitats in smaller blocks or strips.

In winter, set-aside can provide seed for birds as stubble fields or as deliberately sown wildbird seed crops. For birds such as Grey Partridge and seed-eating species, such as finches and buntings, weedy stubbles are very important as a source of food over winter (Evans 1997b) to which set-aside has made a major contribution in the last decade (Evans 1997b). This has not yet been reflected by changes in population indices of these bird species since "high quality" stubbles (that is, from a bird perspective) have probably occupied too small a proportion of the arable landscape. As in summer, the content and composition of set-aside are important determinates of their use by birds and studies have found that weed-rich stubbles follows crops of barley are especially valuable to species such as Grey Partridge and Skylark (Buckingham *et al.* 1999). Where possible, recommendations could be made to farmers to design their rotations so that set-aside follows barley and especially spring-sown barely, where this is grown, to increase the likelihood of it attracting winter seed-eating birds.

Alternatively, the green cover of set-aside can be established through deliberate planting of "wild bird" seed mixtures (a set-aside option), sown as wild bird crops (see the CSS arable options) that provide birds with a seed resource in winter. The contribution of these crops to winter bird survival has not been measured, but research has shown that birds such as Grey Partridge, Tree Sparrow, finches and buntings will concentrate on wild bird crops at high densities provided that the correct content of the crop mixes is established (Boatman *et al* 2001, Henderson *et al.* 2001b; see 2.1.3c above). Key crops for mixes include brassicas (e.g. kale), quinoa, linseed and cereals (e.g. oats or triticale). Unfortunately, some of the rules of set-aside can restrict the successful growth of some key crops because crop components must be grown as mixes to prevent the individual contents (e.g. linseed or cereals) from being harvested for profit. Mixes reduce efficient husbandry of the separate component crops that tend to have different management requirements. As such, healthy crops of kale and quinoa, grown in parallel, single-species strips would provide greater benefits to birds (Boatman *pers. comm.*).

5.7 Wholesale Farm Procedure and Good Arable Practice

Outside of agri-environment schemes or set-aside, significant contributions by farmers towards changes in the environmental quality of farmland might be achieved through widespread adoption of Integrated Farming Management (IFM or Integrated Crop Management for arable farms - ICM) promoted by LEAF – Linking Environment And Farming or the Integrated Farming protocol developed by DEFRA.

LEAF is a membership funded non-governmental organisation (with DEFRA grant aid) that sets formal initiatives specifically aimed at integrating economically viable agriculture with environmentally responsible practices in farming. For LEAF members, LEAF will calculate tailored strategies for individual farms for the optimal use of crop rotations, crop protection, pollution control, soil management, energy use, wildlife/landscape, welfare and other parameters. Careful consideration is given to maintaining or increasing production potential, biodiversity and water quality. LEAF membership promotes a farm audit whereby careful recording of practices allow the farmer to evaluate and improve his practices against the standards of IFM. Care of the environment is at the core of IFM that highlights mixed farming, natural regeneration set-aside (fallow land), bird cover crops, protected field margins, hedgerow management and careful timing of operations on crops to avoid destroying bird's nests.

Codes of Good Agricultural Practice (GAP) as set out by the government (DEFRA 2001) are "common sense" guidelines that can also contribute to IFM (LEAF 2000) and are the obligatory basis on which all agri-environmental scheme agreements are based. However, farmers who adopt most GAP principles do so without an audit system, for their own benefit and for the sake of long-term agronomic efficiency through the responsible use of soil, water and air. The principals set out by DEFRA under GAP, and their principles of Integrated Farming (not IFM) aim to "...reduce inputs, deliver environmental benefits and still maintain or even increase profitability". In this way they echo the basic principals of the IFM system as the Government's recognition of the importance of sustainable agriculture for the long-term future of farming. GAP/IF guidelines recommend avoiding: crop treatments that contaminate hedgerows, margins and boundaries; soil compaction, either mechanical or through grazing, and losses in organic material and fertility through soil erosion and deep cultivation. Also, recommended is the careful use of listed pesticides, an avoidance of excessive applications of fertility substances that raise ammonia to damaging levels and a preference towards composted materials such as well rotted farmyard manure.

LEAF membership involves over 707,000 ha of farmland, 1500 farms including 42 demonstration farms.

5.7.1 Main management options that will benefit birds

- Breeding and feeding options for birds (a more diverse crop rotation).
- Increased year round availability of soil invertebrates and plant food in the soil (soil organic content; less reliance in inorganic fertilisers, use of FYM and minimum tillage).
- Increased availability of weed seeds (herbicide control).
- Increased availability of invertebrate food in crops and margins (pesticide control).
- Food and cover (margin management, wild bird crops, grass margins, retention of mature and dead trees and mature hedges).

5.7.2 Nature of the benefits to bird species included in the Farmland Bird Index

Crop rotation/nutrient retention

Central to both IFM and IF is the adoption of a more complex crop rotation that allows a more efficient use of nutrients, reduce weed and pest epidemics associated with follow-on monocultures and so reduce the farmer's reliance on expensive chemical treatments. The use of crop rotations (with animal manures) will maintain crop structural variety, soil organic content and biological activity (e.g. microbial action and earthworms) that may benefit invertebrates and the birds that feed on them (e.g. breeding adults of most species and Lapwing all year (Cramp *et al.* 1977-1994)). In particular, Integrated Farming recommends rotations that contain not more than 25% of any one crop type, preferring mixes of cereals, legumes and root crops. A more diverse crop rotation can benefit birds, principally, by creating variation in crop structure (in height and density), giving birds options for feeding and nesting (Chamberlain & Gregory 1999). Rotations may also lower the risk of pest contamination (especially of nematodes) between sequential crops that reduces the need for chemical treatments that can damage non-pest invertebrates (HGCA 2002).

Two bird species in particular respond to more complex rotations. First, Lapwing require low growth March and April when nesting on open farmland, followed by taller cover for their precocious chicks in May or June. Spring crops or fallows are appropriate but winter crops grow too quickly to provide suitable habitat (Wilson *et al* 2001, Sheldon *pers comm.*). Second, Skylark require low to medium growth between April and June (Wilson *et al.* 1997). Winter crops are adequate in April but are too dense by May. For Skylarks, spring crops allow later nesting attempts to proceed into June and July (e.g. Wilson *et al.* 1997).

Through close and frequent monitoring of crop conditions, farmers can target weed and pest problems efficiently, as they arise, with appropriate applications of chemicals rather than broad-spectrum pre-emptive applications. In theory, this should benefit birds by allowing non-problem arable weeds to persist. In practice, farmers may need convincing that risks to crops and income are reduced and that profitability is unaffected before this option attracts widespread interest among farmers.

Weeds and plant food

Integrated Farming is perhaps more specific than IFM in providing advice relating to problem arable weeds by grouping a set of species according to the levels of tolerance that farmers can give them. Thus, while some plant species such as Blackgrass, Cleavers *Galium aparine* and Wild Oats *Avena fatua* are not tolerated, others, such as Chickweed *Stellaria media*, Fumitory *Fumaria officinalis*, Knotgrass, Field Pansy *Viola arvensis*, Black Bindweed *Fallopia convolvulus* and Annual Meadow Grass *Poa annua* might be tolerated under certain circumstances or within some crops (HGCA 2002). Certainly, residual levels of non-aggressive weeds such as Field Pansy, Knotgrass and Fumitory, are likely to benefit seed and plant eating species such as Turtle Dove (Browne & Aebischer 2001) and Linnet (Wilson *et al* 1996). They provide food for Grey Partridge, Skylark, Tree Sparrow, Yellowhammer and Corn Bunting either directly or indirectly by supporting invertebrates (such as larval sawflies and lepidoptera) on grasses and herbs (e.g. Moreby *et al.* 1994). Another recommendation of Integrated Management is the implementation of unsprayed Conservation Headlands (see CSS above), to increase the availability of arable weeds for birds (Potts 1997) away from the main productive area of the crop. Recommendations to select herbicides carefully, to apply them as problems arise or to apply then as spot or wipe treatments of problem thistles or docks will also help non-problem arable weeds to persist within and around crops.

Pests and invertebrate food

There is scope within the treatment of crop pests under Integrated Farming to adapt conditions to allow some invertebrates to thrive. For example, by not cultivating set-aside in July and August the retained green-cover on set-aside can reduce wheat bulb fly problems (because eggs are laid on bare soil) in the following crop. Weed cover on set-aside in July and August would also help the survival of young Grey Partridge and Skylark, as well as provide food and nest shelter for late breeding species such as Corn Bunting. On crops, IF also recommends using narrow-spectrum pesticides to attack target pests without harming residual wildlife in the crops, margins or on set-aside. Further recommendations emphasise that by monitoring pest problems and establishing pest thresholds, farmers can judge when to take appropriate preventative action against pests in preference to employing pre-emptive measures that may not always have been necessary. All improvements in precision, monitoring and application, will tend to help maintain non-target invertebrate populations on which birds may feed.

On heavier land, where slug damage is a particular problem, carbamate molluscicides can be damaging to other invertebrates such as earthworms (Jenkins 1984). To reduce potential effects on non-target invertebrates, the Integrated Farming protocol suggests the deep drilling of cereals (>4 cm) which apparently is effective in reducing slug damage to seeds (HGCA 2002). The Link Integrated Farming Systems HGCA project report (no. 173, 2000) of ICM, showed that the limited reductions of pesticides and fertilisers, of around 20%, had little effect on wildlife populations, with no significant difference in the numbers of invertebrates (beetles, spiders and earthworms) recorded between conventional and ICM sites after five years. If only large scale reductions in pesticides are required before significant improvements in invertebrate levels are recorded within crops requires quantitative research, but current information has shown no long term effects on invertebrate populations (Greig-Smith *et al.* 1992).

5.8 Conclusions

The main conclusions of this chapter derive from Tables 4 and 5 in which we relate the resources required by farmland bird index species with those provided by agri-environment schemes and farming practices. These tables highlight the fact that there are actually rather few 'gaps' in resource requirements provided. However, there is a stark contrast between those relatively few options that will *definitely* deliver benefits and a large number that could potentially, but maybe not always, deliver benefits for farmland bird index species.

In other words the scope of current options/management practices within agri-environment schemes and farming more generally could *potentially* meet most of the needs of most of the farmland bird index species, but do not always do so. The failure to realise this potential is related to two key issues. The first is the scale at which these options are deployed – the 'quantity' issue. The second is the precise nature of the habitats provided by the options – the 'quality' issue.

5.8.1 The quantity issue

For most species, we can broadly list what habitat types or management options might be beneficial, but we are still lacking, in most cases, information on 'how much resource is enough'. For example, we know that the loss of seed/weed rich foraging habitats in winter is likely to have been an important factor underlying the decline of many farmland birds. However, we have very little information about how much of this habitat needs to be created to enhance over winter survival and thus breeding populations on a local or national scale (Bradbury *et al.* 2001). Can we begin to answer the question 'how much is enough?'

First, we know that, in some cases, suitable quantities of resources have almost certainly been lacking. For example, many species depend on sympathetic management of arable habitats which have, to date, been under-represented in agri-environment schemes. Most ESAs are in grassland areas and only six (South Downs, Breckland, Clun, South Wessex Downs, West Penwith and the Cotswolds) contain options for the sympathetic management of arable land. In each case the option is for the management of arable field margins, with whole-field options, such as stubbles or spring sown cereals, available for only four ESAs (South Downs, Breckland, West Penwith and the Cotswolds; Clun and South Wessex Downs have options for conservation headlands only) whole-field arable options. It is probable, therefore, that there has, to-date, been too little of these habitats for those species that depend on them (e.g. Skylark, Tree Sparrow and Corn Bunting). To some extent, this gap will be filled by the inclusion of several of the ASPS options in CSS from 2002 onwards. The speed and extent to which these arable options are taken up will be extremely important in determining their effectiveness in reversing the downward trend of the farmland bird index. However, initially they are being carefully targeted to areas known to have current or very recent presence of arable species, including seven widespread farmland birds.

Although there are few data relating to the issue of 'how much is needed' two lines of evidence suggest it may be considerable. First, the recovery of the Cirl Bunting population in south west England is an important 'success story' for agri-environment schemes. However, it involved providing extensive areas of weedy, over-winter stubbles (for foraging in winter) and grass margins (for nesting and foraging in summer) within quite a restricted area. Between 1992 and 1998 Cirl Bunting numbers had increased from ca. 300 to ca. 450 breeding pairs, albeit close to source populations and where nesting habitats were already present. At the same time the area of land under CSS agreement rose from 200 ha to 1300 ha with an increase from 0 to 300 ha of grass field margin (23% of CSS land) and 10 to 700 ha (54% of CSS land) of spring sown barley. It is difficult to relate this to a landscape scale because the proportion of arable land under CSS agreement is not known. However, the extent of grass margins and stubbles is obviously high compared with the national scale where only 23% of cereal stubbles remain to midwinter, equating to ca. 3 ha of cereal stubble per 1-km square (Gillings & Fuller 1999). Furthermore, on average, these figures are likely to be even lower for grassland areas.

Second, there is evidence that very large areas of suitable habitat may be needed to turn around population declines of some species. For example, a number of studies have shown that set-aside can provide important habitat for Skylarks in both summer and winter (e.g. Poulsen *et al.* 1998, Henderson *et al.* 2000). Yet, despite being widespread for almost eight years and covering at around 10% of eligible arable land on average in that time (a total of around 750,000 ha at the peak in 1994), there is very little evidence to suggest that set-aside has helped reverse Skylark declines (Browne *et al.* 2000, Vickery & Buckingham 2001). The levels of set-aside did fluctuate annually (from 5% to 15% of arable land since its introduction) and a large proportion may have been unsuitable for nesting Skylarks, either as a result of the early cutting dates permitted pre-1994 or unsuitable vegetation structure (Henderson *et al* 2001a). However, the failure of this large-change in the arable landscape (Evans 1997a) to reverse Skylark suggests that high quality agri-environment incentives will need to cover considerable areas of land to benefit birds at the population level.

An indication as to the extent to which areas of land may have to be managed sympathetically for farmland birds is also derived from the preceding chapters of this report. Studies of Grey Partridge suggest population recovery (to 1996 level) depends on managing 6 km/km^2 (3.6% by area for 6 m margins) as nesting cover and 3% of arable land as insect rich habitat. Currently between 3% and 4% of CSS agreement land (up to 150,00 ha) is occupied by margins, which accounts for less than 0.2% of the total arable area of England and Wales. Increased contributions will therefore be required from within agri-environment schemes (arable options) and from across the wider farming landscape.

The bird-habitat relationship modeling provides further insights into the scale of habitat change required. For example, a 1% increase in Skylarks in arable farmland is predicted to require 2.15% of farmland to be converted to set-aside, or 3.95% of farmland to be converted to organic farming, or 1.69% of cereals to be converted from autumn to spring sown. Similarly, for Yellowhammers in arable farmland, an increase in breeding pairs of 1% could be achieved by adding 2 m grass margins to 5.15% of all field boundaries or 6 m grass margins to 1.39% of boundaries.

The quality and quantity issues are, of course, related and the same results may be achieved through managing small areas of 'high quality' land for birds or large areas of 'lower quality' (where quality refers to level of resources provided for birds) (cf. the Policy Commission on the Future of Farming and Food: Curry 2002). Curry (2002) recommend that existing schemes, which are the best way to target specific tailored prescriptions at particular areas of special value, should form the upper tiers of a single agri-environmental scheme, with lower tiers aimed at all land managers but with lower payments with less demanding prescriptions as part of an environmental plan. Monitoring of the farmland bird index will from 2001, be based on the national BTO/RSPB/JNCC Breeding Bird Survey (BBS), which means that over 1000 farmland 1-km squares may need to be affected by management change if the indicator is to respond in a meaningful way.

5.8.2 The quality issue

A striking feature of Tables 4 and 5 is the extent of 'grey' options. These are defined as options that could *potentially* increase food availability in summer or winter and/or provide better nesting habitat, but where the outcome is not certain. This is particularly true for grassland prescriptions for two reasons. First, the impact of aspects of grassland management on birds is complex and far less well understood than the impacts of management practices on birds in arable systems (Vickery *et al.* 2001). This alone makes predictions difficult. Second, the prescriptions within CSS and lower tiers of many ESAs may not be 'specific enough' to guarantee that they will deliver these benefits. For example, within ESAs, the management of extensive or permanent pasture is a common option (part of 21 of 22 ESAs). Often the prescriptions relate to grazing and cutting regimes and nitrogen inputs. In the Lake District ESA, for example, tier 1 agreements limit stock numbers on inbye land to within certain bounds (within 0.5 to 2 (normally 1.0-1.5) livestock units per ha) to protect grassland from over or under gazing or poaching (Robinson *pers comm.*). Likewise, in the Pennine Dales ESA the lower tier options on allotment land (mainly rough grassland, heath or dwarf scrub) restrict sheep grazing in spring and summer (2.5 ewes/ha), and ban fertiliser inputs to create (tussocky) conditions for ground nesting birds (waders such as Snipe and Curlew, or Black Grouse in northern areas). In both cases actual implementation of prescriptions varies according to negotiations (and therefore the relationship) between project officer and farmer. It has been suggested that farmer motivation and expertise may be a factor in the poor delivery of environmental benefits from some schemes (Kleijn *et al.* 2001). In many cases environmental management designed on the basis of local conditions may yield more sustainable results and encourage regional habitat heterogenity (Robinson & Sutherland in press) but the flexibility in these lower tiers mean that significant benefits for birds are not guaranteed.

Less flexibility is available to farmers who agree to manage land under higher tier options within the ESA scheme. For example, in the Lake District, precise restrictions ban cutting on meadows before 1 July in most years (except under special derogation) or before 22 July in every 5th year to allow meadow plants to seed. These restrictions would successfully protect nesting lapwings on larger fields. In the Pennine Dales, only very light grazing is allowed in spring and early summer on inbye land that is managed under tier 2 (pasture). This habitat would be more suitable for breeding lapwing than lower tier allotment land. Finally, in the

South Downs, where grazing restrictions on pastures at all tier levels would benefit ground nesting birds, only at higher tiers is rolling or harrowing banned between 31 March and 1 July despite the obvious impact that these activities would have on bird nests or broods. Unfortunately, the tighter management restrictions associated with important higher tier options tend to reduce uptake among farmers (e.g. Topham *pers comm.*). However, in order to enhance ESA grassland for birds in the medium term (i.e. change grey squares to black in Tables 4 and 5), it would be helpful for more farmers to be encouraged to engage higher tier options, perhaps through higher financial incentives in a manner similar to special Managed Habitats.

