



ELSEVIER

Available online at www.sciencedirect.com

SCIENCE @ DIRECT®

Agriculture, Ecosystems and Environment 106 (2005) 1–16

**Agriculture
Ecosystems &
Environment**

www.elsevier.com/locate/agee

Indirect effects of pesticides on breeding yellowhammer (*Emberiza citrinella*)

Antony J. Morris^{a,*}, Jeremy D. Wilson^b, Mark J. Whittingham^c,
Richard B. Bradbury^a

^aThe Royal Society for the Protection of Birds, The Lodge, Sandy, Beds SG19 2DL, UK

^bRSPB, Dunedin House, 25 Ravelston Terrace, Edinburgh EH4 3TP, UK

^cDepartment of Zoology, Edward Grey Institute of Field Ornithology, South Parks Road, Oxford OX1 3PS, UK

Received 5 June 2003; received in revised form 1 July 2004; accepted 21 July 2004

Abstract

Intensification of agriculture is believed to have caused declines in farmland bird populations. One of the key elements of recent agricultural intensification is the increased use of pesticides. However, studies elucidating relationships between individual management practices and responses in bird populations remain rare. Here, evidence is presented of indirect effects of pesticides on behaviour and nestling condition of yellowhammers (*Emberiza citrinella* L.). Insecticide use was associated with reduced abundance of invertebrate food at the field scale resulting, early in the season (when nestlings were fed exclusively on invertebrates), in a negative correlation with yellowhammer foraging intensity. There was also a negative relationship between insecticide use and nestling body condition. While cumulative effects of repeated spraying can have impacts, the occurrence of any insecticide spraying in the breeding season may be more detrimental than multiple sprays at other times. Minimising applications of persistent broad-spectrum insecticides during March–June, the provision of alternative unsprayed foraging habitat and advice on mitigating indirect effects of pesticides to advisers and users are likely to benefit nesting yellowhammers. © 2004 Elsevier B.V. All rights reserved.

Keywords: Agricultural intensification; Farmland; Insecticide; General linear mixed modelling; Vacuum sampling

1. Introduction

The last three decades have seen changes in many aspects of agricultural practice in the UK (Shrubb, 2003), Europe (Donald et al., 2002) and North

America (McLaughlin and Mineau, 1995), most of which have reflected policy and technology-driven increases in agricultural productivity. Over a similar time-span, a number of UK farmland bird species have declined in population or range (Gibbons et al., 1993; Gregory et al., 2002), at least partly because of more intensive agricultural practices (Fuller, 2000). In Europe, the extent of bird population declines in individual countries is correlated with the degree of

* Corresponding author. Tel.: +44 1767 680551;
fax: +44 1767 683508.

E-mail address: tony.morris@rspb.org.uk (A.J. Morris).

agricultural intensification (Donald et al., 2001). A variety of causal processes, including the use of agrochemicals to control weed and invertebrate pests, have been proposed (Potts, 1970, 1986; O'Connor and Shrubbs, 1986; Campbell et al., 1997; Pain and Pienkowski, 1997).

In Britain, the area of land sprayed with each of the three commonest pesticide categories (fungicides, herbicides and insecticides) has increased since the early 1970's (Chamberlain et al., 2000). There is little evidence that current pesticide usage in the UK is directly affecting birds in the same way as the organochlorines did in the 1960's and 1970's (Burn, 2000). However, there may be indirect effects via the depletion of the food supply, as herbicides and insecticides may directly affect the abundance of seed and invertebrates, respectively, while herbicides indirectly impact the diversity and abundance of the plant foods of invertebrates (Potts, 1986; O'Connor, 1992; Campbell et al., 1997).

An indirect effect of pesticides on the abundance of invertebrate food for chicks, on chick survival, and on breeding population size has been demonstrated by a fully replicated field experiment for the grey partridge (*Perdix perdix* L.) (Rands, 1985). However, the possible impacts of increasing pesticide use remain unknown for the many passerine species that have exhibited population declines and range contraction coincident with agricultural intensification. This study aimed to test whether pesticide use on arable farmland was associated with variation in foraging behaviour and reproductive performance of yellowhammers (*Emberiza citrinella* L.) and with variation in abundance of invertebrate food supplies.

The yellowhammer was chosen as the study species for a variety of reasons. First, although it remains a widespread farmland species, it has shown a marked decline in the UK since the 1980s (Gregory et al., 2002). The start of decline in the mid-1980's occurred during, or shortly after, a period when the land area treated with fungicides, and especially insecticides, had increased rapidly (Campbell et al., 1997). Although the greatest declines in yellowhammer abundance were associated with removal of cereals from the landscape, populations have also declined markedly in arable areas due to more subtle changes in management (Kyrkos et al., 1998). Second, nestling diet, although known to include semi-ripe cereal grain,

is mainly comprised of a range of invertebrates (Wilson et al., 1996; Stoate et al., 1998). Some of these have undergone long-term decreases (Benton et al., 2002), although evidence for trends on farmland is limited (Aebischer, 1991). There is also evidence that food taxa (Arachnida, Curculionidae, Staphylinidae, Hymenoptera, larval Lepidoptera and Orthoptera) listed as 'important' in yellowhammer diet by Wilson et al. (1996) have been reduced in abundance in the short-term in response to pesticide use, especially insecticides (Ewald and Aebischer, 1999; Wilson et al., 1999). The scale of short-term effects of pesticides suggests that they may be important factors in determining the abundance of many important food taxa and consequently in determining the behaviour of adult yellowhammers foraging for nestling-food and the resultant development of their nestlings. Arable crops (most of which receive multiple pesticide applications) are extensively used as sources of both plant and invertebrate food (Stoate et al., 1998; Morris et al., 2001).

2. Methods

2.1. Study sites

This study was carried out as part of a research programme on the indirect effects of pesticides on birds, commissioned by the UK Government's Department of Environment, Food and Rural Affairs' (DEFRA). It was based in Oxfordshire, where data on yellowhammer breeding success and foraging habitat selection had already been collected during 1995–1997. Study sites were chosen to reflect the predominant type of local farming (mixed grass/arable) and pesticide regimes that approximated to the national average. During 1999–2001, additional yellowhammer data were collected from three existing, and five new, study sites, plus field-by-field pesticide use data for all the study sites and years. Data on invertebrate food abundance were sampled on a subset of three Oxfordshire sites in 2000 and two sites (one in Oxfordshire and one in Lincolnshire) in 2001 (Table 1). Data on all fungicide, herbicide and insecticide applications were obtained from landowners, tenant farmers and spraying contractors for arable and non-grass forage crops. Data on the timing

Table 1
Number of farms surveyed

Year	No. of farms	Foraging data – no. of nests	Nestling condition data – no. of nests	Nestling starvation data – no. of nests	Invertebrate data – no. of fields
1995	Farms 1 and 2	21	9	14	0
1996	Farms 1–3	19	31	32	0
1997	Farms 1–3	10	22	22	0
1999	Farms 1, 3–7	31	14	18	0
2000	Farms 1, 2, 7	25	25	27	20
2001	Farms 3 and 8	21	21	17	8

and the numbers of applications, the proportion of the study areas sprayed and the mean application rates for each of the active ingredients used on the study sites are given in Appendix A. Details of the crops grown were also collected and included in the analyses.

2.2. Fieldwork

Fieldworkers located the nests within field boundaries, by mapping territorial males and watching for signs of breeding behaviour between May and August. Data on foraging site selection, nestling body condition and nestling starvation were obtained from nests for which pesticide data were available from all arable fields adjacent to the nest (Table 1).

Records of foraging site selection by adult yellowhammers provisioning nestlings aged ≥ 4 days old were derived from 1 to 3 observation periods of 60–90 min. Only nests for which ≥ 10 flights were observed to an identifiable foraging location were included. Observations on foraging flights from the nest were conducted according to Morris et al. (2001).

To obtain data on nestling body condition, nestlings were weighed with a Pesola spring balance and tarsus length, from the depression in the angle of the intertarsal joint to the end of the folded foot, was measured with dial callipers. The age of the nestlings was known from the date of hatch or was estimated by comparison of feather development with known-age broods. Weight and tarsus length were recorded for 4–7 day old nestlings, the period during which growth rate is linear. Nests were visited every 2–4 days to determine outcome and to ascertain whether brood reduction occurred through starvation.

To test whether relationships between pesticide use and breeding yellowhammers were likely to be attributable to pesticide depletion of nestling-food

resources, in 2000–2001 invertebrate populations were sampled in 28 cereal fields adjacent to yellowhammer nests. As yellowhammer nestlings are fed on a wide-range of invertebrates gleaned from the ground or from the base of plant stems (Kear, 1962; Prŷs-Jones, 1977), sampling occurred only via vacuuming.

Sampling was conducted in dry weather, following yellowhammer foraging observations, using a modified petrol-motor leaf-suction device (Stihl BG 75) equipped with a 10 cm diameter suction-hose. Sampling was confined to cereals, because (i) they comprised 70% of arable crops adjacent to nests, (ii) they received a high percentage of all foraging visits (77%) and pesticide applications (79%) and (iii) this minimised the degree to which differences in vegetation structure influenced the efficacy of the catch.

Four samples per field, each of five 10 s sucks over an area of approximately 2 m², were located on transects along the longest diagonal axis of each field at 20, 40, 80 and 160 m into the crop. Samples were stored in bags, frozen and placed in warm water to float off the invertebrates from the soil and vegetation. Invertebrates > 2 mm in length were identified to order or family. Smaller invertebrates were not recorded, as they were considered unimportant components of yellowhammer diet (Bösenberg, 1958). The invertebrate samples were then dried, weighed on a microbalance to obtain a total biomass for the sample, and counted to give the number of individuals per order/family.

2.3. Data analyses

General linear mixed modelling (GLMM) procedures in Genstat 5 were used to represent site or nest as a random factor and to identify predictor variables

Table 2
The variables entered into the full yellowhammer foraging, nestling-condition, brood reduction and nestling-food models

Predictor variables	Response variables			
	Foraging (both models)	Condition	Brood-reduction	Invertebrate food (all models)
Farm		Random factor, 7 levels: farms 1–7	Random factor, 7 levels: farms 1–7	Random factor, 5 levels: farms 1, 2, 3, 7, 8
Nest	Random factor: 66 levels in early nests model; 61 levels in late nests model			
Year	Fixed factor, 6 levels: 1995, 1996, 1997, 1999, 2000, 2001	Fixed factor, 6 levels: 1995, 1996, 1997, 1999, 2000, 2001	Fixed factor, 6 levels: 1995, 1996, 1997, 1999, 2000, 2001	Fixed factor, 2 levels: 2000, 2001
Crop type	Fixed factor, 5 levels: 1 = w: wheat; 2 = barley; 3 = legumes; 4 = maize; 5 = oilseed rape			Fixed factor, 2 levels: 1 = spring-sown cereal; 2 = winter-sown cereal
Date	(Not included but separate early nests and late nests models reflect date)	Fixed factor, 2 levels: 1 = early nests; 2 = late nests	Fixed factor, 2 levels: 1 = early nests; 2 = late nests	Continuous
Nestling age		Continuous		
Brood size		Continuous	Continuous	
Foraging distance	Continuous			
Total number of fungicide applications	Continuous	Continuous	Continuous	Continuous
Timing of fungicide applications ^a	Fixed factor, 2 levels: 1 = none in crop year; 2 = breeding season	Fixed factor, 2 levels: 1 = none in crop year; 2 = breeding season	Fixed factor, 2 levels: 1 = none in crop year; 2 = breeding season	Fixed factor, 2 levels: 1 = none in crop year; 2 = breeding season
Total number of herbicide applications	Continuous	Continuous	Continuous	Continuous
Timing of herbicide applications	Fixed factor, 2 levels: 1 = no breeding season; 2 = breeding season	Fixed factor, 2 levels: 1 = no breeding season; 2 = breeding season	Fixed factor, 2 levels: 1 = no breeding season; 2 = breeding season	Fixed factor, 3 levels: 1 = none in crop year; 2 = non-breeding only; 3 = breeding season
Total number of insecticide applications ^b	Continuous	Continuous	Continuous	
Timing of insecticide applications	Fixed factor, 2 levels: 1 = no breeding season; 2 = breeding season	Fixed factor, 2 levels: 1 = no breeding season; 2 = breeding season	Fixed factor, 2 levels: 1 = no breeding season; 2 = breeding season	Fixed factor, 3 levels: 1 = none in crop year; 2 = non-breeding only; 3 = breeding season

^a In no case was non-breeding-only application of fungicide recorded.

^b In the invertebrate-food dataset, all cases of multiple insecticide applications were confounded with summer use. Therefore, the ‘number of insecticide applications’ predictor was excluded from the analyses. Post-hoc contrasting, following the method of Crawley (1993), was used to examine differences between levels of the ‘timing of pesticide application’ factors.

explaining significant variation in the response variables (Welham, 1993; Milsom et al., 2000). All analyses were conducted using a step-up procedure (in which each variable was added and then deleted from the model in turn, with the most statistically significant variable re-fitted to the model after each iteration) to establish the minimum adequate model (MAM). All predictor variables considered in the analyses are listed in Table 2. Any overdispersion in the data was automatically corrected by Genstat procedures.