The importance of habitat quality is also evident from studies of Skylarks in relation to vegetation structure on set-aside land. Breeding Skylarks show a clear preference for patchy swards with ca 25% bare ground, and density of pairs is lower on fields with more or less than this optimum (Henderson *et al* 2001a). Studies of Lapwing breeding on ASPS option 1B (fallow) land suggests similar preferences for more open, patchy swards (Rob Sheldon 2002 & unpubl data) structures. Thus, although in *theory* fallow land options will deliver benefits for birds in practice many such areas may fail to do so because they require active management to maintain them in early succession. Similarly, Linnets are known to show pronounced preference for stubbles in winter (Wilson *et al.* 1996), and yet not all stubbles are used. Recent work (Moorcroft *et al.* in press) suggests that the quality of the stubble to linnets and other passerines is determined by a combination of food abundance and sward structure that allows access to the food. To some extent the same is true of wildbird cover crops. Recent studies have shown that the numbers of birds these crops support is highly dependent on the seed mix used. Crops like second year kale and quinoa support good numbers of a wide range of species whereas other crops may be particularly beneficial to specific species, for example canary grass for Yellowhammers (Henderson *et al.* 2001b).

There are four general approaches to increasing delivery of the habitat types created. The first is to ensure a greater scientific input to development of the schemes themselves as in the case in Arable Stewardship (Whitfield 2001; Evans *et al.* in press). The second, is rigorous site inspection/supervision, as in the case of Cirl Buntings, but this is probably only an option for scarce or restricted range species. The third is to pay farmers for what they produce, such as the numbers of breeding pairs of birds, as has been suggested in the case of meadow birds in the Netherlands (Musters *et al.* 2001). This may be difficult to administer in practice but would probably lead to an improved composition and structure of grassland. The fourth is to develop 'tighter' management prescriptions or goals based on the habitat characteristic desired, but to retain the flexibility to let tailored options continue to evolve as new research comes to light (Robinson & Sutherland in press, Evans *et al.* in press). Within ESAs this may be achieved through promotion of options within the higher tiers that actively improve habitat rather than simply maintaining the status quo. Benefits may also be maximised by more careful advanced targeting of the options adopted by farmers (through discussion with the project officer), as in the LEAF approach. Whole farm environmental plans or audits in advance of entry into a scheme may maximise the cost-effectiveness of management and are part of the basic requirements of CSS recommendations. For example, it would allow a consideration of which species may benefit most in relation to the location of the farm itself or the positioning of options within the farm (Lapwings will not nest near woods). It would also help realise the potential of apparently poor areas of farmland (for biodiversity) to support new populations of target bird species in future, as they may have done in the more distant past.

5.8.3 Gaps in resource provision for birds

As already stated there are, in fact, rather few key resource requirements for the farmland bird index species birds that cannot *in principle* be met within existing schemes/practices. At the farm scale, arguably the most important option that could be considered in the future is the provision of wet features such as ditches or small waterbodies within arable systems. These appear to be important habitats for a number of species including Song Thrushes, which select damp areas such as wet ditches for foraging (Will Peach pers comm.) and Reed Bunting, which select such habitats for nesting (Nick Brickle pers. comm.). In addition, Yellowhammers choose ditch banks (whether wet or dry) as nesting sites (Bradbury *et al.* 2000) and key invertebrate food groups (primarily those with aquatic larvae) could be provided for species such as Tree Sparrows and Yellow Wagtails. There are currently options for ‘wet ditch or dyke’ management within a small number (five) of ‘wet’ grassland habitat ESAs (e.g. Broads, Somerset Levels and Moors, Essex Coast and Upper Thames) but none within arable systems where they may also be important for foraging Lapwing adults and chicks (Wilson *et al.* 2001).

The use of harvest tailings to supply a source of grain to birds in winter are not part of the agri-environment schemes even though they have been shown to be used by several species such as Corn Bunting and Tree Sparrow (R. Bradbury pers. obs.). There is also currently no mechanism for encouraging grazed areas within arable dominated landscapes, or for replacing silage crops with fodder crops in grass-dominated areas. Both methods encourage mixed-farming that is associated with higher levels of bird biodiversity than most monoculture systems (Robinson *et al.* 2001, Benton *et al.* in press).

There must remain some doubts over the ability of AES and other current opportunities to provide seed, and perhaps also invertebrate, food in late winter and early spring. In the past this may have been obtained, for example, from sources around farmsteads and livestock feeding areas which are no longer commonplace. This period, leading up to and including breeding, is a time when birds may benefit greatly from good food supplies. The provision of stubbles through to late March may help as will cover crops/seed mixtures and the practice of spring cropping. Seed rain from arable weeds that germinate and set seed throughout winter can replenish food resources in stubble habitat to a considerable extent (Robinson 2002). However, there is still a potential issue that, unless stubble is allowed to remain as summer fallow, then these food resources are suddenly removed during ploughing in late winter/early spring (of crops and perhaps also game cover). Thus current resource provision may still result in a bottleneck in provision of seed food for birds by AES in late winter/early spring.

5.8.4 Landscape issue or variation in time and space

At a larger scale, there is growing evidence that habitat diversity, particularly the mixture of grass and arable, is extremely important in winter and summer. The creation of arable pockets within grassland systems is likely to enhance numbers of breeding birds (Robinson *et al.* 2001) and the reverse may also be true (Siriwardena *pers comm.*). For example, Swallows select grazed grass for foraging and may benefit from pockets of pasture within arable systems (Evans 2001). Lapwings do best when arable and grass habitats (for nesting and chick foraging habitat respectively) are juxtaposed (Galbraith 1988, Wilson *et al.* 2001). Most agri-environment schemes increase habitat diversity at a farm scale but less so at the landscape scale. ESAs covering discrete areas may go some way to achieving this but it is perhaps less likely within CSS or farming approaches such as organic and IFS unless uptake is promoted in specific regions.

It may also be possible to maximise benefits derived from options by targeting regions for certain options. Thus creation of stubbles, cover crop and spring cereals could be actively promoted in areas supporting restricted range species such as Tree Sparrow and Corn Bunting. For other species it may be beneficial either to target areas within the centre of their range or areas where distributions of many species overlap.

Finally, for highly sedentary species, it may be paramount to provide both winter and summer resources in close proximity to each other. This is illustrated by the case of the Cirl Bunting, where the delivery of Countryside Stewardship Scheme agreements was greatest when (a) the agreement provided both summer (rough grass) and winter (stubble) habitat, and (b) was provided next to an existing source of Cirl Buntings (Peach *et al.* 2001). In the case of rural House Sparrows, it appears that the species is so sedentary that summer and winter resources must be provided at the same location in order to allow population persistence (Hole 2002).

Priorities for action

The section highlights some clear priorities if the future development and implementation of agri-environment schemes and farming practices are to successfully reverse the decline of the farmland bird index by 2020. The list below is generic and not definitive, requirements will change as farming changes and the advent of new technologies (e.g. Watkinson *et al* 2000) will have an impact on future requirements. However, the points below provide a basis for discussion in the final concluding chapter of this report.

- Actively promote arable options to manage margins (grass/herb, wildbird cover, conservation headlands etc.), stubbles and nest plots, fallows and wildbird crops over a **higher proportion of farmland**.
- Ensure habitats created/managed under agri-environment schemes are of **high quality** for birds e.g. promotion of 'tighter' management prescriptions e.g. higher tiers in ESAs to deliver high quality habitats.
- Maximise benefits by **geographical targeting** of options geographically.
- Regionally plan uptakes to create **habitat heterogeneity** within and between farm spatial scales, and to promote crop rotations and mixed farming (currently only in organic systems).
- Carry out research on **knowledge gaps**. In particular, to determine how much of specific habitats are required and at what spatial scale and more specifically what is required by birds associated with pastoral systems.
- No one measure is a panacea for farmland birds, thus a suit of measures in different landscapes is needed which meets the needs of nesting birds and provides year-round food requirements for different species.
- Time lags between habitat changes and responses by bird populations will mean that agri-environmental measures will improve with time but need to be implemented urgently.

Species	Code	Population estimate	Long-term trend: CBC 1968-98	Recent trend: BBS 1994-1999
Kestrel *Falco tinnunculus*	K	50,000	-5	-30
G. Partridge *Perdix perdix*	P	150,000	-84	-43
Lapwing *Vanellus vanellus*	L	200,000	-34	-20
Stock Dove *Columba oenas*	SD	24,000	+157	+10
Woodpigeon *Columba palumbus*	WP	2,100,000	+86	0
Turtle Dove *Streptopilia turtur*	TD	75,000	-69	-18
Barn Owl *Tyto alba*	BO	4500		
Skylark *Alauda arvensis*	S	1,000,000	-53	-16
Yellow Wagtail *Motacilla flava*	YW	50,000		
Whitethroat *Silvia communis*	WH	660,000	-57	+6
Jackdaw *Corvus monedula*	JD	390,000	+72	+21
Rook *Corvus frugilegus*	RO	855,000		+8
Starling *Strurnus vulgaris*	SG	1,100,000	-70	+6
Tree Sparrow *Passer montana*	TS	110,000	-95	+11
Greenfinch *Carduelis chloris*	GR	530,000	+4	+20
Goldfinch *Carduelis carduelis*	GO	220,000	+9	+1
Linnet *Carduelis cannabina*	LI	520,000	-59	-14
Yellowhammer *Emberiza citrinella*	Y	1,200,000	-54	-16
Reed Bunting *Emberiza schoeniclus*	RB	220,000	-49	+5
Corn Bunting *Milaria calandra*	CB	160,000	-83	-26

Table 1 Population estimates and temporal trends for 20 bird species in the farmland indicator. CBC = Common Birds Census, BBS = Breeding Bird Survey. The species codes here are used in the tables of this report.

	Countryside Stewardship Agreements										**Tir Gofal**
	1. Chalk & Limestone	**2. Lowland Heath**	**3. Old Meadows**	**4. Field boundaries**	**5. Countryside Around Towns**	**6. Orchard**	**7. Waterside**	**8. Uplands/ moorland**	**9. Coastal strip**	**10. Pilot Arable Options**	
Coverage area / length	19,327 ha	13,159 ha	No data	10,000 km	No data	1,059 ha	No data	133,436 ha	7,502 ha	No data from reports	60,000 ha
Margins	■									■	■
Beetle banks										■	■
Conservation headlands	■									■	■
Wild bird mix										■	■
Spring crops										■	■
Stubbles										■	■
Fallows										■	■
Undersowing											▨
Mixed cropping										■	■
Minimum inputs	■						■			■	■
Soil organic content											
Arable reversion	■								■		■
Extensive Pasture	■	■	■				■	■	■	■	■
Wet grassland							■	■			
Hay meadows			■			■	■	■	■		■
Extensive silage											
Grassland margins	■										
Hedgerows	■			■	■			■			
Scrub											■
Trees	■			■	■	■	■	■	■		
Old trees											
Ditches	■			■			■				■
Buildings	■										■
Nest sites						■					■

Table 2 Habitat management matrix for the Countryside Stewardship Scheme in England and Tir Gofal in Wales. Figures are given where data were available for the approximate areas or lengths of countryside managed under each agreement. Filled squares denote the presence of that habitat category within the management remit of the scheme. Main source: DEFRA 2001.

	North				West		East				
	1. Lake District	**2. Pennine Dales**	**3. North Peak**	**4. South West Peak**	**5. Clun Hills**	**6. Shropshire Hills**	**7. Breckland**	**8. The Broads**	**9. Suffolk River Valleys**	**10. Essex Coast**	**11. Nth Kent Marshes**
Total area:	245000	46500	55000	33800	21000	38500	94000	36000	44000	28600	13700
Eligible:	240000	46000	44000	30000	14000	32000	8000	24000	33000	26000	12000
Agreement	110000	25000	21000	15000	10000	14700	5500	8700	8000	3200	3300
% uptake:	44	54	48	51	71	46	69	36	24	12	27
Arable management											
Field margins							234	6			
Beetle banks											
Conservation headlands					0		234				
Wildbird cover											
Spring crops											
Stubbles							●				
Fallows											
Undersowing											
Arable crops											
Mixed cropping											
Min. inputs	3000	20000	4000	14000	4000	2446	5000	15000	7300	3000	2700
Reversion					215		127	275	539	465	663
Grassland management											
Ext. pasture	50000	20000	4000	14000	4000	9000	2600	15000	7300	3000	2700
Wet grassl.	●						2600	6600	●	●	●
Hay meadows	600	1237		2500		6800		6000	●	550	616
Ext. silage	●	5400	●	●	●	●			●	●	
Boundaries & buildings management											
Hedgerows	●	●	●	●	●	●	●	●	●	●	
Scrub	●					●			●		
Woodland	●		●	●	●		●	●	●		
Tree		●	●	●			●	●	●		
Ditch				●	●		●	●	●	●	●
Buildings	●	●	●	●	●		●				
Summary objectives and performance (increase(+), decrease (-), stable (+/-))											
ESA key aims for maintenance/ enhancement:	Heather; enclosed pasture; wetlands	Meadows, pastures & rough grazing	Uplands, meadows, pastures	Uplands, meadows, & pastures	Grassl.; Arable margins	Grass & moor	Chalk grassl.; Arable margins	Perm/wet grassland; Hedges & ponds	Valley perm./ wet grass.	Perm. grass, wet marsh	Grazed marshes; Rever-sion
ESA target spp. (among indicators)		L, YW	L, S, YW.	L., S.		L, S.		L, YW		Waders, L.	L.
Resulting changes in habitat.	S-nat. grassl. (+/-)	Managed grassl. (+)	Hedgerow (+)	Meadow uptake (+)	Grassl. (+/-)	Grassl. (+/-)	Grass-heath (+/-)			Some reversion	Low uptake
Resulting changes in bird numbers		YW (+/-) Waders (+/-)	Species popls. (+/-)	Waders (-)				Grassland spp. (+?)		Winter birds (+/-)	L. popl. (+/- or -)

Table 3 Habitat management matrix for English ESAs. Figures show the approximate area managed under each habitat category while ● denotes the presence of that category in that ESA where the area involved cannot be quantified (main source: Ecoscope 2001). The management objectives of each ESA and their known impact on indicator bird species is summarized in the final four rows. Species abbreviations are L = Lapwing, S = Skylark and YW = Yellow Wagtail, RB = Reed Bunting).

	S & w Midlands		South			South-west					
	12. Upper Thames	13. Cotswold Hills	14. South Downs	15. Test Valley	16. Avon Valley	17. South Wessex Downs	18. Somerset Levels	19. Exmoor	20. Blackdown Hills	21. Dartmoor	22. West Penwith
Total area:	27200	85700	69000	4700	5200	45000	27000	80000	39000	100275	7600
Eligible:	27000	62000	50000	3100	3800	40000	23800	60000	25300	80000	6800
Agreement:	3400	30000	11000	974	850	20000	11000	44000	9000	21000	6100
% uptake	13	48	22	31	22	52	48	65	35	27	90
Arable management											
Field margins						•					
Beetle banks											
Conservation headlands			61			30					
Wild bird cover											
Spring crops											
Stubble		•	•								•
Fallows											
Undersowing			•								
Mixed crops											
Min. inputs	5000	8000	10000	821	1000	19000	12000	24000	4000	5900	6000
Grassland management											
Reversion	570	1870	6000	153	27						
Ext. pasture	5000	8000	10000	1200	1000	19000	12000	24000	4000	5900	3000
Wet grassland	415		467	•	•						
Hay meadows		•	10000	•		2700	2400	•	103	137	
Ext. silage	•	•		•	•			•			
Boundaries & Buildings management											
Hedgerow	•	•			•	•	•	•	•	•	•
Scrub		•	•		•	•	•		•	•	
Woodland		•	•	•			•				
Tree	•	•		•	•		•		•		
Ditch	•			•	•	•	•		•		
Buildings		•	•	•		•		•	•	•	•
Summary objectives and performance (increase(+), decrease (-), stable (+/-))											
ESA key aims for maintenance/ enhancement:	Perm. & wet grassl.	Limestone grassl; Hedges; Reversion	Chalk & valley grass; Arable margins	Valley & wet grassl.	Grassl. & wet grassl.	Perm. grassl.; Arable margins	Grassl. & wet grassl.	Moor & Pasture.	Perm. grassl. & spp-rich meadows.	Unimpr. grassl. & hay meadows.	Rough grassl.
ESA target spp. (among indicators)	L.		Botanical		L.		L. (YW, RB?)				
Resulting changes in habitat.	Wet grassl. (+)	Grass (+/-), hedges (-), reversion low.	Grassl. (+) Margins (+)	Rever-sion. (+)	Grassl. (+/-)	Rever-sion (+) Margins poor	Grassl., 60% (+). Wet grassl (+)	Grassl. (+/-)	Uptake good. Low hedgerow uptake.	Pasture (+), hay meadows 69% (+).	Grassl. (+/-)
Resulting changes in bird numbers					L (+/-)		Breeding spp (+/-)	Bird popls. (+/-)			

Table 3. Continued.