A large range of active substances were used on the study sites but the total number of applications was relatively low (Appendix A) and, in many cases, tended to be site specific. This meant the models had insufficient power to detect effects of individual active substances. For fungicides, the problem was exacerbated by the use of mixes of chemicals applied at the same time. As a result, all fungicide applications were combined for the purpose of the analyses. In preliminary analyses, herbicides were divided into broad and narrow-spectrum products, based on the classification of Ewald and Aebischer (1999) and using additional information from Whitehead (2000). However, as there were no significant differences in effects on invertebrate numbers or foraging patterns, all herbicide products were subsequently combined. 'Spot-spray' herbicide treatments were discounted from the analyses if they were applied to <25% of the field area, as food resources in much of the field were not in direct contact with the active substance(s). Pyrethroid compounds accounted for 85% of insecticide sprays on the study sites (Appendix A). As there were too few data on non-pyrethroid insecticides to examine their effects separately, all insecticide treatments were combined for the purpose of the analyses.

For each of the three pesticides categories, two measures of use per field were derived for inclusion in the models: (i) the total number of applications during the crop-year, (ii) the timing of the last application prior to yellowhammer egg hatching, expressed as 'none' (untreated during the relevant crop-year), 'non-breeding season only' (late August–February) or 'breeding season' (March–early August) applications. The two measures were chosen as they represented two types of ecologically realistic scenarios: cumulative effect over the agricultural year and short-term

impact during the breeding season, when demand for invertebrate food is greatest. Mean application rate (total mass of pesticide applied/total sprayed area) was not considered, as it is generally thought to be a poor indicator for assessing pesticide impacts on wildlife (Campbell et al., 1997).

Samples from the 20, 40, 80 and 160 m locations were pooled for each of the 28 different cereal fields. Thirteen fields were sampled on multiple occasions, as more than one yellowhammer brood was found adjacent to that field. In these cases, mean invertebrate abundance was calculated from all sampling visits. For fields sampled on multiple occasions and receiving a pesticide application in the intervening period, the mean number of applications was used in the analysis. To find which pesticide variables explained significant variation in the invertebrate fauna, models (with Poisson error structures and log link functions) were constructed for four invertebrate response variables: (i) the total number of invertebrates sampled that have been recorded in yellowhammer diet, (ii) the total number of invertebrates sampled classified as 'important' in yellowhammer diet (as defined by Wilson et al., 1996), (iii) the total number of invertebrates with body size ≥ 5 mm, (iv) the total invertebrate sample biomass. As these four measures originated from the same samples, Bonferroni correction for multiple tests was applied.

Yellowhammer nestling diet includes cereal grain, once it becomes semi-ripe (Wilson et al., 1996; Stoate et al., 1998). As pesticide applications do not adversely affect the supply of grain, it was expected that pesticides would be less influential in determining foraging patterns of birds exploiting this food resource. From the foraging observations, the thresholds for the availability of edible grain were determined as June 21st for barley and July 7th for wheat. Therefore, nests were divided into two seasonal groups, 'early' (which also included three late nests where grain was unavailable, there being no cereal fields within 300 m) ($n = 66$), and 'late' ($n = 61$).

The early and late nests were analysed using separate models. The log area of the entire field was specified as an offset, such that the number of foraging visits per ha for each field equated to a 'foraging density' (Morris et al., 2001). The energetic cost of each foraging trip was assumed to be of similar value, regardless of the area visited and field condition. As

most nests did not contain nestlings contemporaneously and the pesticide applications to which they were subjected often varied, ‘nest’ was modelled as a random factor, as it represented an individual pair of birds not replicated elsewhere in the analysis.

To maximise the chance of cumulative environmental effects on condition, while minimising the bias of confounding effects related to nestling age, nestling condition was calculated using mass and tarsus length from the final measurement made during the linear growth phase of the nestlings (Bradbury et al., 2003) and nestling age was also included in the model. In addition, the last measurement provided the closest measure to fledging condition, an important measure of fitness because it correlates with subsequent survival in many species (e.g. Magrath, 1991).

Variation in the mean condition of each brood was analysed as a function of the predictor variables, assuming a normal error distribution and an identity link function. Following García-Berthou (2001), a brood condition score was calculated taking the log brood mean mass, with log brood mean tarsus length included as a covariate in the model to control for the age of nestlings. All analyses were simplified by excluding any broods in which there was reduction prior to the measurement used to calculate the condition index. Brood reduction could complicate the analysis of nestling condition for two reasons. First, assuming the nestlings that die were those in poorest condition, brood mean condition is immediately inflated relative to unreduced broods. Second, the remaining nestlings may be able to respond to the reduced sibling competition by increasing condition or growth rate, again inflating these measures relative to unreduced broods (Shkedy and Safriel, 1992). Nevertheless, brood reduction is a more drastic event than a reduction in nestling condition, so variation in brood reduction from starvation was also analysed.

A model constructed with a binomial error structure and a logit link function was used to find which variables explained a significant proportion of nestling mortality arising from starvation. The proportion of nestlings starved in a brood was specified as the response variable, with the binomial denominator specified as one (to represent the original brood size at hatching), depredated broods being excluded from the analysis.

3. Results

3.1. Pesticides

Of 351 field-years, 62% were sprayed with fungicide, 99% with herbicide and 49% with insecticide, with 97% of fungicides, 76% of herbicides and 27% of insecticides being applied during the period March–August. There were only weak to moderate correlations between the number of fungicide and the number of herbicide applications ($r_s = 0.50$), the number of fungicide and the number of insecticide applications ($r_s = 0.46$) and the number of herbicide and the number of insecticide applications ($r_s = 0.36$), indicating that the data were unlikely to be unduly affected by multicollinearity arising from the simultaneous application of different types of pesticides. For the four commonest arable crops on the study sites (winter wheat, $n = 219$ fields; winter barley, $n = 76$ fields; oilseed rape, $n = 59$ fields; spring barley, $n = 46$ fields), the annual number of pesticide applications per crop was compared to the mean of the national average figures quoted by Thomas et al. (1997) and Garthwaite and Thomas (1999). There were fewer applications than the national average for insecticides on winter barley, for herbicides on spring barley and for fungicides on rape, winter barley and wheat. The number of applications on the study sites was never significantly higher than the mean national average. Pyrethroid compounds accounted for 86% of the insecticide treated area on the study sites, slightly higher than the national figure of 79% (Garthwaite and Thomas, 1999).