Scheme	Sub-scheme	Management options	Declining indicator species in summer (s) and winter (w)																
			P		L		TD	S		SG		TS		Y		RB		CB	
			S	W	S	W	S	S	W	S	W	S	W	S	W	S	W	S	W
CSS	Chalk & Limestone	Grazing			▨			▨		▨	▨								
		Margins*	■	■			▨					▨		■	▨	▨	▨	■	▨
		Reversion						▨										▨	
	Lowland Heath	Grass-heath.	▨	▨				▨											
		Reversion						▨											
	Waterside Landscape	Meadows			▨	▨				▨	▨					▨		▨	▨
		Willows/reeds														■		▨	
	Coastal	Meadows	▨		▨	▨		▨	▨	▨	▨								▨
		Marshes			■	▨		▨	▨	▨	▨								
	Uplands & Moorland	Hay meadows	▨		▨			▨		▨	▨			▨				▨	▨
		Inbye land	▨		▨			▨		▨	▨								
	Old Meadow & Pasture	Grazed			▨			▨	▨	▨	▨								
		Spp-rich	▨	▨	▨			▨	▨	▨	▨							▨	
	Field boundaries	Hedgerows					▨					▨		■					
		Ditches										▨				▨	▨		
		Trees					■					▨							
	Orchards	Old trees								■		■							
		Wildflowers	▨								▨			▨					
	Arable Options	Margins*	■	■			▨					▨		■	▨	▨	▨	■	▨
		Cover crops*		■									■		■		■		■
		Stubbles		▨		▨					■		■		■		■		■
		Fallows	▨		■		■	■		▨		■		■		■		■	■
		Spring crops	▨		▨			▨										■	
ESAs		Ext. pasture			■	▨		▨	▨	▨	▨								
		Hay	▨		▨			▨	▨	▨	▨				▨			▨	▨
		Wet grass			■	▨		▨		▨	▨	▨							
		Margins	■	■			▨	▨				▨		■	▨	▨	▨	■	▨
		Con. Headl's*	■				▨												
		Hedges					▨							■					
		Trees					■			▨		▨							
		Ditches										▨				▨	▨		
Organic farming		Mixed crops			■		▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨
		Leys	▨		▨			▨	▨	▨	▨								
		Boundaries					▨							■					
		Margins*	■	■			▨				▨	▨		■	▨	▨		■	▨
Tir Gofal	Extensive grassland				▨	▨		▨	▨	▨	▨								
	Grassland restoration				▨	▨		▨											
	Hay meadows		▨		▨			▨	▨	▨	▨							▨	▨
	Wet grassland				■	▨		▨		▨	▨								
	Unsprayed crops		▨				■	▨	▨	■	■			▨	▨	▨	▨	▨	▨
	Stubbles			▨		▨			■		▨		▨		▨	▨	▨	▨	■
	Margins*		■	■			▨					■		■	▨	▨	▨	■	▨
	Cover crops*			■									■		■		■		■
	Hedgerows						▨					▨		■					
	Scrub						▨							▨		▨			
Set-aside	Fallows		▨		■		■	■		▨		■		■		■		■	
	Stubbles			▨		▨			■		▨		■		■		■		■
	Cover crops*			■									■		■		■		■
Integrated farming	Rotations		▨		▨		▨	▨											
	Soil (cultivation)		▨		▨	▨		▨		▨		▨	▨	▨	▨	▨	▨	▨	▨
	Weeds		▨	▨	▨		▨	▨				▨	▨	▨	▨	▨	▨	▨	▨
	Pest control		▨		▨			▨		▨		▨		▨		▨		▨	

Table 4 Potential association between the habitats managed under Countryside Stewardship (CSS), the English ESAs, Organic Farming, Tir Gofal, Set-aside and Integrated Farming guidelines and nine declining species on the DEFRA headline indicator list (for abbreviations see Table 1). Black squares = habitats highly likely to be beneficial; Grey squares are habitats that are potentially beneficial but where the specific nature of the habitat will be critical (e.g. for Lapwing, the suitability of grazed land depends on the type of livestock and the grazing regime adopted); white squares indicate that knowledge of the habitat is poor for that species and that major benefits are uncertain or unlikely, or that the habitat is inappropriate. N.B. *Margins = grass or grass/herb margins; Cover crops = wild bird mixes; Cons. headl's = conservation headlands.

Table 5 Association between habitats managed under Countryside Stewardship (CSS), the English ESAs (ESA), Organic Farming (OF), Tir Gofal (TG), Set-aside (SAS) or Integrated Farming (IF) and bird species on the DEFRA farmland indicator list (for abbreviations see Table 1). Black squares are habitats that are highly likely to be beneficial; Grey squares are habitats that are potentially beneficial but where the specific nature of the habitat is critical (e.g. for Lapwing using grazed land, this depends on the type of livestock used and the grazing regime adopted); white squares indicate that either knowledge of the habitat is poor for that species and that major benefits are uncertain or that the habitat is inappropriate. *Margins = grass, grass/herb margins; Cover crops = wild bird mixes; Cons. headl's = conservation headlands.

Habitat	Management	Indicator species																																				
		K		P		L		SD		WP		TD	BO		S		YW	WH	JD		RO		SG		TS		GR		GO		LI		Y		RB		CB	
		S	W	S	W	S	W	S	W	S	W	S	S	W	S	W	S	S	S	W	S	W	S	W	S	W	S	W	S	W	S	W	S	W	S	W	S	W
Cons'. headl'ds	CSS,ESA, TG, IF			■																											▒						▒	
Beetle banks	CSS	▒	▒	▒									▒	▒	▒																							
Margins	CSS,ESA,TG OF	■	■	■	■							▒	■	■			▒								▒		▒	▒	▒	▒	▒	▒	■	▒	▒	▒	■	▒
Cover crops	CSS, TG, SAS				■				▒		▒											▒				■		■		▒		■		■		■		■
Spring cereals	CSS,OF, IF			▒		▒									▒				▒		▒																■	
Stubbles	CSS,OF, TG SAS, IF		▒		▒				▒		▒					■						▒				■		■		■		■		■		■		■
Fallows	CSS,OF, TG, SAS, IF	▒		▒		■		■		■		■	▒		■		■		■		■		▒		■		■		■		■		■		■		■	
Spray controls	CSS,ESA,OF, TG, SAS, IF			▒	▒	▒						■			▒	▒	▒		▒		▒		▒		▒	▒	▒	▒	▒	▒	▒	▒	▒	▒	▒	▒	▒	▒
Organic content	OF			▒		▒	▒								▒		▒		▒	▒	▒	▒	▒	▒														
Mixed farming	CSS,TG, OF	▒	▒		▒	▒	▒					▒			▒	▒			▒	▒	▒	▒	▒	▒		▒		▒		▒		▒		▒		▒	▒	
Leys	OF			▒		▒									▒				▒		▒		▒	▒														
Ext. pasture	CSS,ESA, TG	▒	▒			■	▒		▒		▒				▒		▒		▒		▒		■	▒													▒	
Hay meadows	CSS,ESA, TG	▒	▒	▒		▒							▒		▒		▒										■	■			▒						■	▒
Wet pasture	CSS,ESA, TG					■	▒						▒	▒	▒		▒						▒	▒	▒												▒	
Grazing marshes	CSS,ESA, TG					■	▒						▒	▒	▒	▒	▒		▒		▒		▒	▒								▒			▒	▒		
Hedges	CSS,ESA, TG,OF									▒		▒						■							▒	▒					■		■					
Scrub	TG									▒		▒						■							▒						■		■					
Trees/Old trees	CSS,ESA OF	■						■		▒		■	■						■				■		■		▒		▒									
Ditches	CSS,ESA, TG,OF																▒								▒										■	▒		

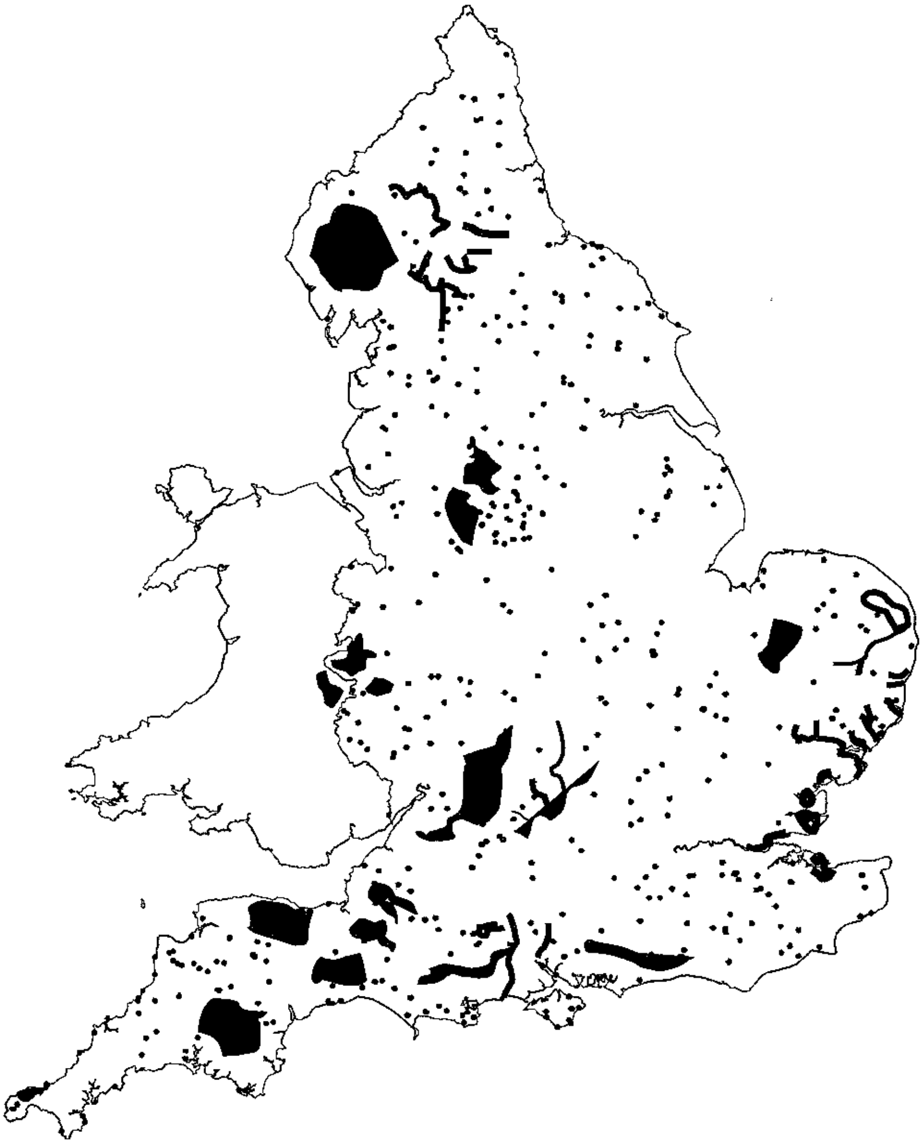

Figure 1. The distribution, in England, of Environmentally Sensitive Areas (labeled according to Table 2), and Countryside Stewardship Plots from a sample of 400 agreements. Note ESAs and CSS sites are mutually exclusive.

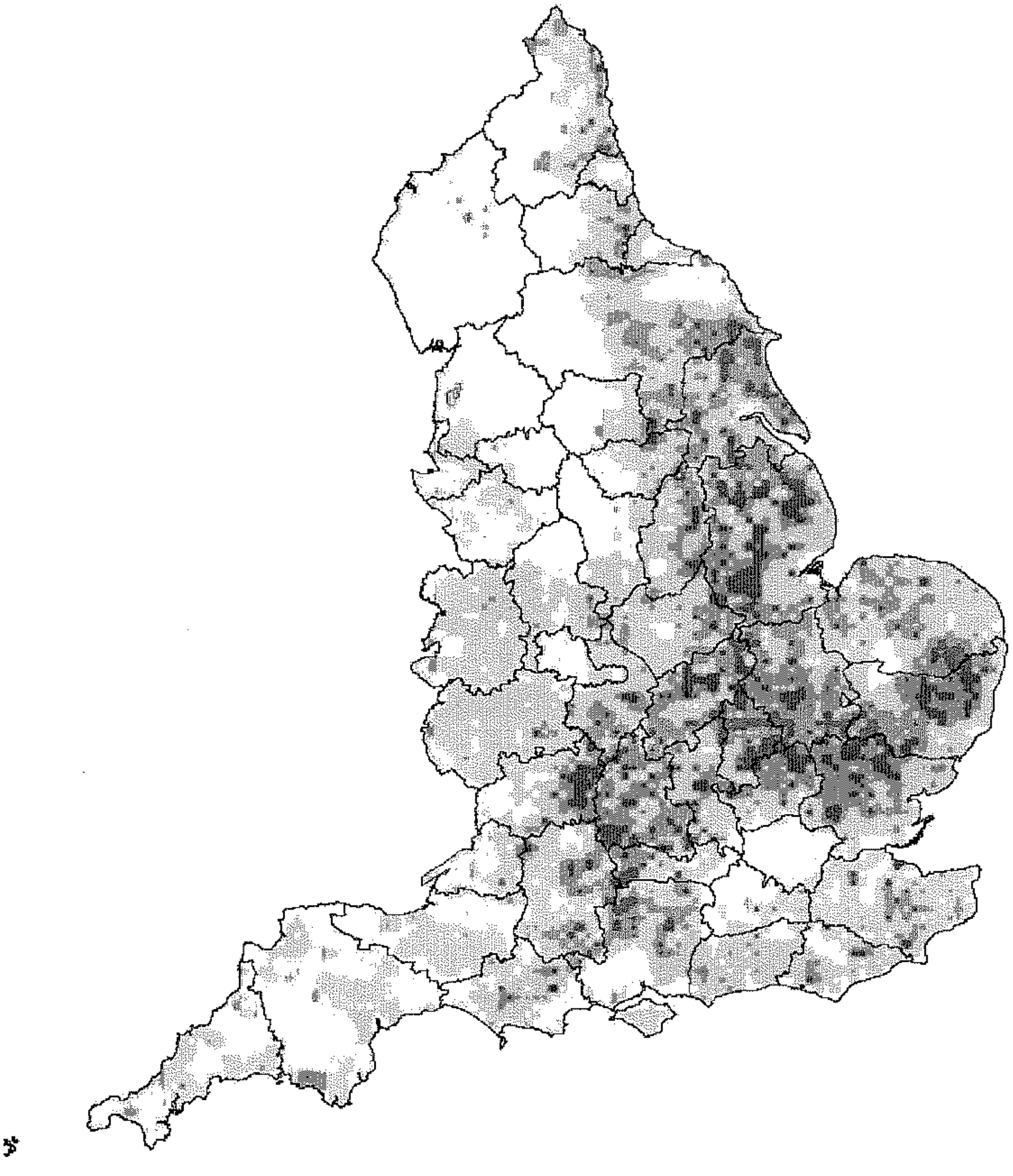

Figure 2 The distribution of set-aside in 2000 in England. The pale grey areas (e.g. Somerset) represent 100-150 ha per 5 km square; the darkest areas represent 300-694 ha per 5 km square.

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CHAPTER 6 CONCLUSIONS: SUMMARY OF CURRENT KNOWLEDGE AND KEY RESEARCH PRIORITIES TO MEET THE PSA TARGET FOR FARMLAND BIRDS

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6.1 Introduction

A key element of this project is to 'use information derived from the preceding chapters to recommend ecological or demographic research required to identify, strengthen and quantify management recommendations for birds'.

In this chapter we first summarise current knowledge required to meet the PSA target (section 6.2). We then describe a number of important research requirements in five sections. First we consider what would be required to improve both the demographic and bird-habitat models (sections 6.3, 6.4 and 6.5), second we consider what research would be required to gain a better understanding of either individual species or farming systems (section 6.6). These sections simply describe research required, they do not identify whether there are any relevant, ongoing research projects or attach priorities to this research. In the final section (6.7) the potential benefit of these research needs, with respect to meeting the PSA target, are listed in Table 3. In this table we have also attempted to identify the research we feel is high priority, based on the extent to which they would assist in meeting the PSA target. It is important to note that this research identified is explicitly in relation to meeting the PSA target and not farmland bird research in general.

6.2 Summary of Current Knowledge

Before we identify future research requirements, it is important to summarise the salient points from the preceding chapters. For many species on the index, quantified predictions of the extent of population recovery following specified management are clearly not possible with existing data. Possible research to address this issue is discussed later in this chapter. Nonetheless, from the data presented in the preceding chapters, we can construct a framework for meeting the PSA target in the most effective and efficient manner. First, for all the declining species in the index, we identify when in the year to act and what management options to take, to give the most effective predicted population response (Table 1). Thus, for example, we identify from Chapter 1 that Tree Sparrows are currently most limited by over-winter survival, such that we should target over-winter food supplies in order to have the greatest initial impact on this species. However, as discussed in the table legend, this may not itself lead to sustained population recovery – it is merely a diagnosis of how to have the greatest initial impact. At some unknown point, the most limiting factor will cease to be limiting, at which point the population will be limited by another demographic rate (in the case of Tree Sparrow, a switch from limitation by over-winter survival to limitation by productivity or summer adult survival). A strategy for sustained population recovery would therefore have to be much more inclusive of all potential limiting factors. In the case of the severely-declining Cirl Bunting, for example, research has shown that populations responded to provision of both summer feeding (rough grass for large invertebrates such as Orthoptera) and winter feeding (weedy stubble) habitat, but populations increased most when both summer and winter habitat were provided in close proximity to each (Peach *et al.* 2001). In addition, the Grey Partridge model shows clearly how one limiting factor gives way to another as the corresponding bottleneck is released (Potts 1980, 1986).

The data in Table 1 are presented on a species-by-species basis and give some idea of the relative predicted strength of response of each species to a number of management options. Alternatively, the table can be structured by treatment. In Table 2, we show the predicted efficiency (in terms of number of species benefiting) of management options. Only the top seven, broad options are listed. While we identify these as key options for delivery for a range of species, there are also a few options that, though more limited in the range of species which they benefit, are nevertheless extremely good for the targeted species. In this category, we would include unsown patches within cereal fields, to provide nesting habitat for lapwings. Among the broad options in Table 2, we again stress the need to explicitly consider the *quality* of each option in order for it to meet its goal. For example the importance of weedy versus clean stubbles or the importance of appropriate management of set-aside land. Assuming suitable quality, we then estimate the quantity, cost and targeting needed to optimise the chance of delivery. These tables provide an extremely valuable and important basis for the design of future policies which seek to deliver the PSA target.

6.3 Future Research Requirements

6.3.1 Improving demographic models for farmland birds

For a number of the farmland bird index species, data derived from long term national surveys and intensive autecological studies are sufficient both to identify the demographic parameter responsible for population declines and provide an indication of the extent of changes in these rates that would be required to reverse them (see summary Table 12, Chapter 1). However, for several species this remains unknown or at least uncertain, often because of a lack of data on survival (particularly post-fledging and seasonal patterns), number of breeding attempts and recruitment. Demographic monitoring of farmland birds has already been identified as a high priority within the BTO ringing scheme (Baillie *et al.* 1999). Survival data are difficult to collect at a large scale and for many species. However, it may be possible for a small number of targeted farmland species, e.g. Corn Bunting and Tree Sparrow, from a network of say 10 sites with intensive work using colour-marking and resighting. For many of these parameters the first priority is to develop new field and analytical methods. In addition, with the exception of Grey Partridge, the role of density dependence in farmland bird populations is poorly understood, particularly with respect to its role within different life-cycle stages. Current approaches investigate density-dependence of changes in abundance, but density dependence of demographic factors such as breeding performance and survival need to be investigated by a combination of intensive and extensive approaches for at least one passerine farmland species. One key aspect of population processes that has not been addressed in studies of farmland birds is that of trade-offs between survival and breeding success (life-history trade-offs). Thus, birds breeding in a poor environment may maintain their productivity at the cost of their own future survival or vice versa. These trade-offs have important consequences for measuring and modeling demographic parameters and are considered in more detail in section 6.6.

Finally, spatial scale effects may be important because dispersal and movement between metapopulations may affect the response of national population changes to the implementation of local management actions. The influence of habitat management on individual dispersal distances and natal philopatry needs investigating for Grey Partridge and at least one passerine farmland species.[dispersal is a much-neglected area, and even for Grey Partridge we do not know enough about the whys and wherefores. I consider this a particularly important area that needs investigation].

Finally, we cannot assume that present cropping patterns will remain constant. A changing economic environment, including the impacts of CAP reform, EU enlargement and changes to WTO agreements, could substantially change the proportions of different crops grown in the UK. Making demographic predictions of the response of farmland bird populations in such altered circumstances would be dangerous, as it would require extrapolation beyond the realm of the data on which the models were created (Bradbury *et al.* 2001). In addition, the impact of climate change is likely to accelerate rapidly over the next 20 years, when management plans to improve the status of birds forming part of the PSA target will be implemented. It is important to understand how climate change will enhance or inhibit the predicted affects of changes in agricultural management. A straightforward first step would be to investigate how demographic factors of birds listed on the PSA target are affected by climate variability, using historical databases. Thus the demographic models can include the potential benefits or disbenefits of climate changes scenarios to improve the accuracy of their predictions of how the populations will change. It is possible that non-linearities in the response of demographic factors to climate change will alter the variable that is most susceptible to changes in land management.

Thus, improving existing demographic models, particularly integrated population models (IPMs) generated from BTO and GCT data-sets requires the following key research elements:

Collection of more extensive data on survival and dispersal
- **Development of techniques to measure seasonal patterns of survival**
- **Collection of more extensive data on number of breeding attempts**
- **Collection of more extensive data on recruitment**
- **Measuring and incorporating density dependence of demographic rates**
- **Consideration of spatial scale effects**
- **Consideration of the impact of climate change on demographic models**
- **Consideration of the impact of policy reforms on demographic models**

Addressing these knowledge gaps will; (i) identify key demographic rates, where this is unknown and so ensure cost-effective targeting of first phase management action; (ii) improve the models and hence the accuracy with which we can predict effects of changes in demographic rates (as a result of habitat management) on population trends.

6.3.2 Improving bird-habitat models

Our qualitative understanding of the relationship between farmland bird species and different habitat types is good. However, quantitative data exist only for a small number of species (Lapwing, Woodpigeon, Skylark, Yellow Wagtail, Whitethroat, Rook and Yellowhammer, chapter 3). Of these, only Lapwing, Skylark, Yellow Wagtail and Yellowhammer are declining. Even then, the quality of data differs widely between species. Key research to address this issue falls into three broad categories (i) improving models that already exist, (ii) developing models for the remaining species in the farmland bird index, (iii) considering other modeling approaches (i.e. other than 'deterministic stage-structured population modeling' and habitat selection models used in this contract, Bradbury *et al.* 2001).

The bird-habitat models presented in Chapter 3 could be improved in a number of important ways. First, their robustness could be tested by validating them in different landscape types to assess the extent to which they apply across farmland in Britain. Habitat selection is likely to be landscape-specific, for example, and the importance of particular crops is likely to differ according to temporal and spatial availability of alternative habitats. In addition there may also be density dependent habitat selection in which key habitats differ under conditions of high and low population densities. Applying models to other landscape types and differing bird densities could also be used to examine these landscape-specific and density dependent effects.

Second, the models focus only on breeding season habitat whereas habitat in late winter e.g. stubble availability, is also likely to be important in determining breeding density. Third, spatial scale effects may be important, for example, the fine spatial scale used for modeling Yellowhammer distribution is suitable for this territorial species but not for Linnet, which ranges over large distances. Models at larger spatial scales may be more appropriate for Linnet and similar species. In addition, monitoring tends to be at the 1 km square level (through BBS) - where models (and action) are based at smaller scales, reconciling the scale of monitoring and action (agri-environment measures) is also important.

Noting that such bird-habitat models have never been done for Grey Partridge, it would be very interesting to do for Grey Partridge what has been done for Skylark and Yellowhammer; from such a base, it should be relatively easy to graft on density dependence and dispersal. The only species for which data exist already, but have not been analysed in such a strict bird-habitat context, is the Grey Partridge. For this species, information on spatial distribution, breeding success and land use exist for a 62-km^2 area of Sussex over 33 consecutive years. Data are required to construct such models for other farmland bird index species, particularly those currently in decline or at historically very low levels. This could be achieved through further intensive studies of individual species. In addition, models could be constructed by careful assessment of the response of bird numbers to changes in the landscape as a result of agri-environment schemes, for example by revisiting ASPS plots or careful survey and sampling of the new arable options within CSS.

Deterministic stage-structured population modeling and habitat selection models have been used in this contract. A variety of other modeling approaches, such as rule-based models, could be considered (Bradbury *et al.* 2001) and it would be useful to determine the utility of other bird-habitat modeling methods.

Finally, changes in technology (e.g. GMHT crops), large-scale policy reforms and climate may have considerable impacts on the spatial distribution and nature of agricultural practice over the next 20-50 years. The impact of the widescale introduction of GMHT crops on farmland birds remains unknown but it could be profound (Watkinson *et al.* 2000). With respect to climate change, it has been predicted, for example, that the distribution of wheat could move out of East Anglia, to be replaced by novel crops currently little grown in the UK (e.g. drought-resistant crops). Biomass crops such as short-rotation coppice and miscanthus may also become more widespread. It will be important to develop bird-habitat models for crops that are likely to be introduced with climate change or policy reform, so that the potential impacts of these changes on species in the farmland bird index can be predicted.

Thus key research needs to be addressed to improve bird-habitat modeling are

To improve existing models

- **Test model robustness - validation in new landscapes to assess geographic generality**
- **Incorporate habitat variables from one season when predicting distribution in another season (e.g. stubble to predict summer distribution of certain granivorous passerine species)**
- **Considering spatial scale effects - models at larger spatial scales may be more appropriate for non territorial and highly mobile species**
- **Develop bird-habitat models for crops that are likely to become widespread as a result of climate change over the next 20 years**
- **Develop models that may be used to predict the impact of the introduction of new technologies or policy reform on birds e.g. GMHT crops.**

To create models for other species

- **Intensive bird-habitat studies for some species e.g. Grey Partridge and Yellow Wagtail**
- **Revisiting ASPS sites to monitor population density with respect to availability of specific options**
- **Using a 'supplement' to the BBS, with improved habitat monitoring to assess the relationships between bird numbers and availability of options within CSS (* see section 6.6.2)**
- **Investigating *post hoc* whether it is possible to estimate how much (rather than just presence) of specific habitats was required for Cirl Bunting recovery? Though Cirl Bunting is not in the PSA target, this exercise would nevertheless be very useful in elucidation the scale of change that was required to reverse the decline of a farmland bird.**

Determine the utility of other bird-habitat modeling methods

- **Applicability and development of individuals-based behavioural population models e.g. for winter granivorous passerines**
- **Applicability and development of other bird-habitat modeling methods e.g. rule-based, genetic algorithms**

Bird habitat models can provide an extremely useful tool for predicting the effect on populations of future habitat manipulations and assessing the relative cost-effectiveness of a range of management options. The research priorities listed would increase the value of bird-habitat models by increasing their accuracy and generality and by increasing the number of species for which they could be constructed. Some of these research activities (e.g. revisiting ASPS sites) would also provide an assessment of whether these agri-environment schemes have delivered their 'bird objective'

6.3.3 Linking IPMs and bird-habitat models for farmland bird species

The most valuable models, in terms of relating management change to population are those that link resource availability to demographic parameters and hence to population size. To date this has only been possible for the Grey Partridge. Construction of "grey partridge-type" models could be attempted for species where data exist on habitat specific breeding success e.g. Skylark. These could link changes in habitat to changes in productivity, which would inform demographic models. These models are complex, and developing such integrated models for many species in the farmland bird indicator would be an ambitious undertaking. However, in the long term, generalised predictions are not going to be possible without this level of understanding for at least a few carefully selected species of passerine. An extension of this would be to develop novel approaches which link resource, demography, density and abundance relationships, for example by using individual covariate models (i.e. food, condition, survival relationships) of individual based population models (Bradbury *et al.* 2002). Thus linking demography and habitat change requires:

- **Developing models for case studies e.g. Skylark and Linnet**
- **Development of new approaches e.g. individual covariate models**

6.3.4 Improving our understanding of species resource requirements

This field of research encompasses several broad questions. First, *what* nesting and foraging resources do individual farmland bird index species require and *when*? Second, *how much* of these resources do they require and *where*? For most species, we are able to answer the first question with one important caveat – we are far less able to answer it for birds within grassland than arable. For this reason we consider resource requirements in grassland systems as a specific issue.

6.3.5 What resources do individual species require and when?

Our understanding of what resources (nesting and foraging requirements) individual species require and when is relatively good for most of the index species, with one notable exception - Yellow Wagtail. Our knowledge of resource requirements of some of the increasing or stable species is also poor but within the framework we have adopted in Chapter 1 we do not consider research into this to be a priority. However, knowledge is poor with respect to one general aspect of resource requirements, namely the interaction between abundance and availability. For example, sward structure in grassland or stubbles will strongly influence the extent to which weed seed and grain and invertebrates present are actually available to the birds (Vickery *et al.* 2001, Whittingham & Markland 2002). There is thus potential for simple sward management practices such as creating exposed soil surfaces (Perkins *et al.* 2000) to significantly increase food availability for some birds.

One key aspect of population processes that has not been addressed through intensive autecological studies of farmland birds is that of life-history trade-offs. A number of theoretical studies have demonstrated trade-offs between survival and breeding success in birds. Thus, birds breeding in a poor environment may maintain their productivity at the cost of their own future survival. For example, studies of Red-Backed Shrikes *Lanius collurio* in Switzerland showed pairs breeding in poor habitats invested more time in flight activity than those in good habitats, and consequently lost more body weight, possibly compromising their own survival prospects (Leugger-Eggimann 1997). Similarly, the costs of surviving in a poor environment may be manifested in reduced breeding performance. For example, pheasants given supplementary food in spring (before breeding) produce twice as many chicks as otherwise, because they are able to retain good body condition and have repeat breeding attempts (Draycott 2002). There are suggestions that similar processes may be affecting at least Turtle Dove and Com Bunting, caused by a lack of summer and late winter food respectively. Such trade-offs may result in deterioration in habitat quality in one season being correlated with change in a demographic rate in another season. Assessing the extent to which such trade-offs exist has important implications with respect to when to provide resources. The research required to tease these processes apart is complex and time consuming. A small number of case studies including, for example, one species for which breeding success seems to be limiting e.g. Linnet, and one for which survival seems to be limiting e.g. Yellowhammer, may be very valuable in this context. A related issue is also the extent to which provision of resources under current AES prescriptions provide sufficient food in the late winter/early spring or does this represent a bottleneck in food abundance/availability?

For most species, we have much poorer knowledge in grassland than arable systems of (i) species resource requirements (ii) impact of changes in management. A number of research activities are required to address these issues. First, most intensive studies have focused on birds within arable or mixed systems. Repeating a large number of these studies in grassland would not be cost-effective. However, a small number of case studies, e.g. for a field nesting species such as Skylark and a hedgerow nesting species such as Yellowhammer, within grassland would provide very valuable information on the resources these birds require and when.

Second, a more detailed understanding of the way in which the grassland bird community is affected by grassland management practices is also required. This could be achieved by broad scale, year round, bird-habitat surveys conducted on grassland under different management intensities. Third, our understanding of how birds use key habitats, such as hay meadows and extensive grassland, within grassland landscapes and how management alters this is relatively poor. For example, what is the exact effect of the timing and intensity of grazing? Re-seeding or reduced fertilizer inputs? Indeed, do we know the best way to extensify grassland management? Fourth, knowledge to date suggests that the uniformity of intensively managed grassland habitats, and the loss of arable pockets within predominantly grassland areas, has reduced their suitability of these landscapes for many birds as nesting and foraging habitat (Robinson *et al.* 2001, Vickery *et al.* 2001). Increasing spatial heterogeneity, at a range of scales, should be a goal in the management of grassland for farmland birds either through (i) extensification of grassland itself or (ii) active management of margin or within field plots either as species rich grass swards or low intensity arable land. The effectiveness of potential management prescriptions could be assessed through large-scale, replicated trials with carefully selected controls in which the response of bird populations could be assessed through measurement of numbers, productivity and/or use. Intensive studies of individual species or groups of species could also be conducted as part of these experiments, collecting detailed information about how birds utilise newly created resources.

Finally, although we have concentrated on the need to understand better how birds respond to changes in the management of grassland habitats, the application of "grass pockets" in arable areas may be of benefit to species such as Starling, and similar work on these species would be potentially valuable.

Thus improving our knowledge of what to provide and when these resources are needed requires:

Individual species resource requirements

- **Autecological study of Yellow Wagtail in arable and grassland**
- **Intensive 'case studies' of life-history trade-offs**

Resources required in grassland landscapes

- **Intensive 'case-studies' of species in grassland landscapes**
- **Intensive 'case-studies' of key habitats in grassland landscapes**

- **Broad scale studies of the bird community associated with grassland managed under different intensities**

Assessing the effectiveness of different management options in grassland systems

- **Large scale experiments to assess bird numbers and use, and food resources on manipulated margin plots to assess the value of (i) reduced grazing, cutting and inorganic nitrogen inputs on grassland (i.e. mimicking extensification) and (ii) low intensity arable land e.g. as whole-crop silage, fodder crops or winterbird cover crops**
- **Large scale experiments to assess bird numbers and food resources of manipulated fields or in-field plots to assess the value of (i) reduced grazing, cutting and inorganic nitrogen on grassland (i.e. mimicking extensification) and (ii) low intensity arable land e.g. as whole-crop silage, fodder crops or winterbird cover crops**
- **Large scale experiments (as above) to assess the optimal spatial distribution of low-intensity arable/grass pockets**

Assessing the effectiveness of different management options in arable systems

- **Large scale experiments (as above) to assess the optimal spatial distribution of arable/low-intensity grass pockets**
- **Further work on how to provide late winter/early spring seed resources**

6.3.6 How much resource do individual species require and where?

To date most research on farmland birds has been relatively qualitative. A major research issue that needs to be addressed if this qualitative research is to be translated into cost-effective action is how much of the key resources/habitats is required? Our attempts to do this in this contract are described in preceding chapters. There are two further ways in which this might be achieved; through novel modeling approaches and large scale experiments.

Deterministic stage-structured population modeling and habitat selection models have been used in this contract. Among other existing modeling approaches, individuals-based population models, if applied to winter granivorous passerines, may be able to answer the question 'how much stubble is enough' by relating individual food requirements, seed intake rates and seed availability (Sutherland 1996).

A large scale experiment to address the issue 'how much is enough' is likely to be very complex to establish. However, the new arable options within CSS could provide a natural experiment that may yield valuable information in relation to this issue. It is possible that a 'supplement' to the existing BBS (the monitoring scheme that will be used to generate the farmland bird index in the future) could be designed that would enable the effectiveness of these new options to be assessed and provide some information concerning the required scale of habitat creation. The BBS plots not affected by CSS options would provide a control against which changes could be measured. The feasibility of using the BBS to monitor changes in populations of common breeding birds in ESAs in England and Wales suggest a large number of common species occur at enough sites to assess population trends. These trends could be reliably compared to population trends on random squares outside ESAs (Noble & Newson 2002). It would be valuable to repeat this exercise for CSS. Given the more fragmented nature of CSS agreement land, compared with ESA agreement land, this research would provide valuable information upon which to assess the need and design of suitable 'supplement' to the BBS.

Rather little is known about three other aspects of resource use (i) how new resources should be spatially distributed and how this differs between species, (ii) how long they need to be in place to have an effect and; (iii) the interaction between quantity and quality. Considering first the question of spatial distribution of resources. For highly territorial species, such as Yellowhammer, which exclude each other from territories, it will probably be necessary to have numerous patches of say, food resources, spread ubiquitously across the landscape. For less territorial species, such as Linnet and Turtle Dove, it may be possible to provide such food resources as occasional high quality patches of habitat, which many individual birds can travel to for communal use. There are two ways to address this issue; intensive studies of the movement of birds using colour-marking and radio telemetry and or large scale experiments investigating the use of resources created experimentally in a spatially explicit way.

The interaction between quantity and quality of resources will be important and we know very little about this. For example, what are the relative benefits of creating small high quality food patches as opposed to large low quality ones? On another scale, are nature reserves actively managed for wildlife within intensive farmland better than broad integration of less intensively managed habitat into the farmland landscape with holistic wildlife, water quality and flood defense goals (Sutherland 2002). Similarly, the effects over time are poorly understood. For example, how long does a habitat need to be in place to have an effect on bird populations and how does this differ between species and habitats? These questions have important implications for the development of options within agri-environment schemes and wider farm management practices.