3.2. Associations between pesticide use and yellowhammers

All final nestling-food models retained timing of insecticide use (Table 3). Invertebrate numbers and biomass were significantly lower in fields treated with insecticide during the breeding season. Invertebrate numbers were generally lower in fields without any application than in fields receiving non-breeding season applications (Figs. 1 and 2). The total number of fungicide applications was negatively correlated with the abundance of ‘all invertebrates’ (Fig. 3). There was a curvilinear relationship, with a markedly negative component, between total number of

Table 3

Significance levels^a, direction of relationships^b and post-hoc contrasting between factor-levels^c for the variables retained in the foraging, nestling-condition, nestling-starvation and nestling-food models

Model	Foraging: early nests	Foraging: late nests	Condition	Starvation	Nestling-food: all invertebrates	Nestling-food: 'important' invertebrates	Nestling-food: invertebrates \geq 5mm	Nestling-food: sample biomass
Timing of insecticide applications	Wald 4.63 [*] ; df = 1; 1 > 2	ns	ns	ns	Wald 23.50 ^{***} ; df = 2; (2 \gg 1) \gg 3	Wald 13.69 ^{***} ; df = 2; (2 = 1) \gg 3	Wald 27.56 ^{***} ; df = 2; (2 = 1) \gg 3	Wald 13.72 ^{***} ; df = 2; (2 = 1) \gg 3
Total number of insecticide applications	ns	ns	$F_{2,122}$ 6.59 [*] ; \downarrow	ns				
Total number of fungicide applications	ns	ns	ns	ns	Wald 7.49 ^{**} ; df = 1; –	ns	ns	ns
Total number of herbicide applications	ns	ns	ns	ns	ns	ns	Wald 8.91 ^{**} ; \downarrow df = 1;	ns
Year	ns	ns	$F_{5,122}$ 7.47 ^{***}	ns	ns	ns	ns	ns
Date			ns	ns	Wald 4.05 [*] ; df = 1; +	ns	ns	Wald 6.77 [*] ; df = 1; +
Brood size			$F_{1,122}$ 6.80 ^{**} ; –	ns				
Foraging distance	Wald 28.61 ^{***} ; df = 1; –	Wald 18.46 ^{***} ; df = 1; –						

Shading = not included in full model.

^a Wald tests: *** $P < 0.001$, ** $P < 0.01$, * $P < 0.05$; ns: variable not retained in MAM. In the four nestling-food models, all differences at $P < 0.01$ were significant after Bonferroni correction for multiple tests.

^b (+) positive effect; (–) negative effect; (\downarrow) curvilinear effect (initial positive effect with strong negative tail).

^c Insecticide timing: (1) no, (2) winter only, (3) summer, (=) non-significant differences, (>) significant differences between adjacent ranks at $P < 0.05$, (\gg) significant differences between adjacent ranks (or, when bracketed, both preceding ranks) at $P < 0.01$.

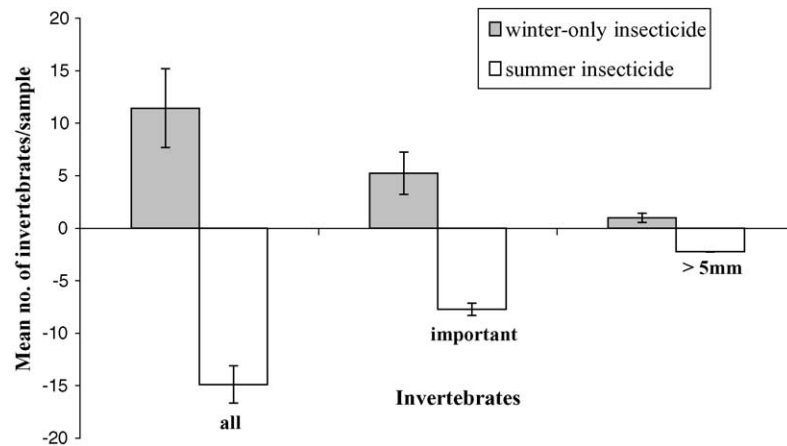


Fig. 1. GLMM predictions of the mean number of individuals per sample (with standard errors) for (1) all invertebrate taxa recorded in yellowhammer diet, (2) invertebrate taxa most important in yellowhammer diet, (3) 'large' invertebrates ≥ 5 mm, for fields with winter-only insecticide and summer insecticide relative to fields with no insecticide.

herbicide applications and total number of invertebrates with body size ≥ 5 mm (Fig. 4).

As the use of pesticides during the non-breeding period generally had no adverse effects on nestling-food abundance, in the bird models, fields with no or non-breeding season applications were grouped and compared to breeding season applications.

For early nests, the timing of fungicide or herbicide applications did not significantly affect foraging patterns. However, there was a significant effect of the timing of insecticide application. Fields that

received breeding-season insecticide were used significantly less than fields that received no breeding season insecticide (Table 3). Back-transformation of parameter estimates for the early nests showed that foraging density in fields without breeding season applications of insecticide was nearly four times higher than in fields with breeding season applications (Fig. 5). Adults rarely foraged more than once in fields with breeding season insecticide compared to fields without breeding-season insecticide (9.5% versus 32.4% of all such fields). Foraging density was also

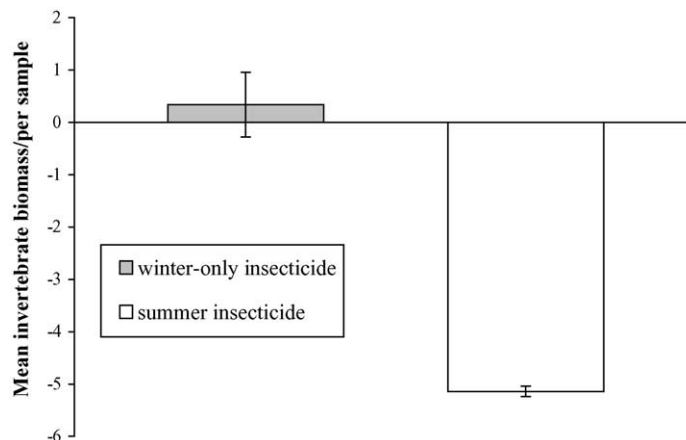


Fig. 2. GLMM predictions of the mean invertebrate sample biomass (mg) (with standard errors) for fields with winter-only insecticide and summer insecticide relative to fields with no insecticide.