Thus improving our knowledge of how much resource individual species need, and where, requires:

- **The application of individuals-based population models to farmland ecosystem**
- **The occurrence and abundance of birds on BBS squares within existing CSS**
- **Use of a supplement to BBS to monitor effectiveness of new CSS**
- **Intensive studies of movements of birds in summer and winter in relation to spatial distribution of resources**
- **Large scale experiments to investigate optimal spatial distribution of resources**

6.3.7 Understanding the side-effects of providing extra resource

The research requirements outlined so far make the tacit assumption that habitat measures targeted at a particular farmland species or group of species will act as though the species lived in splendid isolation. In fact ecology tells us that there are complex links between species that can lead to undesirable side-effects. When recommending the provision of extra resource, we need to be certain that the side-effects of doing so do not negate the direct benefits that have been described and quantified in earlier sections. There are two particular cases that stand out.

The first is the bird-table effect, whereby the provision of extra food resources in small areas leads to concentrations of birds than in turn attract the attention of predators. There have already been anecdotal reports of Sparrowhawks attacking farmland songbirds that have been attracted to Wild Bird Cover crops that offer them winter food. The question is whether the enhanced survival wrought by the provision of food outweighs the reduction in survival occasioned by the increase in predation. More constructively, what are the best type of crops to grow, in what shape and size of plot, and in what position relative to alternative cover, in order to minimise predation risk? This is not a trivial issue.

The second is the corridor effect, whereby the provision of nesting habitat in strips along field boundaries can act as hunting corridors for mammalian predators such as foxes, badgers or mustelids, with negative effects on the breeding success of ground-nesting birds. This has been shown to be the case for Conservation Reserve land in North America. Again, questions revolve around how the nature of the vegetation, its management and strip width influences the corridor effect and hence the magnitude of the unwanted side-effects that counteract the beneficial impact of resource provision.

These questions are not trivial, but nor are they necesarily difficult to resolve. Recommendations arising from research into these areas would feed directly into management prescriptions, and ensure maximum benefit from their deployment.

Thus understanding such side-effects requires

- **Study of optimum configuration of winter food provisioning so as to maximise their effectiveness in the presence of negative side effects**
- **Study of optimum configuration of nest-habitat provisioning so as to maximise productivity in the presence of negative side effects**

6.4 Summary

The research activities outlined above are listed in Table 3. Some of these are already the subject of ongoing research and, where this is the case, details are provided. We have attempted to prioritise this research based on its potential value with respect to meeting the PSA target. Within the suite of research activities outlined we suggest four key priorities:

(i) A feasibility study to assess the extent to which a 'supplement' to the BBS could be used to (a) assess the effectiveness of the CSS particularly the new arable options being rolled out nationwide and future AE developments in delivering their 'bird objectives' and (b) assess whether such a 'supplement' could also provide data on the extent of habitat/resource required to reverse population declines. Such a supplement would require the selection of additional BBS squares selected to cover land managed under a range of agri-environment scheme options and controls (these would not be included in the BBS population index to avoid bias). A feasibility study would include power analyses to determine how many squares would be required to detect a given level of difference in bird numbers.

(ii) Revisiting ASPS plots and carrying out detailed surveys of birds and their habitat in winter and summer would provide important information with respect to improving bird habitat models, assessing whether options within this agri-environment scheme have delivered their 'bird objective, and some data on the extent of habitat/resource required to enhance local bird populations.

(iii) A large scale experiment designed to assess the cost-effectiveness of the introduction of arable pockets into grassland landscapes in the form of whole field, in-field or margin plots. This could address (i) relative benefits of arable pockets in a grass only and mixed landscape (ii) cost-effectiveness of different crop types (e.g. arable crops and stubbles versus wildbird cover crops) and/or plot sizes; (iii) optimal spatial arrangement of these resources and (iv) patterns of utilisation by different species in summer and winter (to identify mechanisms underlying any differences and to determine the optimum dispersion of food patches).

(iv) Either, (i) to attempt to develop models that link resource availability (i.e. management change) to demographic parameters and hence to population size for one to three key case species e.g. Skylark, or (ii) to further develop the potential of individuals-based models to generate accurate quantified predictions of the amount of resource needed for wintering granivorous passerine populations to survive over-winter.

Table 1 Timing and nature of action needed to deliver most effective population responses for key declining species

Species[1]	Limiting demographic factor[2]	Habitat management options[3]	Predictions[4]	Secondary limiting demographic factor[5]	Habitat management options[3]	Predictions[4]
Tree Sparrow	Winter survival	**Stubble**^{A} **Set-aside stubbles**^{A} **Supplementary feeding** **Wild bird cover** **Arable Pockets**^{P} Spray control^{A}	↑↑ ↑↑ ↑↑ ↑↑ ↑↑ ↑	Productivity	**Hedgerow trees** **Nestboxes** **Aquatic habitats** **Set-aside fallows**^{A} Spray control^{A} Hedges Field margins	↑↑ ↑↑ ↑ ↑ ↑ ↑ ↑
Corn Bunting	Winter survival[6]	**Stubble**^{A} **Set-aside stubbles**^{A} **Supplementary feeding** **Arable Pockets**^{P} **Wild bird cover** Spray control^{A} Grassland extensification^{P}	↑↑ ↑↑ ↑↑ ↑↑ ↑↑ ↑ ↑	Productivity[6]	**Spray control**^{A} **Field margins** **Spring cereals (undersown)**^{A} **Grassland extensification**^{P} Arable pockets^{P} Pastoral pockets^{A}	↑↑ ↑↑ ↑↑ ↑ ↑ ↑
Grey Partridge	Productivity	**Spray control**^{A} **Field Margins**^{A} **Spring cereals (undersown)** Set-aside fallows^{A} Wild bird cover Hedges	13.8%[7] ↑↑ ↑ ↑ ↑↑ 1.19%	Winter survival	**Wild bird cover** **Arable Pockets**^{P} Stubble^{A} Set-aside stubbles^{A} Spray control^{A} Supplementary food	↑↑ ↑↑ ↑↑ ↑↑ ↑↑ ↑
Turtle Dove	Productivity	**Set-aside fallows**^{A} **Spray control**^{A} **Scrub creation** Hedges Field margins Wild bird cover	↑↑ ↑↑ ↑↑ ↑ ↑ ↑↑			
Starling	Productivity	**Sward height control**^{P} **Pastoral pockets**^{A} **Hedgerow trees** Minimum tillage^{A}	↑↑ ↑↑ ↑ ↑	Winter survival	**Sward height control**^{P} **Pastoral pockets**^{A} **Stubble**^{A} Minimum tillage^{A}	↑↑ ↑↑ ↑ ↑
Reed Bunting	Winter survival	**Stubble**^{A} **Set-aside stubbles**^{A}	↑↑ ↑↑	Productivity	**Aquatic habitats** **Set-aside fallows**^{A}	↑↑ ↑

Species^{1}	Limiting demographic factor^{2}	Habitat management options^{3}	Predictions^{4}	Secondary limiting demographic factor^{5}	Habitat management options^{3}	Predictions^{4}
		Supplementary feeding **Wild bird cover** **Arable Pockets**^{P} Spray control^{A}	↑↑ ↑↑ ↑↑ ↑		Field margins Oil-seed rape^{A} Spray control^{A}	↑ ↑ ↑
Yellow-hammer	Winter survival	**Stubble**^{A} **Set-aside stubbles**^{A} **Supplementary feeding** **Wild bird cover** **Arable pockets**^{P} Spray control^{A} Grassland extensification^{P}	↑↑ ↑↑ ↑↑ ↑↑ ↑ ↑ ↑	Productivity	**Hedges** **Field margins: 6 m** **2 m** **Arable pockets**^{P} **Ditches** Aquatic habitats Scrub	0.52% 0.71% 0.21% 0.23% 0.13% ↑ ↑
Skylark	Productivity	**Arable pockets**^{P} **Set-aside fallows**^{A} **Organic farming** **Spring sowing**^{A} Beetle banks Spray control^{A}	1.48% 0.58% 0.42% 0.26% ↑ ↑	Winter survival	**Stubble**^{A} **Set-aside stubbles**^{A} **Supplementary feeding** **Wild bird cover** **Arable pockets**^{P} Spray control^{A}	↑↑ ↑↑ ↑↑ ↑↑ ↑↑ ↑
Lapwing	Productivity	**Arable pockets**^{P} **Pastoral pockets**^{A} **Set-aside fallows**^{A} **Sward height- control**^{P} **Spring sowing**^{A} Spray control^{A} Spring cereals (undersown)	↑↑ ↑↑ 0.58% ↑ 0.05% ↑ ↑	Winter survival	Reduced ploughing^{A} Pastoral pockets^{A} Sward height- control^{P}	↑ ↑ ↑
Kestrel	Productivity	**Hedgerow trees** **Field margins** Set-aside fallows^{A} Grassland extensification^{P} Nestboxes	↑↑ ↑↑ ↑↑ ↑↑ ↑	Winter survival	**Field margins** Grassland extensification^{P} Stubbles^{A}	↑↑ ↑ ↑

Table 1 Continued.

Species[1]	Limiting demographic factor[2]	Habitat management options[3]	Predictions[4]	Secondary limiting demographic factor[5]	Habitat management options[3]	Predictions[4]
Barn Owl	Winter survival	**Field margins** Grassland extensification[P] Pastoral pockets[A]	↑ ↑ ↑	Productivity	**Nestboxes** **Hedgerow trees** **Field margins** Set-aside fallows[A] Grassland extensification[P]	↑↑ ↑↑ ↑ ↑ ↑
Yellow Wagtail	Productivity	Aquatic habitats Arable pockets[P] **Broad-leaved crops[A]** Grassland extensification[P] **Spring sowing[A]** Spray control[A]	↑↑ 1.18% 0.91% ↑↑ 0.07% ↑			
Whitethroat	Productivity[8]	**Field margins: 6 m** **2 m** **Hedges** **Arable pockets[P]** **Scrub** Spray control[A]	4.73% 2.31% ↑↑ ↑↑ ↑↑ ↑			
Linnet	Productivity	**Field margins** **Set-aside fallows[A]** **Oil-seed rape[A]** Arable pockets[P] Pastoral pockets[A] Spray control[A] **Scrub** **Hedges** Grassland extensification[P]	↑↑ ↑↑ ↑↑ ↑↑ ↑↑ ↑↑ ↑ ↑ ↑	Winter survival	**Stubble[A]** **Set-aside stubbles[A]** **Supplementary feeding** **Wild bird cover** Spray control[A]	↑↑ ↑↑ ↑↑ ↑↑ ↑

Table 1 Continued.

[1] The table includes data for all nine species listed in Chapter 2 as being currently declining and thus the most important targets for land management action. These species have been ranked in descending order of rate of decline. Those species below the dashed line did not show a negative population growth 1990-2000. However, they have been included as being worthy of land management action (and hence possible contributors to an improved trend in the farmland bird index) due to inclusion on the amber list of Birds of Conservation Concern (Gibbons *et al.* 1996).

[2] Demographic rate considered to be currently limiting population size (see Chapter 1, Table 4). These have been generalised into productivity (number of breeding attempts or fledglings per attempt) and non-breeding (over-winter survival) effects.

[3] Habitat management options considered likely to affect the limiting demographic factor. Those in bold are those for which we have high confidence of an effect, while those which are not in bold are those where an effect is possible but where we cannot yet be certain (see also chapter 5). While many of these are currently available as options in agri-environment schemes, not all are. These options have been approximately ranked in importance. In all cases it is presumed that options are produced to tight prescriptions giving the optimum benefit to birds e.g. stubbles are weed-rich. The options given are often generic terms covering a range of different prescriptions, which will vary by farming landscape, region and target. Options appropriate only to pastoral land (including grassland in mixed farmland) are superscripted ^{P}. Conversely options applicable to arable land (including arable land within mixed farms) are superscripted ^{A}.

Explanatory notes for management options:
Aquatic habitats – wet ditches, farm ponds and other wet areas.
Spray controls – conservation headlands and management option OS2 (also includes restrictions on drilling dates and density).
Arable pockets – areas of arable land within regions previously entirely pastoral, created by the conversion of land management, e.g. silage fields to fodder crops.
Oil-seed rape – where this crop is recommended as being beneficial it refers to crops sprayed off to desiccate rather than cut (which destroys second brood nests, Burton *et al.* 1996)
Field margins – also includes beetle banks.
Spring cereals – includes undersown spring cereals
Stubble – includes undersown stubble

[4] The predicted effect of increasing the named resource. In most cases this is indicated with symbols estimating the scale of increase, with ↑ = slight increase and ↑↑ = significant increase. For a number of species, quantitative estimates are available from the results of modelling presented in Chapter 3. These are presented as the percentage increase in population size resulting from a 1% increase in the named option. All values given are for the estimated increase when options are applied to a mixed farmland landscapes; predictions for pastoral and arable landscapes can be found in Chapter 3.

[5] The demographic factor considered to be the likeliest to limit population growth if the currently limiting factor was increased to a level at which it ceased to be limiting. Shading indicates the migratory species for which limiting factors in the non-breeding season are obviously beyond the influence of UK agricultural policies.

[6] It is not known whether productivity or winter survival is currently limiting Corn Bunting population size in Britain.

[7] An increase in suitable habitat for Grey Partridge is an increase from 2.35% to 3.35% of the arable area.

[8] Survival on the sub-Saharan wintering grounds is likely to be the demographic rate that limits the population size of Whitethroat in Britain.

			Comments			
Treatment	**Number of species benefiting[1]**	**Main benefits**	**Quality – management recommendations to ensure delivery**	**Cost[2]**	**On-farm area needed**	**Targetting Required[3]**
Arable pockets & pastoral pockets[A/P]	9 + 3	Provides habitat heterogeneity, vital for some species e.g. Lapwing, Starling. Can provide habitat for pastoral species in arable regions and vice versa.	• Whole crop silage followed by winter stubbles in pastoral regions • Low-intensity grazed pasture in arable regions	High High	Small Small	F & L L
Non-cropped habitats (hedges, hedgeline trees, aquatic habitats)[A/P]	9 + 2	Breeding sites for many species in trees, scrub and hedges. Year-round food resource. Aquatic habitats (ponds, wet ditches) important feeding habitats for species such as Tree Sparrow & Reed Bunting	• Hedge cutting frequency reduced • Hedge/trees planted • Pond management/creation • Wet ditch maintenance	Low High High Low	Large Small Small Large	None F & L F None
Set-aside[A]	8 + 3	Provides nesting sites for field-nesting species. Weed seeds and invertebrates during the breeding season, and seeds for wintering granivores.	• OS3 management • Decrease pesticide in preceding crop • Decreased and later cutting	High Low Low	Small Large Large	F & L None None
Non-cropped field margins[A/P]	7 + 4	Provides food-rich strips (invertebrates, grass & weed seeds) adjacent to nest sites. Also a source of winter food, including mammal prey for raptors and owls.	• Beetle banks • Margins (2 or 6 m), sown or regenerated, cut/uncut matrix	Low Low	Small Large	None None

Table 2 Ranking of management options by number of key declining species to benefit.

			Comments			
Treatment	**Number of species benefiting[1]**	**Main benefits**	**Quality – management recommendations to ensure delivery**	**Cost[2]**	**On-farm area needed**	**Targetting Required[3]**
Winter stubbles[A] [these two options need to be split]	7 + 2	Correctly managed winter stubbles provide invaluable food resources for granivorous birds.	• Reduced herbicides in preceding crop for weedy stubbles • Retain stubbles into spring • OS1 management	High Low Low	Small Small Small	None None None
Wild bird cover[A/(P)]	7 + 1	Provides important winter food resource for granivores, also source of both invertebrate and seed food in the summer. Provides nesting cover.	• Small selection of crop types in adjacent strips (kale, quinoa, cereals, linseed)	High	Small	None
Spring cereal[A]	5 + 0	Spring cereals provide a suitable sward structure for ground-nesting birds such as Skylark and Lapwing If undersown, provides weedy stubbles through to following summer ley.	• Undersowing reduces herbicide inputs for weedy crop (more chick-food insects) and stubbles • Undersowing retains stubbles into following summer ley	Medium Medium	Small Small	None None
Spray control[A]	3 + 8	Provides invertebrate-rich zones for breeding birds, vital for Grey Partridge and others. Allows floristic diversity which provides both summer & winter food resources.	• Reduced summer insecticide/herbicide inputs on whole crops • OS2 (includes stubble) • Conservation headlands • Decreased herbicide input for weedy stubbles (e.g. Cirl Bunting special project)	Low High Low High	Large Small Large Small	None None None None

Table 2 Continued.

			Comments			
Treatment	**Number of species benefiting**[1]	**Main benefits**	**Quality – management recommendations to ensure delivery**	**Cost**[2]	**On-farm area needed**	**Targetting Required**[3]
Extensification of grassland management^P	**3** + 5	Heterogeneous sward structure can provide suitable foraging areas with protective cover for ground-nesters. Increased floristic diversity can provide year-round food resources	• Reduced fertiliser inputs • 6 m pasture margins • Delayed cutting (for ground-nesters and seed-setting) • Reduced stock intensity • Targeted grazing regimes to produce desired sward structures	? Low Low ? High	Large Large Large Large Small	None None None None F & L

Table 2 Continued.

[1] From the total of 14 species listed in Table 1. This number is broken down into those for which we have high confidence of an effect (emboldened) and those where an effect is possible but where we cannot yet be certain (not emboldened).

[2] Cost: our estimate of whether such an option would be relatively expensive (high) or cheap (low) to implement.

[3] Targeting required: our estimate of whether the options need no targeting, targeting within farms (F) or targeting within landscapes (L).

Table 3 Summary of key ecological or demographic research required to identify, strengthen and quantify management recommendations to meet PSA target for farmland birds.