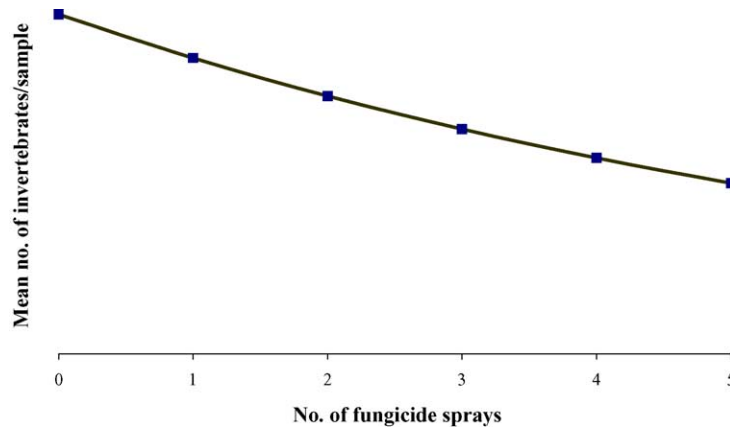


Fig. 3. GLMM predictions of the mean number of invertebrates per sample for all taxa recorded in yellowhammer diet for increasing numbers of fungicides relative to no application.

negatively related to increasing distance of a foraging location from the nest (Table 3). The total number of product applications did not have a significant influence on use of fields by yellowhammers provisioning nestlings for any of the three pesticide types considered.

No pesticide variables was retained in the late nests model, although, as in the previous model, foraging density declined with distance from the nest (Table 3). There was no difference in the mean distance of foraging flights between the two models.

There was a curvilinear relationship between mean brood condition and number of insecticide

applications (Table 3), with a weak positive relationship up to one application and a strongly negative relationship with two or more applications (Fig. 6). Other non-pesticide predictors retained in the MAM were year and size of brood, with poorer mean condition in larger broods. No predictor accounted for significant variation in brood reduction arising from starvation (Table 3). For 23 nests with known initial brood size and data on the abundance of 'important' invertebrates from adjacent cereal fields, there was a significant negative relationship between 'important' invertebrates and starvation (Wald 4.19; $P = 0.041$; $df = 1$).

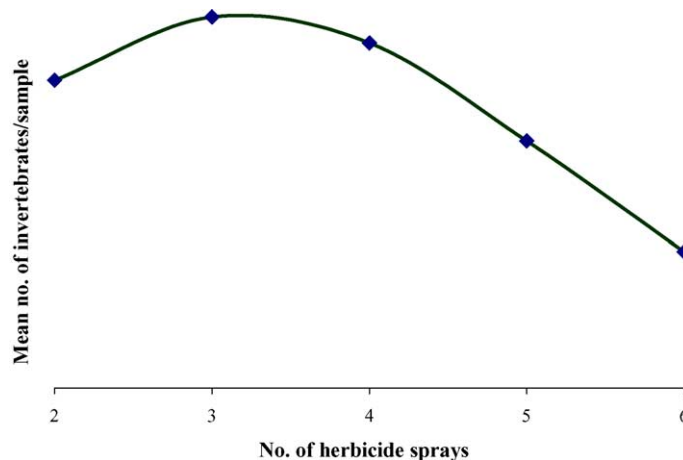


Fig. 4. GLMM predictions of the mean number of individual 'large' invertebrates (≥ 5 mm) per sample for increasing numbers of herbicides relative to no application.

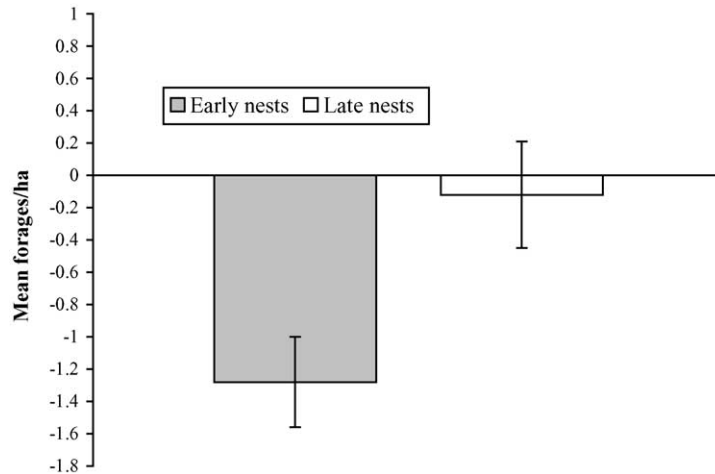


Fig. 5. GLMM predictions for early and late yellowhammer nests of the mean number of foraging visits per ha (with standard errors) to fields with summer insecticide relative to fields with no-summer insecticide.

4. Discussion

Most of the pesticide associations noted were attributable to insecticide use. According to Burn (2000), direct effects on small passerines may be under-recorded, yet the effects on yellowhammers noted in this study were more likely to arise from depletion of invertebrate food. The significant negative relationship between abundance of ‘important’ invertebrates and brood reduction from starvation supports this conclusion. Insecticide use in summer is one factor that can result in a significant decrease in invertebrate abundance. Only six cereal fields sampled

for invertebrates were sprayed with insecticide in the breeding seasons 2000–2001, with five of these fields situated on one farm in 2001. This arose in part because aphid numbers on Oxfordshire farms were very low during the duration of the study (A. Richards, personal communication), and because of access constraints imposed by the outbreak of foot and mouth in 2001. Nonetheless, the results agree with the findings of other authors, who have reported negative correlations between insecticide use and the abundance of invertebrates consumed by yellowhammer nestlings, including Orthoptera, larval Lepidoptera, Symphyta and Opiliones (e.g. Ewald and Aebischer,

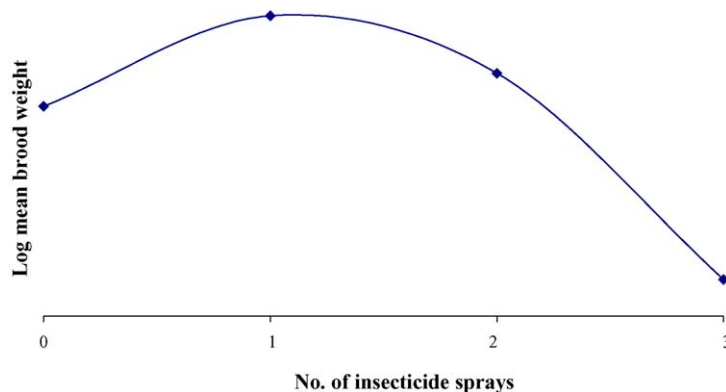


Fig. 6. GLMM predictions of the mean log body weight for yellowhammer broods subjected to increasing numbers of insecticides relative to no application.

1999; Wilson et al., 1999). The significantly greater abundance of ‘all invertebrates’ in fields with non-breeding season insecticide application than in fields with no application reflected differences in the numbers of Diptera, which were nearly twice as abundant in fields with non-breeding season insecticide. There was no apparent link between Dipteran abundance and any crop management practice but there was a positive relationship with the presence of rivers and wet ditches next to fields with non-breeding season sprays.

Differences in the way foraging yellowhammers respond to insecticide applications between early and late nests was further evidence of an indirect effect of pesticides. Fields with breeding season applications of insecticide were used less by early broods, which have a wholly insectivorous diet, than fields without such applications. Later on, foraging visits to fields treated with insecticide in the breeding season were more frequent but, in many cases, birds were taking semi-ripe grain rather than invertebrates. This is consistent with insecticides influencing food supply, but not with direct sub-lethal mechanisms, which are unlikely change over the breeding season. Abundance, accessibility and distance of food resources are known to be significant factors in determining yellowhammer foraging patterns (Morris et al., 2001, 2002). Studies of other emberizid species have also found negative relationships between insecticide-induced reductions in invertebrate nestling-food abundance and foraging behaviour, including the corn bunting (*Miliaria calandra* L.) in the UK (Brickle et al., 2000) and the chestnut-collared longspur (*Calcarius ornatus* Townsend) in the USA (Martin et al., 2000).

There was no significant link between insecticide use and brood reduction from starvation, whereas Aebischer and Potts (1998) found that broad-spectrum use of insecticides on cereals over six summers in Sussex was associated with a reduction of about one-third in grey partridge chick-survival rate. Bradbury et al. (2003) noted that poor nestling condition in yellowhammers was not reflected in increased starvation rates. However, insecticides may still have an impact at the population level, as, in other species, nestlings leaving the nest with poor body condition are less likely to survive to reproductive age (Magrath, 1991).

Breeding-season pesticide applications were the most frequent and more important correlate, but

associations were also noted with repeated insecticide use over the crop year, although it is difficult to decouple timing from cumulative effects. Insecticide applied during the breeding season was most likely associated with variation in breeding-season food supply, behaviour and breeding performance, yet such correlations with timing have rarely been documented, and have considerable conservation implications. Indications of cumulative effects are also important, as some crops foraged in by yellowhammers, e.g. potatoes and peas, routinely receive two or more insecticide applications during the breeding season. Ewald and Aebischer (1999) and Brickle et al. (2000) have found that the numbers of insecticide applications affects the abundance of invertebrate nestling-food. The same correlation was found to affect corn bunting nestlings, by removing invertebrate food (Brickle et al., 2000), and further studies are desirable.

In cereal crops, insecticides are known to affect key invertebrate foods that are not usually considered ‘pests’ (e.g. Arachnida, Staphylinidae, and Hymenoptera), as well as reducing the abundance of target invertebrates (most commonly aphids), which form only a small component of yellowhammer diet (Wilson et al., 1996). Insecticides also affect invertebrate biomass and abundance of larger invertebrates that are selected by yellowhammers and other bunting species (Bösenberg, 1958; Evans et al., 1997).

In two cases, there were significant relationships between invertebrate abundance and the numbers of fungicide and herbicide applications. Herbicides are most likely to affect invertebrate numbers via the removal of host plants, as most of the herbicides concerned have no known insecticidal properties on key nestling foods (Whitehead, 2000). The negative effects of fungicides may be attributable to the insecticidal properties of some active substances (Sotherton and Moreby, 1988).

5. Conclusions

The results suggest that most negative relationships between insecticide use and nesting yellowhammers were associated with breeding season applications, but that cumulative effects of repeated use throughout the crop year may also be detrimental. With a decline in cereal grain prices from the mid-1990’s, an increase in

environmental awareness and the spread of cost-effective integrated farming techniques, most summer insecticide sprays now occur only when pests exceed thresholds values (Asby and Renwick, 2000). This is likely to reduce the indirect effects of insecticides on breeding birds. Although summer applications were a relatively small proportion of total insecticide use, this study suggests that effects were still tangible when sprays were applied during critical periods. The key period for the yellowhammer appears to be from territory settlement in March through to the end of June or early July, after which nestlings are no longer solely reliant on invertebrate food. However, for other bird species, which are later nesting, multi-brooded, or for which the seeding crop does not augment nestling diet, insecticide use may continue to be detrimental until the crops are harvested.

The provision of information and advice to agronomists and farmers on how to minimise the indirect effects of insecticides during the key summer period constitutes a first step towards benefiting breeding birds. In the UK, the production of Crop Protection Management Plans as part of the 'Voluntary Initiative' already offers strategies to reduce overall pesticide use and effectively target inputs. Financial assistance to reduce pesticide use is now available in the form of low-input options in agri-environment schemes in several countries.

In terms of specific measures, broad-spectrum insecticides should not be applied to crops during the breeding season, unless there are serious implications

for loss of yield. With mortality among non-target invertebrate food taxa often approaching 100%, seven days after spraying with broad-spectrum pyrethroids, and with substantial effects persisting for up to two months after the application (Sotherton, 1990), low persistence target-specific insecticides should be identified and adopted. If application of insecticides during the breeding season is deemed essential, mitigating measures should be sought. These may include the provision of alternative unsprayed foraging habitats (e.g. grass margins) that are now funded as part of agri-environment schemes in many countries. Although there are practical difficulties, it may be desirable for indirect effects to be considered as part of product risk assessment.

Acknowledgements

The authors thank DEFRA for funding this study and wish to acknowledge the kind co-operation of all landowners, farmers and spraying contractors, who provided access to their land and pesticide records. The other consortium partners on MAFF contract CTD9804 provided considerable assistance: Central Science Laboratory, Centre for Ecology & Hydrology and Game Conservancy Trust. In particular, Pete Robertson, Tim Milsom, Dave Parrot and John Allcock at CSL were very giving of their time. Mark Clook and two anonymous referees provided useful comments on earlier drafts of this manuscript.

Appendix A Active substances used on the study sites 1995–2001

Active substance	Total no. of sprays	No. of summer sprays	Cumulative % of study area sprayed	Rate (l or kg/ha)
Fungicides	610	583	224.0	
Azoxystrobin	61	61	26.4	0.48
Carbendazim	38	29	13.7	0.47
Carbendazim + chlorothalonil	6	6	2.0	0.50
Carbendazim + flusilazole	18	13	4.2	0.54
Carbendazim + flutriafol	10	10	3.3	0.88
Chlorothalonil	35	35	11.2	0.93
Chlorothalonil + flutriafol	16	16	5.0	1.62
Chlorothalonil + propiconazole	15	15	4.0	1.48