General research aim	Specific research requirement	Benefit of research with respect to PSA target	Ongoing research in this area	Priority
1. Improving demographic models for farmland birds	i) collect more extensive data on survival and dispersal	Identifies key demographic rates so management action can be targeted; increases accuracy of predicted effects of habitat management on population trends	JNCC/Dulverton Trust project *Survival and movement of seed eating birds on Scottish farmland* (BTO and Tay Ringing Group) RSPB overwinter survival project	High, for targeted 'case studies'Low across all species Major improvements will require a large research programme and the outcome of minor model improvements is unlikely to significantly alter recommendations or action.
	ii) development of techniques to measure seasonal patterns of survival	Identifies key periods of mortality more accurately so management action can be targeted. Likely to be most achievable for sedentary species		Moderate/Low
	iii) collect more extensive data on number of breeding attempts	Identifies key demographic rates so management action can be targeted; increases accuracy of predicted effects of habitat management on population trends	BTO/JNCC partnership project *Constant nest monitoring plots* (new volunteer scheme under development)	High for targeted 'case studies' Moderate across all species
	iv) incorporate density dependence	Identifies how key demographic rates are affected by changes in abundance so management action can be targeted; increases accuracy of predicted effects of habitat management on population trends		Moderate/Low An assessment of the strength of density dependence for one or two example species could be relatively easily achieved and be valuable in predicting/understanding the indicator trend.

General research aim	**Specific research requirement**	**Benefit of research with respect to PSA target**	**Ongoing research in this area**	**Priority**
	v) incorporate spatial scale effects	Increases the accuracy of predictions from demographic models	The *MONARCH* project (funded by a consortium led by English Nature) is developing approaches to investigating the impact of dispersal on bird distributional changes	Moderate/Low
	vi) inclusion of climate change and policy reform in demographic models	Increases the accuracy of predictions from demographic models	BTO/JNNC Partnership are funding some general work on the impacts of climate on survival rates and aspects of breeding performance.	Moderate
2. Improving bird habitat models	i) test generality of existing models	Increases the generality with which models can predict effect on populations of future habitat manipulations	BBSRC project *Testing the transituationality of bird habitat models in agricultural landscapes* (Oxford University & CEH, BTO, RSPB)	Priority depends on the outcome of the ongoing research – potentially high
	ii) incorporate winter habitat variables into models	Increases the accuracy with which models can predict effect on populations of future habitat manipulations	BBSRC project *Testing the transituationality of bird habitat models in agricultural landscapes* (Oxford University & CEH, BTO, RSPB)	Priority depends on the outcome of the ongoing research – potentially moderate/high
	iii) consider spatial scale effects	Increases the number of species for which accurate models can constructed and matches scales of action and monitoring		Moderate/High, since scales of monitoring must be matched to action

Table 3 Continued.

General research aim	Specific research requirement	Benefit of research with respect to PSA target	Ongoing research in this area	Priority
	iv) intensive bird-habitat studies	Increases the number of species for which models can be constructed	Data may come from current RSPB Tree Sparrow project	Moderate/Low Except for Grey Partridge, requires a relatively large investment for a small improvement of bird habitat models
	v) revisit ASPS sites to assess bird density in relation to availability of AE options	Increases the number of species and management options for which models can constructed and assesses the benefit of the ASPS options		High Could also yield valuable information with respect to (i) areas of options required and (ii) to the efficacy of agri-environment schemes
	vi) use bolt-on to BBS to assess impact of future AE schemes and quantify resource requirementements	Increases the number of species and management options for which models can constructed and assesses the benefit of the AE schemes. Also, could provide data to examine the scale at which a range of resources should be provided and to assesses the benefit of the AE schemes		High A feasibility study on the use of a 'BBS bolt on' could lead to long-term, cost-effective monitoring of the effectiveness of schemes like CSS and quantification of the scale of resources required .
	vii) examine cirl bunting case study to identify quantity of resources needed to reverse population trend	This would provide valuable information with respect to how much of a resource is required to reverse the decline of a sedentary species		Moderate This would be easily achieved but the outcome might be very species specific

Table 3 Continued.

General research aim	**Specific research requirement**	**Benefit of research with respect to PSA target**	**Ongoing research in this area**	**Priority**
	viii) examine applicability of other bird-habitat modelling approaches, such as individuals-based models	These models could allow much more accurate predictions of how much of a food resource a species requires.	DEFRA project EPG/1/5/188 *Modelling the effects on farmland food webs of herbicide and insecticide management in the agricultural ecosystem* (UEA & ADAS, BTO, CSL, CEH, GCT)	Priority depends on the outcome of the ongoing research – potentially moderate/high. Could have great potential for giving accurate predictions.
	ix) Develop models that may be used to predict the impact of the introduction of new technologies on birds e.g. GMHT crops.	These models could provide a framework for predicting the impact of a range of changes in farm management practice resulting from the introduction of new technology	DEFRA project EPG/1/5/188 *Modelling the effects on farmland food webs of herbicide and insecticide management in the agricultural ecosystem* (UEA & ADAS, BTO, CSL, CEH, GCT)	Priority depends on the outcome of the ongoing research – potentially moderate/high.
	x) develop bird habitat models for crops likely to be introduced as a result of climate change or policy reform	To allow the application of bird-habitat models as agricultural practice changes under the influence of climate change and/or policy reform		Low The predictions of climate change and associated changes in cropping patterns are still relatively imprecise.
3. Linking IPMs and bird-habitat models	i) develop case studies for selected species such as skylark, house sparrow and song thrush	Increase the number of species for which habitat management change can be linked to demographic rate change and hence to population change		Moderate These models are the key mechanism by which habitat management change can be linked to population change
	ii) develop new approaches e.g. individual covariate models	Links demography and a wider range of environmental factors through measures such as bird condition		Low – need to develop 'basic' individuals-based models first

Table 3 Continued.

General research aim	Specific research requirement	Benefit of research with respect to PSA target	Ongoing research in this area	Priority
4. Improving our understanding of resource requirements	i) intensive single species study of yellow wagtail in grass and arable	Identifies the cause of decline and key resource needs of this declining species	PhD in arable planned for Jan 2003 (UEA & AW, BTO, EN). Studies in Nene Washes (RSPB)	Priority depends on the outcome of the ongoing research
	ii) studies of interactive effects of sward structure and food abundance in determining food availability	Identifies relative importance of two factors and suggests relatively low cost, effective management options	SAFFIE CASE studentships: granivorous passerines & stubble (Oxford, RSPB), starlings and grass (Oxford, BTO)	Moderate
	iii) life-history trade-offs for 1-3 case-species	Provides an indication of how important these trade-offs might be and thus identifies when to provide resources		Moderate/Low
	iv) intensive case studies of 1-3 species in grassland	Identifies the cause of decline and key resource needs of species in grassland systems	Ongoing RSPB/EN research on yellowhammers in Shropshire	Moderate
	v) intensive case studies of key habitats in grassland e.g. hay meadows and their management	Identifies key aspects of quality and management of key resources and bird species likely to benefit		Moderate
	vi) broad scale studies of bird communities in grassland	Improves our understanding of the effect of grassland management on birds and their food resources	DEFRA project BD 1435 *Changes in lowland grassland management: effects on invertebrates and birds* (BTO & CABI, IGER) RSPB/EN project in Shropshire *Birds in intensive pastoral systems*	Low Depending on the outcome of ongoing research

Table 3 Continued.

General research aim	Specific research requirement	Benefit of research with respect to PSA target	Ongoing research in this area	Priority
	vii) large scale experiments manipulating margin plots to mimic extensive management of grass or introduction of arable pockets – monitoring bird numbers and use		DEFRA project BD1444 *Potential for enhancing biodiversity on intensive livestock farms* (IGER & BTO, CAER)	High – current research suggests that arable pockets are extremely important
	viii) large scale experiments manipulating whole fields or in- field plots to mimic extensive management of grass or introduction of arable pockets		DEFRA project BD 1444 *Potential for enhancing biodiversity on intensive livestock farms* (IGER & BTO, CAER)	High – current research suggests that arable pockets are extremely important
	ix) large scale experiments manipulating whole fields or in- field plots to mimic introduction of grass pockets in arable landscapes	Investigate the value of introducing grass pockets for improving the national status of grassland specialists on the PSA target list	RSPB-funded research at BTO using broad-scale data bases.	Priority depends on the outcome of the ongoing research
	x) intensive studies, including large-scale experiments, of movements of birds in summer and winter to investigate optimal spatial distribution of resources	Identifies the spatial scale at which to provide resources.	DEFRA project BD1618 *The consequences of spatial scale for agri-environment schemes designed to provide winter food resources for birds* (BTO & UEA, RSPB) includes small scale radio-tracking and colour-marking of birds in winter	Moderate - Priority depends on the outcome of the ongoing research but accurate direct measurements of bird movements requires a relatively large investment

Table 3 Continued.

General research aim	Specific research requirement	Benefit of research with respect to PSA target	Ongoing research in this area	Priority
5. Understanding the side-effects of providing extra resource	i) study to optimise configuration of winter food provisioning so as to maximise its effectiveness in the presence of negative side-effects	Ensure that beneficial effects of resource provision are not negated by unwanted ecological side-effects		Moderate – A pilot study can confirm whether there is a problem that needs addressing. If so, moderate investment should yield good results that can feed through to prescriptions
	ii) study to optimise configuration of nest-habitat provisioning so as to maximise productivity in the presence of negative side-effects	Ensure that beneficial effects of resource provision are not negated by unwanted ecological side-effects		Moderate – A pilot study can confirm whether there is a problem that needs addressing. If so, moderate investment should yield good results that can feed through to prescriptions

Table 3 Continued.

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APPENDIX 1: THE DEMOGRAPHY OF THE KESTREL *FALCO TINNUNCULUS* IN BRITISH FARMLAND: AN INTEGRATED POPULATION MODEL

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1.1 Introduction

The Kestrel *Falco tinnunculus* is the most common and familiar raptor across most of Britain, frequently being visible hunting over farmland and roadside verges. The species' abundance, however, tends to mask the fact that its population actually declined considerably from the mid-1970s to the mid-1980s and has yet to recover from that decline (Baillie *et al.* 2001). Prior to the population decline, Kestrel abundance had shown a steady increase as previously poor breeding success improved after the banning of organochlorine pesticides in the mid-1960s (Baillie *et al.* 2001). Through "bio-accumulation" in animals progressively higher in the food chain, organochlorine seed-dressings such as dieldrin had caused a thinning of egg-shells in many predatory birds, to the extent that incubation became impossible (refs).

The decline in Kestrel abundance after the mid-1970s ran in parallel with similar declines in many other species common on British farmland, which have been attributed to the effects of the intensification of agriculture (Marchant *et al.* 1990, Fuller *et al.* 1995, Siriwardena *et al.* 1998, Krebs *et al.* 1999). It is possible that the Kestrel trend shares the same root cause, but this has yet to be demonstrated. One approach to elucidating the causes of population changes is to identify the demographic mechanism that underlies them. By combining data on changes in abundance, annual breeding success and annual survival in an integrated population model, we can identify which of the latter two demographic rates, and, indeed, which of their component parameters, have changed to produce the population trends observed. This approach has been used successfully to clarify the processes behind the declines of species such as Song Thrush *Turdus philomelos*, Goldfinch *Carduelis carduelis*, Linnet *C. cannabina* and Reed Bunting *Emberiza schoeniclus* (Baillie 1990, Thomson *et al.* 1997, Peach *et al.* 1999, Siriwardena *et al.* 2001).

In this paper, new analyses are presented of the abundance, breeding success and survival of Kestrels in Britain since 1965. Data on abundance are derived from the British Trust for Ornithology's (BTO's) Common Birds Census (CBC), data on breeding success from the BTO's Nest Record Scheme (NRS) and data on survival from recoveries of dead birds ringed under the BTO's Ringing Scheme. The temporal variation in all components of breeding success that can be measured and in both first-year and adult survival is investigated, and an integrated model is then constructed to allow the relative importance of each variable for population change to be assessed.

1.2 Methods

Abundance: the Common Birds Census (CBC)

The CBC was the primary scheme monitoring the abundance of British farmland birds between its inception in 1962 and its replacement by the BTO/JNCC/RSPB Breeding Bird Survey in 2001. The scheme consisted of survey plots in which registrations of breeding birds were recorded annually, over 12 survey visits, by volunteer observers. Registrations were subsequently converted into maps of territories and thus counts by trained BTO staff. Although there was considerable survey plot turnover over the life of the scheme, this had little effect on its habitat coverage, which, for farmland plots, was representative of agricultural land-use in Britain south of the Humber estuary and east of the Severn (Fuller *et al.* 1985). Full details of CBC field methodology are published elsewhere (e.g. Marchant *et al.* 1990).

Long-term trends (1965-2000) in Kestrel abundance on farmland CBC plots were derived from CBC data using Generalised Additive Models (GAMs) in which count was modelled as a function of a categorical plot effect and a year effect smoothed using 10 degrees of freedom over the 36 years considered (Fewster *et al.* 2000). Bootstrapping by site (with 199 replicates) was then used to derive 95% confidence intervals, showing the precision of the estimated long-term trend.

It is reasonable to assume that a consistent relationship between the key demographic rates and abundance existed through any period during which a consistent trend occurred in abundance (stability or a smoothly increasing or declining pattern). Identifying such periods would therefore facilitate demographic analyses, particularly if data quantity is a potential limiting factor. Periods with consistent trend directions were identified objectively by estimating the second derivative (the rate-of-change of the rate-of-change) of the smoothed population trend and of each of the 199 bootstrap replicates described above (after Siriwardena *et al.* 1998, Fewster *et al.* 2000). Years during which the 95% confidence interval of the second derivative did not include zero can be interpreted as ones in which the smoothed CBC trend was turning significantly, and the one of a run of "significantly turning" years in which the significance was greatest can be taken as the best estimate of a single "turning point" (Siriwardena *et al.* 1998, Fewster *et al.* 2000). Periods between significant turning points in the CBC trend for Kestrel, identified in this way, were used in analyses of breeding success and survival as described below.

Breeding Success: the Nest Record Scheme (NRS)

Volunteer recorders have submitted nest record cards (NRCs) to the NRS since 1939. Each card features a record of a single breeding attempt over two or more visits to the nest concerned, on each of which the contents of the nest are recorded. Some or all of first egg date, clutch and brood size, chick:egg ratio and daily nest failure rates can be estimated for each NRC, depending on data quality. NRCs also include a record of the habitat in which the nest was found. The NRS is unstructured, so the spatial distribution of the data is not controlled, but the scheme receives cards from all over Britain. The scheme is reviewed in detail by Crick & Baillie (1996).

The habitat data on NRCs (Crick 1992) was used to select Kestrel data from farmland habitats for 1965 to 2000. The following variables were then derived from these data to investigate temporal changes in breeding performance per breeding attempt: first egg date (the date on which the first egg is likely to have been laid, excluding cases where the date is not known within 10 days: day 1 = 1 January), clutch size (the maximum number of eggs found in a nest), brood size (the maximum number of young found in a nest), chick:egg ratio (the ratio of brood size to clutch size where the whole nest did not fail) and daily nest failure rates before and after hatching (see below). Clutch size data were rejected if egg laying could have continued after the last visit of the recorder. Our measure of brood size is likely to overestimate the brood size at fledging, but will approach it if mortality early in nestling life (when chicks are most vulnerable) is the most significant form of partial brood loss. The chick:egg ratio measure used here will therefore incorporate these early losses, as well as hatching success (the proportion of the eggs in the clutch that hatch successfully). The number and timing of the visits (relative to nest progress) recorded on each NRC determines which of the above variables can be calculated, so the sample sizes for the analyses differ between variables.

The relationships between each nest record variable and time were investigated using generalised linear models in the GENMOD procedure of SAS (SAS Institute, Inc. 1996). NRC sample sizes were insufficient to allow the values of the variables to be estimated annually, so time (1965-2000) was parameterised using periods (blocks of years) during which the population trend had been consistent in direction (see above).

Daily nest failure rates were estimated using a formulation of Mayfield's (1961, 1975) method as a logit-linear model with a binomial error term, in which success or failure over a given number of days (as a binary variable) was modelled with the number of days over which the nest was exposed during the egg, nestling and whole nest periods as the binomial denominator (Crawley 1993; Etheridge, Summers & Green 1997; Aebischer 1999). Numbers of exposure days during the egg, nestling and whole nest periods were calculated as the mid-points between the maxima and minima possible given the timing of nest visits recorded on each NRC (note that exposure days refer only to the timespan for which data were recorded for each nest and do not represent the full length of the egg and/or nestling periods). Chick:egg ratio was also modelled with a logit link and binomial errors, brood size forming the numerator and clutch size the binomial denominator. Individually, clutch and brood sizes were modelled with identity links and normal errors, as were first egg dates. The significance of the variation between year-blocks was tested by comparing the fit of a model incorporating the temporal variation with that of an intercept-only (constant) model using a likelihood-ratio test (SAS Institute, Inc. 1996).

In order to reveal the net effects of the variation in each variable, the block-specific estimates of clutch size, chick:egg ratio and daily nest failure rates were combined to estimate the number of fledglings produced per breeding attempt, using the following formula (after Hensler 1985, Siriwardena *et al.* 2000):

$$FPA_t = CS \times CER \times (1 - EFR)^{EP} \times (1 - NFR)^{NP}, \qquad \text{eqn 1}$$

where FPA_t is the number of fledglings produced per breeding attempt, CS is clutch size, CER is chick:egg ratio, EFR and NFR are the egg and nestling period daily nest failure rates, respectively, and EP and NP are the lengths of the egg and nestling periods in days. EP and NP were taken to be the mid-points of the ranges given in Cramp & Perrins (198?): 32 and 30 days, respectively. Confidence intervals for FPA_t were calculated following the methods used in Siriwardena *et al.* (2000).

Survival: Ring-Recoveries

The BTO's ring-recovery database contains the information necessary to allow the investigation of changes in annual survival rates through the modelling of probabilities of recovery at intervals after birds are ringed (Aebischer 1985?). All available recoveries of birds ringed from 1965 onwards were used here to estimate Kestrel annual survival rates up to 1997. Because data on the numbers of birds ringed each year were not available, conditional models were fitted following the method of Aebischer (1985). Such models make an implicit assumption that the probability that a dead ringed bird will be reported does not vary over time within a cohort, i.e. for the set of birds ringed in a given year. Although there is evidence that these reporting probabilities tend to have fallen gradually over the last few decades, which is likely to cause bias in estimates of survival that do not take account of the decline (Baillie & Green 1985?), the effect will be small when the majority of birds are recovered within a few years of ringing because little change in the cohort-specific reporting rate will have occurred over that short period.