Appendix A

Active substance	Total no. of sprays	No. of summer sprays	Cumulative % of study area sprayed	Rate (l or kg/ha)
Cyproconazole	2	2	0.5	0.20
Cyproconazole + cyprodinil	1	1	0.4	0.80
Cyproconazole + prochloraz	20	20	5.4	0.87
Cyprodinil	17	17	8.9	0.38
Difenoconazole	7	2	2.7	0.51
Epoxiconazole	89	89	36.0	0.46
Epoxiconazole + fenpropimorph + kresoxim-methyl	13	13	6.1	0.60
Epoxiconazole + kresoxim-methyl	40	40	12.7	0.58
Fenbuconazole	3	3	0.8	0.04
Fenbuconazole + propiconazole	4	4	1.4	1.38
Fenbuconazole + tridemorph	2	2	0.5	2.00
Fenpropidin	44	44	15.3	0.31
Fenpropidin + fenpropimorph	20	20	5.4	0.36
Fenpropimorph	11	11	6.6	0.32
Fenpropimorph + fenpropidin	1	1	0.3	0.50
Fenpropimorph + flusilazole + tridemorph	6	6	2.7	0.90
Fenpropimorph + propiconazole	8	8	3.4	0.40
Flusilazole	12	8	5.4	0.28
Fluzilazol + carbendazim	3	0	1.1	0.33
Iprdione	1	1	0.5	0.57
Iprodione + thiophanate-methyl	7	7	2.6	2.37
Prochloraz	2	1	0.5	1.15
Prochloraz + fluquinconazole	8	8	2.8	1.00
Propiconazole + tebuconazole	4	4	1.1	0.28
Quinoxifen	16	16	6.6	0.11
Quintozene	4	4	1.1	0.50
Tebuconazole	40	40	14.4	0.46
Tebuconazole + triadimenol	16	16	4.0	0.50
Tridemorph + cyproconazole	2	2	0.5	?
Trifloxystrobin	4	4	2.5	0.90
Vinclozolin	4	4	1.7	0.50
Herbicides	824	385	286.9	
2,4-D	2	1	0.9	1.75
Amidosulfuron	19	19	6.2	0.03
Atrazine	20	20	7.7	2.09
Bentazone	2	2	0.9	1.00
Bifenox + chlorotoluron	2	0	0.5	9.40
Bromoxynil	13	13	4.0	1.10

Appendix A

Active substance	Total no. of sprays	No. of summer sprays	Cumulative % of study area sprayed	Rate (l or kg/ha)
Bromoxynil + clopyralid	9	9	2.9	1.00
Bromoxynil + ioxynil	2	2	0.5	1.00
Bromoxynil + prosulfuron	1	1	0.4	0.50
Carbetamide	2	0	1.5	0.22
Chlorotoluron	5	0	1.9	6.80
Clodinafop-propargyl	30	30	8.7	0.16
Clodinafop-propargyl + diflufenican	3	0	1.0	1.00
Clodinafop-propargyl + trifluralin	27	6	7.6	2.46
Clopyralid	2	2	1.1	0.75
Cyanazine	2	0	1.2	1.00
Cycloxydim	20	16	6.6	0.64
Dicamba + paclobutrazol	14	14	4.3	1.83
Diclofop-methyl + fenoxaprop-P-ethyl	8	6	3.7	2.09
Diflufenican + isoproturon	108	20	36.3	1.30
Diquat	7	2	1.4	1.67
Ethofumesate + phenmedipham	1	1	0.2	0.75
Fenoxaprop-ethyl	6	0	1.1	1.00
Fenoxaprop-P-ethyl	5	5	1.9	1.05
Fluazifop-P-butyl	15	2	4.4	0.44
Flupyr-sulfuron-methyl	11	3	3.7	0.02
Flupyr-sulfuron-methyl + carfentrazone-ethyl	2	0	0.5	0.0 g
Flupyr-sulfuron-methyl + metsulfuron-ethyl	6	2	2.0	0.0 g
Fluroxypyr	76	73	23.0	0.46
Fluroxypyr + bromoxynil + ioxynil	1	1	0.1	0.50
Glyphosate	107	45	43.1	2.36
Isoproturon	99	16	36.1	3.09
Isoproturon + diflufenican	5	0	1.4	0.74
Isoproturon + pendimethalin	25	0	8.6	4.00
Isoproturon + simazine	1	0	0.3	0.40
Mecoprop	5	3	1.0	1.79
Mecoprop-p	11	11	4.9	1.82
Metazachlor	15	2	4.3	1.49
Metazachlor + quinmerac	6	0	2.0	1.98
Metsulfuron-methyl	31	30	10.1	0.02
Metsulfuron-methyl + thifensulfuron-methyl	8	8	3.5	0.05
Pendimethalin	20	0	7.5	2.73
Propaquizafop	20	8	7.0	0.70
Propyzamide	3	0	0.7	0.14
Terbutylazine + terbutryn	1	1	0.5	2.50
Tralkoxydim	2	2	1.1	1.40
Tri-allate	32	0	13.9	1.88

Appendix A

Active substance	Total no. of sprays	No. of summer sprays	Cumulative % of study area sprayed	Rate (l or kg/ha)
Trifluralin	12	9	4.5	1.38
Insecticides	204	59	67	
Carbamate – pirimicarb	15	15	4.3	0.18
Organochlorine – gamma-HCH	11	8	3.6	1.30
Organophosphate – dimethoate	4	4	1.6	1.70
Pyrethroid – alpha-cypermethrin	5	5	1.5	0.10
Pyrethroid – cypermethrin	124	22	41.8	0.26
Pyrethroid – deltamethrin	1	1	0.8	0.20
Pyrethroid – esfenvalerate	4	0	0.9	0.15
Pyrethroid – lambda-cyhalothrin	12	4	5.2	0.14
Pyrethroid – permethrin	28	0	7.1	0.25