Recoveries-only survival rate models were fitted using MARK (White & Burnham 1999). Recoveries only of birds ringed between April and October (inclusive) each year (but recovered dead at any time) and which satisfied the standard data quality rules, such as the stipulation that birds not be held or transported before release (see Siriwardena *et al.* 1998b for further details), were used. Two sets of models were fitted. First, data only from birds ringed as fully-grown (independent first-summer birds or adults) were used, as in the analyses of Siriwardena *et al.* (1998b). Second, birds ringed as pulli and as fully-grown adults (but not those ringed as fledged, independent first-years) were used. The first approach has the advantages that the first-year survival rate that is estimable relates primarily to the over-winter period and that the fates of all ringed individuals can reasonably be assumed to be independent of one another. However, rather few Kestrels have been ringed as fledged, independent juveniles and the sample size of birds ringed as adults is not large. The vast majority of Kestrel recoveries come from birds ringed as pulli, so including them greatly increases the potential power of survival analyses; coincidentally, it also provides an estimate of first-year survival which encompasses the fledging-to-independence period as well as the subsequent winter. Omitting birds ringed as fledged, independent juveniles at this point means that the estimation of this survival rate is not biased by the inclusion of birds ringed when they had already survived the post-fledging period. The disadvantages of using data from birds ringed as pulli are that the dominant influence on the results is likely to be a period of high mortality when the birds have just fledged, which could therefore obscure effects on over-winter survival, and that the fates of birds ringed in the same nest are assumed to be independent when this is unlikely to be the case. Note, however, that nestling Kestrels, belonging to a species whose eggs hatch asynchronously, are likely to have fates less closely linked to those of their siblings than the chicks of most passerines. Ultimately, it was found that the results from the data set using only fledged, independent juveniles were much the weaker and that they indicated no patterns of variation in survival that were not apparent when birds ringed as pulli were used instead, so the former analyses are not discussed further in this paper.

Several basic survival rate models were fitted to the adults-and-pulli ring-recovery data set, starting with a model, S_{at}, allowing independent annual variation in both adult and first year survival. Further models were then fitted, simplifying S_{at} by removing the age-specific variation or by re-parameterizing the temporal variation as blocks of years (from the periods of consistent trend direction described above) or as a simple linear trend. The basic set of models is defined in Table 1. Comparisons of the fits of more complex and simplified models using Akaike's Information Criterion (corrected for over-dispersion: White & Burnham 1999) showed whether the differences in complexity (e.g. age-dependence in survival or variation between year-blocks) between the models concerned could be supported statistically.

Combining Abundance and Demography: Integrated Population Models

The analyses of variation in survival and breeding performance described above will identify changes in demographic rates that are consistent in direction with their having been causal factors in population change, but will not show whether the demographic changes have been large enough or that they are not outweighed by other demographic variation. Modelling the population consequences of the empirical estimates of individual demographic rates allows us to investigate both their relative importance and the ultimate implications of concurrent changes in different parameters (see, e.g., Thomson *et al.* 1997, Siriwardena *et al.* 1999, 2000a, Freeman & Crick 2000). The following equation was used to investigate the dependence of abundance upon annual survival and annual breeding performance per breeding attempt:

$$N_{t+1} = (N_t \times S^{AD}{}_t) + (N_t \times S^{FY}{}_t \times q \times FPA_t) \qquad \text{eqn 2}$$

where N_{t+1} is predicted abundance, N_t is the all plots CBC index year t, $S(ad)_t$ and $S(1^{st})_t$ are, respectively, annual adult and first-year survival rates in year t and FPA_t is the number of fledglings produced per breeding attempt in year t, as defined above. q is an unknown that represents all the demographic variation not accounted for in the variables that are considered explicitly. When $S^{FY}{}_t$ is based on the survival of fledged independent juveniles, q therefore includes post-fledging survival rates and the number of breeding attempts pairs are able to make. Kestrels, however, are single-brooded (Cramp & Simmons 198?) and, in this study, juvenile survival rates have been derived from birds ringed as pulli. Here, therefore, q represents mostly sources of error in the estimation of the various parameters, together with minor unmeasured demographic variables such as the proportion of adult birds that attempt to breed. Holding q constant therefore carries an implicit assumption that there has been no net variation in the influence of these sources of error.

Year-block-specific estimates of FPA and estimates of adult and first-year survival from the time-varying models of ring-recovery data were entered into the model above to estimate abundance (N_{t+1}) recursively, given a starting index value of one and an initial value of q=1. The best constant estimate of q, and therefore the best model using the input data used, was then determined by calculating the sum-of-squares of the differences between the model abundance indices and those from the farmland CBC (derived as described above) and by varying q iteratively in steps of 0.01 until the sum-of-squares was minimised.

Examining the fit of models with different parameters held constant or allowed to take time-varying values to the CBC trend allowed conclusions to be drawn about the importance of the variation observed in each parameter.

1.3 Results

Trends in Abundance

The farmland CBC trend, derived from the counts on XXX plots, is shown in Figure 1. The population showed a smooth, clearly statistically significant increase through the 1960s and up to about 1977, after which followed a decade during which abundance declined significantly. The population has been stable since about 1987. Examining the population trend for all CBC plots showed no important differences from the pattern for farmland alone.

Turning points in the farmland index (where the second derivative was significantly different from zero) were identified in the two periods where a visual inspection of the trend would suggest (Figure 1). The years in these two periods where the second derivative was most significantly different from zero and which were therefore taken to be the best points at which to divide the time series into blocks of consistent trend direction were 1977 and 1987.

Changes in Breeding Performance

The NRC sample sizes used to estimate the values of the various breeding performance parameters and to test for differences between blocks of consistent CBC trend are shown in Table 2. All the parameters tested differed significantly between the trend periods except chick:egg ratio (Figure 2). Particularly striking are the trends for falling nest failure rates over time during the recent period of stability (Figure 2). In addition, brood sizes have been relatively high during the recent stable period and clutch sizes were small during the 1978-1987 period of decline (Figure 2b,c). These patterns are reflected in a higher level of fledgling production during the recent stable period (Figure 2h; not testable explicitly, but the non-overlapping confidence intervals indicate a significant difference). Despite the lower clutch size during the decline, overall fledgling production was slightly higher then than during the previous period of increase, suggesting that a decline in breeding performance cannot have been the mechanism for the change in the population trend direction. However, the high breeding performance in the recent period of stability relative to the previous period of population decline suggests that this increase in productivity could have contributed to the cessation of the decline.

Note also that, although the timing of breeding has tended to be earlier in more recent years, this appears to have followed a shift to later laying after the period of increase from 1965 to 1977 (Figure 2a).

Changes in Survival

The numbers of ring-recoveries from each year of ringing of adults and pulli that were used in the analyses of survival are shown in Table 3. Figure 3 shows the fully age- and year-specific survival rates estimated under model S_{at}, as well as the trend-period-specific means taken from S_{ab}. There was no evidence of a long-term trend in the survival of either adult or juvenile (first-year) birds and the only indication of variation with respect to CBC trend was a slight increase in juvenile survival, relative to the preceding years, during the stable period after 1987 (Figure 3). In keeping with the limited evidence for temporal variation in survival apparent in Figure 3, comparing the fits of the basic survival models explicitly, using AIC values, provided no clear support for any time-trend or changes between trend periods (Table 4). Taking a difference in AIC of two or more units to indicate that a model with a lower AIC is superior, as is conventional, it can be concluded that there was clear age-dependence in survival (models S, S_b and S_t had very large AIC values) and that the annual time-dependence in model S_{at} was definitely not supported by the data (AIC value much higher than the top five models in Table 4). Indeed, the fact that model S_a had the lowest AIC value of all might suggest that there was no supportable temporal variation in survival at all, but this conclusion might be too strong given that four models each incorporating some form of temporal variation had only slightly larger AIC values (Table 4). This indicates that it would be unwise to conclude that no temporal variation has occurred, but that the variation cannot have been strong. Nevertheless, examining the results of the best models that incorporate some temporal variation should reveal the maximum likely importance of the small long-term changes that have occurred.

The two models incorporating linear time trends had the next lowest AIC values to model S_a (Table 4). Each estimated negative trends in survival, but they were very shallow and, in each case, their 95% confidence intervals overlapped zero ($S_{a,trend}$, adults: slope -0.0092, 95% CI –0.0243 – 0.0058; $S_{a,trend}$, juveniles: slope -0.0022, 95% CI –0.0149 – 0.0105; $S_{a+trend}$: slope -0.0050, 95% CI –0.0150 – 0.0054). This suggests that linear-trends are unlikely to provide a useful approach for describing the little temporal variation that has occurred in survival.

The AIC results suggest that model S_{a+b}, with parallel variation between trend periods for the two age-classes, was a marginally better model than S_{ab} (Table 4). However, this may principally reflect the fact that the former model contained two fewer parameters, more than providing an equally good fit. A glance at the estimates themselves (Figure 3) suggests that slightly more block-specific variation occurred in first-year than in adult survival. Fitting further models in which only one of adult and juvenile survival was allowed to vary with trend-period while the other was held constant provided some support for this, the model allowing only juvenile survival to vary ($S_{juv,b}$) having the smallest AIC for a model incorporating trend-period-specific variation (Table 4). In summary, the best description of the temporal variation in Kestrel survival seems to be that has not varied strongly but that there is some evidence that small long-term variations have occurred, particularly in juvenile survival.

Integrated Population Models

Figure 4 shows integrated population models fitted to the farmland CBC index from Figure 1. Two models were fitted using trend-period-specific FPA_t values (as in Figure 2h) and the estimated survival rates from either model S_{at} or model S_{ab}. Although these models were over-parameterised with respect to the data (only slightly so in the case of S_{ab}), using their outputs means that the best estimates of the possible temporal variation are used and prevents the drawing of conclusions from simple models that reflect a paucity of data more strongly than a true lack of variation in survival.

It is clear from Figure 4 that neither IPM fits the long-term CBC trend well. This means that the value of q, i.e. the influence of measurement error and any unmeasured demographic factors, has to be allowed to vary over time to produce a good fit. In biological terms, either there were significant sampling errors or biases in the demographic parameter estimates or there has been considerable, influential variation in, for example, the proportion of adults that attempt to breed. To reveal the amount of variation in q that was required to produce models that matched the CBC trend well, independent estimates of the unknown parameter were obtained, by the same iterative method, for each of the three CBC trend periods and using each of the two sets of estimated survival rates. Thus, the estimation of q for the 1977-1986 and 1987 onwards blocks was conducted by starting the sum-of-squares minimisation procedure from the year preceding the trend period concerned, the value for this year being taken from the model fit for the previous period.

It was inevitable that this procedure would provide much better fits to the CBC index and they are shown in Figure 5. The relatively large divergences between the annual survival model and CBC trend in the 1990s reflect the influence of the highly variable estimates of juvenile survival during this period (Figure 3). Much of this apparent variation is likely to reflect sampling error, and checking an annual index of abundance from the farmland CBC (i.e. an unsmoothed one) confirmed that the smoothed trend used in the model-fitting process did not mask any inter-annual fluctuations similar to those produced by the model.

The values of q providing the best-fitting models were, for the periods 1965-1976, 1977-1986 and 1987 onwards, 0.42, 0.27, and 0.29 respectively when survival rates estimated under model S_{ab} were used and 0.43, 0.26 and 0.32 when the outputs from S_{at} were used (Figure 5). It is notable that a large change in q is necessary only between the first and second trend periods. In other words, bias or variation in unmeasured demographic factors has not greatly affected the data sets and their inter-relationships after 1977, so the demographic parameter estimates and IPM fits (Figure 5) presented here for this period are likely to represent the real situation for the British population.

1.4 Discussion

The IPM results provide a reasonable description of the farmland CBC trend for Kestrel from 1977 onwards. This means that we can conclude that the cessation of the species' population decline and its subsequent stability are likely to have been driven by a combination of the slight increases in each of juvenile survival and breeding success that were identified by analyses of ring-recovery and nest record data, respectively. Running further IPMs with single, constant values for the unknown q, but considering only the data from 1977 onwards, and holding each of juvenile survival or fledgling production constant showed that the block-specific variation in both was required to produce a model population trends similar to that observed in the CBC. This suggests that both demographic changes have been important in stabilising the population.

This study provides no indication of the demographic processes that drove the switch from population increase to decline in the late 1970s. The explanation for the failure of the IPM procedure used here to identify this mechanism may be biological or methodological, i.e. due to "natural" factors not adequately accounted for by the approach or due to sampling biases, respectively. Regarding biological factors, the IPM approach makes several implicit assumptions about the variation in demographic factors that has not been measured directly and is therefore taken as having had no net effect on changes in abundance. The variation that had to be introduced into the unknown q to generate a reasonable fit to the entire CBC trend for Kestrel could show the quantitative effect of changes in these influences over time. The key possible influences that could have varied here, but were assumed to be constant, are (i) the existence of birds that are recorded in the indexing of abundance (as "holding territory") but do not breed, (ii) the existence of birds that survive but are not recorded in CBC counts (perhaps because they are itinerant) and do not breed, (iii) the probability that birds whose initial nesting attempts fail then lay replacement clutches and (iv) the occurrence of partial brood losses. The latter may be the most important for an asynchronously hatching species such as Kestrel because brood reduction is common (ref?).

Several key sampling biases may have varied over time and thus caused a lack of correspondence between the "populations" of Kestrels that have been monitored by each of the monitoring schemes used here. First, the spatial, geographical or habitat coverage of each monitoring scheme could have changed in different ways over time. Although the broad coverage of the CBC has remained consistent over time (Fuller *et al.* 1985, Marchant *et al.* 1990), ringing and nest recording activity has had no spatial structure imposed upon it, so the extent to which the same populations are sampled by the three schemes could have changed over time. In addition, even if broad-scale habitat coverage has been consistent (as the selection of NRCs by habitat code should ensure here), changes could have occurred in unmeasured, but important, habitat features such as the quality of roadside verges that are used to different extents by the birds that "contribute" to each scheme.

A second potential source of bias arises from the increase in the provision of nest boxes over time. This would cause no problem for an IPM analysis if the proportion of birds using boxes in a given year were the same in all the data sets, but it is quite likely that ringing and nest recording activity has become increasingly biased towards birds in boxes, as opposed to natural sites, whereas the CBC should be independent of changes in nest site type. Data on whether ringed birds come from nest boxes are not available, but there has been a highly significant increase in the proportion of NRCs arising from nest box nests (tested by trend period, χ^2=1122.0, 2df, P<0.001). *A priori*, boxes might be expected to provide greater protection than natural sites, especially if the latter consist of open nests, and thus lead to greater breeding success. Examining the nest record data with respect to nest site showed that nest failure rates tended to be higher in natural sites, but that this was only significant when the whole nest period was treated as having a single failure rate (likelihood-ratio test χ^2=16.94, 1df, P<0.001), although the result for the egg period also approached significance (χ^2=3.20, 1df, P=0.073). This suggests that the breeding success of the wider Kestrel population in recent years may have been over-estimated in the present study relative to that earlier in the time series considered.

Further statistical tests were conducted to investigate the influence of nest boxes on the results, examining both whether controlling for the nest site type (box or other) affected the temporal changes found and whether temporal changes have been similar in among nests in box and other sites. Controlling for nest site type, there were significant differences among trend periods in the failure rates in the egg period (P=0.025), nestling period (P=0.066) and whole nest period (P<0.001), showing declining failure rates over time. Adding the control therefore strengthens the conclusion drawn above that breeding success has increased (Figure 2). Interestingly, while these significant changes in failure rate were detectable for nests in "natural" sites (egg period P=0.041; whole nest period P<0.001; nestling period NS), there were no significant changes for birds in boxes. It appears, therefore, that improvements in breeding success have not occurred because of the provision of nest boxes and that the success of birds using boxes has not improved, perhaps because these birds have always enjoyed high breeding performance.

The effects of biased sampling involving nest boxes do not appear to explain the failure of the IPM process to predict the observed CBC trend, but data are not readily available to investigate the other possible biases. The mechanism for the switch from increase to population decline in the mid-1970s therefore remains unknown, but the effects of boxes discussed above may mean that the improvements in breeding success were under-estimated by the IPM process conducted here. Given that the statistical support for historical changes in survival was, at best, equivocal, it seems that the key demographic rate underlying major changes in Kestrel abundance in Britain since 1977 has been fledgling production per breeding attempt. As described in the Introduction, organochlorine pesticides have, historically, been important influences on Kestrel breeding success, but had been banned long before 1977. Indeed, it is difficult to suggest plausible ecological causes for the increase in breeding performance after 1986. Nest predation pressure is more likely to have increased along with rising corvid abundance (Baillie *et al.* 2001) than it is to have declined, especially for birds in natural (sometimes open) sites and there is little evidence that the probably negative effects of agricultural intensification on prey availability are likely to have eased. One possibility is that the increase reflects the release of a density-dependent constraint on breeding performance due to the fall in abundance from 1977 to 1986. The recent period of stability and the demographic rates associated with it would then represent a new demographic equilibrium at a population level considerably lower than the mid-1970s peak.

Unfortunately, the problems encountered in this study in deriving a good demographic model for the relevant period means that the IPM approach can contribute little to understanding the causes of the major population increase and decline shown by the CBC. A better understanding of Kestrel demography and ecology might be reached through contemporary intensive studies and some idea of the influences of the biases identified above might be gained by further investigation of the historical data. For ring-recovery data, such investigations would require the computerisation of data on the circumstances in which birds were ringed and careful thought would be required if spatial, geographical and habitat influences were to be investigated effectively.

Model notation	**Definition**
S_{at}	Survival allowed to vary from year to year and independently for adults and first-years
S_t	Survival allowed to vary from year to year but constrained to be constant with respect to age
S_{a+t}	Survival allowed to vary from year to year, but in parallel for adults and first-years
S_{ab}	Survival allowed to vary between year-blocks and independently for adults and first-years
S_{a+b}	Survival allowed to vary between year-blocks and in parallel for adults and first-years
$S_{a,trend}$	Survival constrained to follow linear trends for both adults and first-years
$S_{a+trend}$	Survival constrained to follow parallel linear trends for adults and first-years
S_b	Survival allowed to vary between year-blocks, with no variation with respect to age
S_a	Time-constant survival rates estimated for adults and first-years
S	A single, time-constant survival rate estimated from all recoveries

Table 1 Definitions of basic models used to estimate Kestrel survival rates.

Trend period	**Nest record card sample size**						
					Daily Nest Failure Rates (No. of Failures)		
	First egg date	**Clutch size**	**Brood size**	**Chick:egg ratio**	**Egg period**	**Nestling period**	**Whole nest period**
65-76 (+)	68	157	348	122	131 (12)	206 (11)	292 (31)
77-86 (-)	119	228	485	142	203 (10)	251 (7)	486 (22)
87-99 (=)	244	557	1036	436	398 (11)	625 (9)	958 (22)

Table 2 Numbers of nest record cards used for the analysis of each component of breeding performance.

Ringing Year	Adults	Pulli
1965	4	57
1966	4	49
1967	3	60
1968	5	78
1969	2	55
1970	1	66
1971	2	80
1972	2	98
1973	4	93
1974	3	81
1975	7	104
1976	5	92
1977	17	104
1978	16	100
1979	9	108
1980	10	96
1981	12	123
1982	14	80
1983	10	108
1984	8	125
1985	10	97
1986	10	82
1987	7	86
1988	8	145
1989	8	120
1990	4	133
1991	5	105
1992	5	125
1993	3	105
1994	2	108
1995	1	99
1996	2	108
1997	0	63
1998	2	63
1999	1	45
2000	0	33

Table 3 Annual ring-recovery sample sizes for birds ringed as adults and as pulli.

Model	QAICc	No. of Parameters in Model
Basic models		
S_a	3537.5	2
$S_{a+trend}$	3538.6	3
$S_{a,trend}$	3540.1	4
S_{a+b}	3541.0	4
S_{ab}	3542.7	6
S_{at}	3613.8	70
S_t	3774.6	35
S	3847.5	1
S_b	3850.8	3
Additional models		
$S_{juv,b}$	3539.1	4
$S_{ad,b}$	3540.9	4

Table 4 Models of the age- and time-specific variation in Kestrel annual survival rates. Models are shown in order of corrected quasi-likelihood AIC (QAICc) values, the lowest first. QAICc values are an adaptation of the standard AIC to cater for overdispersion in the data (White & Burnham 1999). The value for the overdispersion coefficient $\hat{c}$=2.326 for these models was taken from the ratio of the deviance of the full model S_{at} to its degrees of freedom (White & Burnham 1999).

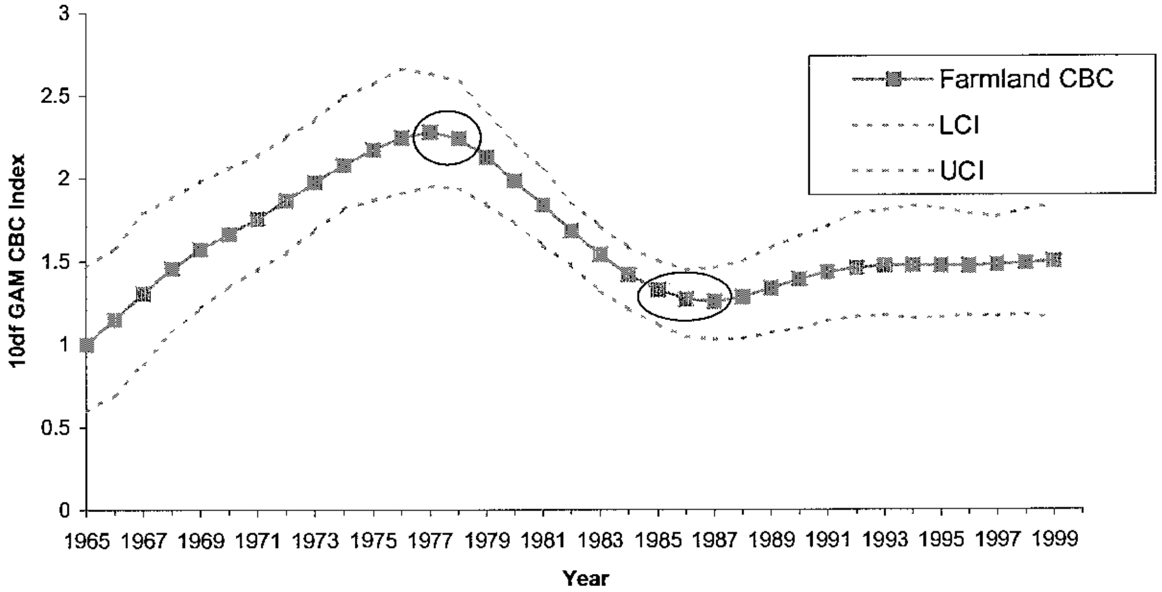

Figure 1 Smoothed farmland CBC index (derived using a GAM with 10df) for 1965-1999. The dotted lines show 95% confidence intervals and the circled points show years in which the trend was found to be turning significantly.

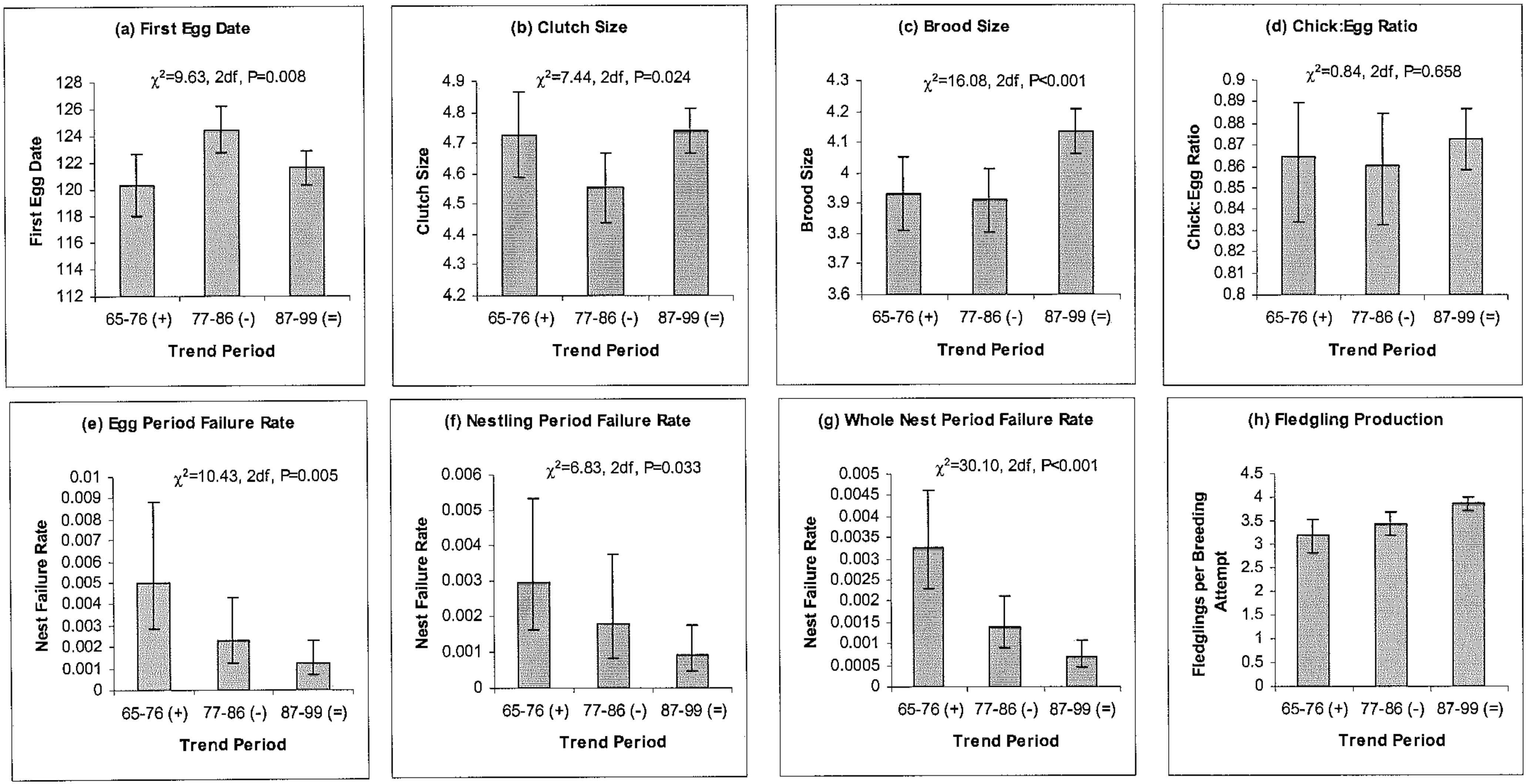

Figure 2 Variation in components of breeding performance between periods of consistent CBC trend. Error bars show 95% confidence intervals in each case and c2 statistics and P-values refer to likelihood-ratio tests of the significance of the variation between trend periods: (a) first egg date, (b) clutch size, (c) brood size, (d) chick:egg ratio, (e) egg period, (f) nestling period, (g) whole nest period, (h) fledgling production FPA_t.

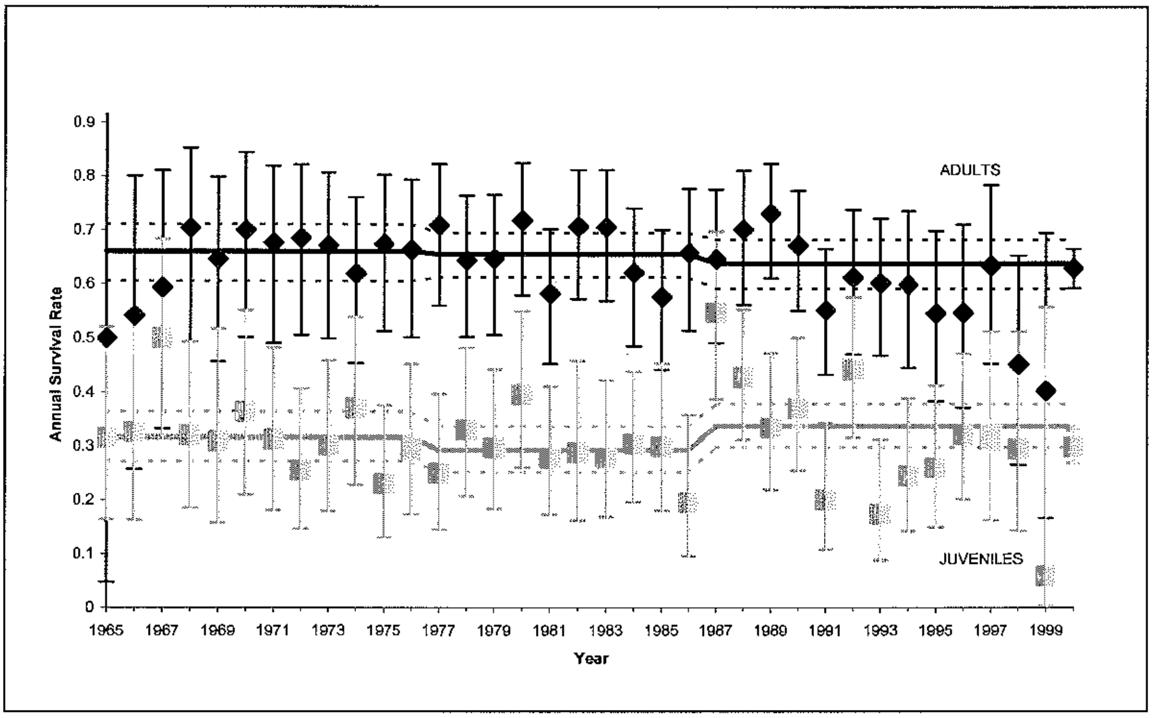

Figure 3 Annual estimates of adult and juvenile survival rates from model S_{at} (points with error bars, which show 95% confidence intervals) and from model S_{ab} (solid lines showing block-specific estimates, around which the dashed lines show 95% confidence intervals).

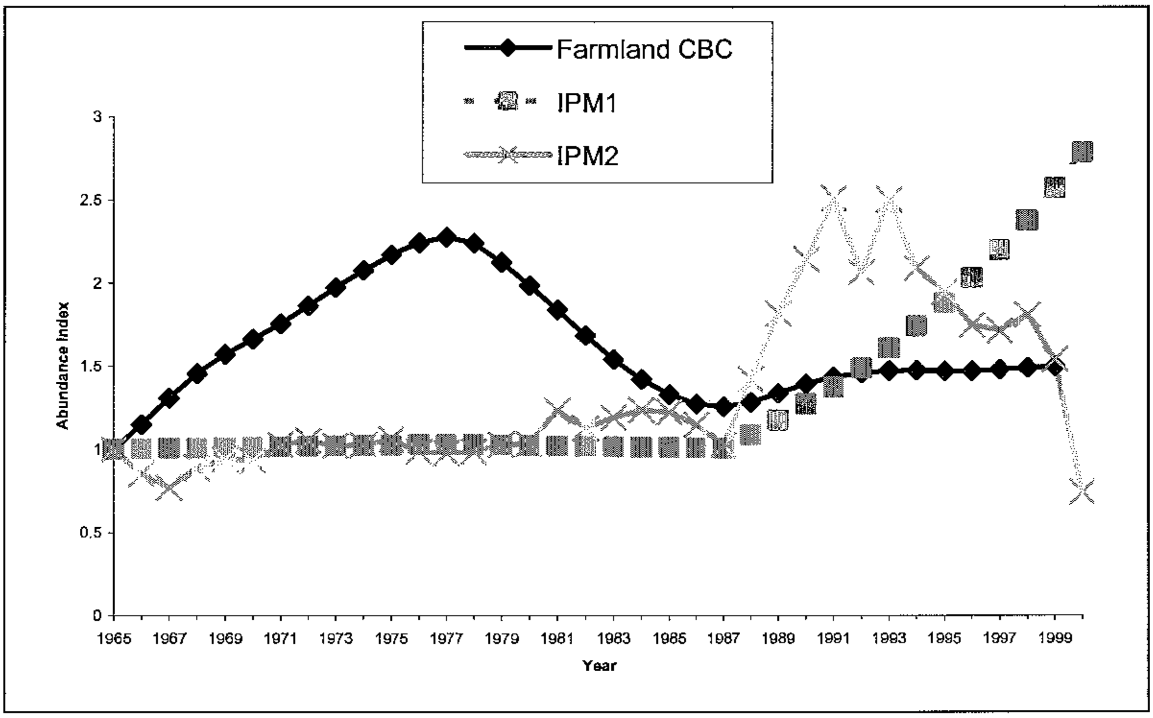

Figure 4 The farmland CBC trend for Kestrel and IPMs fitted using trend-period-specific estimates of fledgling production and either trend-period-specific or annual estimates of adult and juvenile survival. Single estimates of q were made for each model by iteration to provide the closest possible fit to the CBC trend.

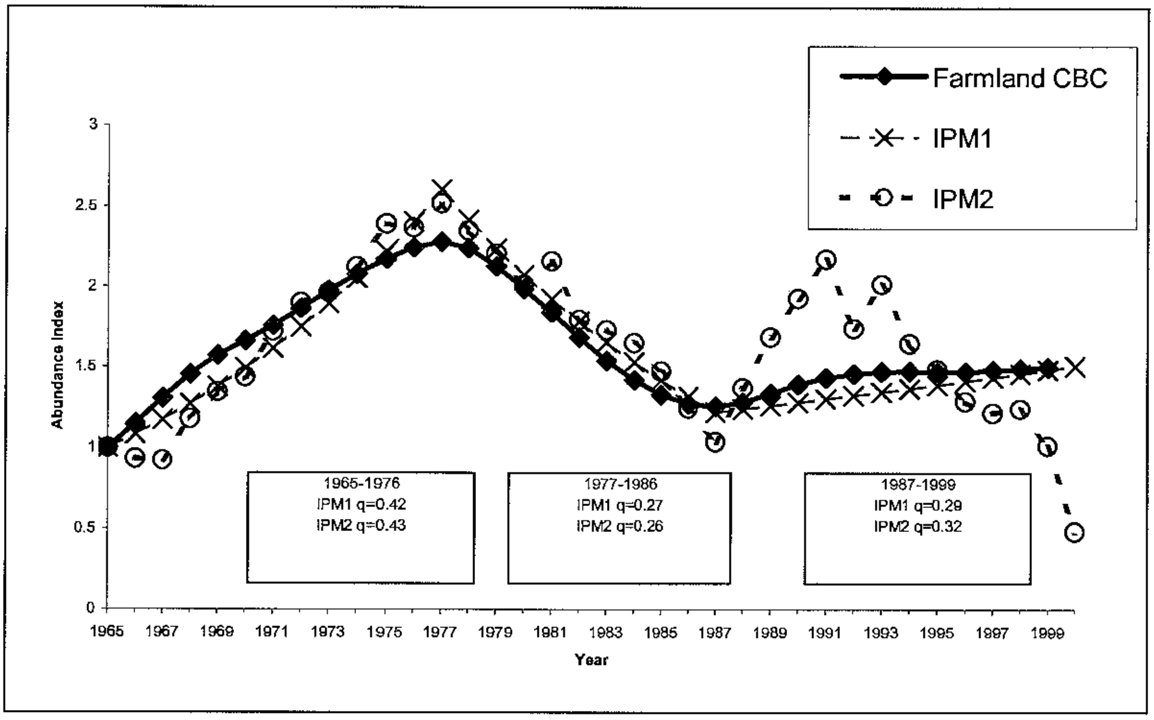

Figure 5 The farmland CBC trend for Kestrel and IPMs fitted using trend-period-specific estimates of fledgling production and either trend-period-specific or annual estimates of adult and juvenile survival. Three estimates of q, one for each trend period, were made for each model by separate iteration processes to provide the closest possible fit to the CBC trend.