References

- Aebischer, N.J., 1991. Twenty years of monitoring invertebrates and weeds in cereal fields in Sussex. In: Firbank, L.G., Carter, N., Darbyshire, J.F., Potts, G.R. (Eds.), *The Ecology of Temperate Cereal Fields*. Blackwell Scientific Publications, Oxford, pp. 305–331.
- Aebischer, N.J., Potts, G.R., 1998. Spatial changes in Grey Partridge distribution in relation to 25 years of changing agriculture in Sussex, U.K.. *Gibier Faune Sauvage* 15, 293–308.
- Asby, C., Renwick, A., 2000. *Economics of Cereal Production 1998/99*. MAFF Special Studies in Agricultural Economics. Report No. 48. University of Cambridge.
- Benton, T.G., Bryant, D.M., Cole, L., Crick, H.Q.P., 2002. Linking agricultural practice to insect and bird populations: a historical study over three decades. *J. Appl. Ecol.* 39, 673–687.
- Bösenberg, K., 1958. Zur Nestlingsnahrung der Goldammer. *Falke* 5, 58–61.
- Bradbury, R.B., Wilson, J.D., Moorcroft, D., Morris, A.J., Perkins, A.J., 2003. Nestling condition and growth rates in four farmland passerines with contrasting UK population trends. *Ibis* 145, 295–306.
- Brickle, N.W., Harper, D.G.C., Aebischer, N.J., Cockayne, S.J., 2000. Effects of agricultural intensification on the breeding success of corn buntings *Miliaria calandra*. *J. Appl. Ecol.* 37, 742–755.
- Burn, A.J., 2000. Pesticides and their effects on lowland farmland birds. In: Aebischer, N.J., Evans, A.D., Grice, P.V., Vickery, J.A. (Eds.), *Ecology and Conservation of Lowland Farmland Birds*. British Ornithologists' Union, Tring, pp. 89–104.
- Campbell, L.H., Avery, M.I., Donald, P., Evans, A.D., Green, R.E., Wilson, J.D., 1997. A review of the indirect effects of pesticides on birds. JNCC Report No. 227. JNCC, Peterborough.
- Chamberlain, D.E., Fuller, R.J., Bunce, R.G.H., Duckworth, J.C., Shrubbs, M., 2000. Change in the abundance of farmland birds in relation to the timing of agricultural intensification in England and Wales. *J. Appl. Ecol.* 37, 771–788.
- Crawley, M.J., 1993. *GLIM for Ecologists*. Blackwell Scientific Publications, Oxford.
- Donald, P.F., Green, R.E., Heath, M.F., 2001. Agricultural intensification and the collapse of Europe's farmland bird populations. *Proc. Roy. Soc. Lond. Biol. Sci.* 268, 25–29.
- Donald, P.F., Pisano, G., Rayment, M.D., Pain, D.J., 2002. The common agricultural policy, EU enlargement and the conservation of Europe's farmland birds. *Agric. Ecosys. Environ.* 89, 167–182.
- Evans, A.D., Smith, K.W., Buckingham, D.L., Evans, J., 1997. Seasonal variation in breeding performance and nestling diet of ciril buntings *Emberiza cirilus* in England. *Bird Study* 44, 66–79.
- Ewald, J.A., Aebischer, N.J., 1999. Pesticide use, avian food resources and bird densities in Sussex. JNCC Report No. 296. JNCC, Peterborough.
- Fuller, R.J., 2000. Relationships between changes in lowland British agriculture and farmland bird populations: an overview. In: Aebischer, N.J., Evans, A.D., Grice, P.V., Vickery, J.A. (Eds.), *Ecology and Conservation of Lowland Farmland Birds*. British Ornithologists' Union, Tring, pp. 5–16.
- García-Berthou, E., 2001. On the misuse of residuals in ecology: testing regression residuals vs. the analysis of covariance. *J. Anim. Ecol.* 70, 708–711.

- Garthwaite, D.G., Thomas, M.R., 1999. Pesticide usage survey report 159: arable farm crops in Great Britain 1998. Ministry of Agriculture, Fisheries and Food, London.
- Gibbons, D.W., Reid, J.B., Chapman, R.A. (Eds.), 1993. The New Atlas of Breeding Birds in Britain and Ireland. Poyser, London, pp. 1988–1991.
- Gregory, R.D., Wilkinson, N.I., Noble, D.G., Robinson, J.A., Brown, A.F., Hughes, J., Procter, D., Gibbons, D.W., Galbraith, C.A., 2002. The population status of birds in the United Kingdom, Channel Islands and Isle of Man: an analysis of conservation concern 2002–2007. *British Birds* 95, 410–448.
- Kear, J., 1962. Food selection in finches with special reference to interspecific differences. *Proc. Zool. Soc.* 138, 163–204.
- Kyrkos, A., Wilson, J.D., Fuller, R.J., 1998. Farmland habitat change and abundance of yellowhammers *Emberiza citrinella*: an analysis of Common Birds Census data. *Bird Study* 45, 232–246.
- Magrath, R.D., 1991. Nestling weight and juvenile survival in blackbird *Turdus merula*. *J. Anim. Ecol.* 60, 335–351.
- Martin, P.A., Johnson, D.L., Forsyth, D.J., Hill, B.D., 2000. Effects of two grasshopper control insecticides on food resources and reproductive success of two species of grassland songbirds. *Environ. Toxicol. Chem.* 19, 2987–2996.
- McLaughlin, A., Mineau, P., 1995. The impact of agricultural practices on biodiversity. *Agric. Ecosys. Environ.* 55, 201–212.
- Milson, T.P., Langton, S.D., Parkin, W.K., Peel, S., Bishop, J.D., Hart, J.D., Moore, N.P., 2000. Habitat models of species' distribution: an aid to the management of coastal grazing marshes. *J. Appl. Ecol.* 37, 706–727.
- Morris, A.J., Whittingham, M.J., Bradbury, R.B., Wilson, J.D., Kyrkos, A., Buckingham, D.L., Evans, A.D., 2001. Foraging habitat selection by yellowhammers (*Emberiza citrinella*) in agriculturally contrasting regions in lowland England. *Biol. Conserv.* 98, 197–210.
- Morris, A.J., Bradbury, R.B., Wilson, J.D., 2002. Determinants of patch selection by yellowhammers *Emberiza citrinella* foraging in cereal crops. *Asp. Appl. Biol.* 67, 43–50.
- O'Connor, R.J., 1992. Proceedings of the BCPC Brighton Conference – Pests and Diseases, Indirect effects of pesticides on birds 1097–1104.
- O'Connor, R.J., Shrubbs, M., 1986. *Farming and Birds*. Cambridge University Press, Cambridge.
- Pain, D.J., Pienkowski, M.W., 1997. *Farming and birds in Europe. The Common Agricultural Policy and its Implications for Bird Conservation*, Academic Press, London.
- Potts, G.R., 1970. Studies of the changing role of weeds of the genus *Polygonum* in the diet of partridges. *J. Appl. Ecol.* 7, 567–576.
- Potts, G.R., 1986. *The Partridge: Pesticides, Predation and Conservation*. Collins, London.
- Prýs-Jones, R.P., 1977. Aspects of Reed Bunting ecology, with comparisons with the Yellowhammer. Unpublished Ph.D. Thesis, University of Oxford.
- Rands, M.R.W., 1985. Pesticide use on cereals and the survival of Grey Partridge chicks: a field experiment. *J. Appl. Ecol.* 22, 49–54.
- Shkedy, Y., Safriel, U.N., 1992. Nest predation and nestling growth rate of two lark species in the Negev Desert. *Israel. Ibis* 134, 268–272.
- Shrubbs, M., 2003. *Birds, Scythes and Combines*. Cambridge University Press, Cambridge.
- Sotherton, N.W., 1990. The effects of six insecticides used in UK cereal fields on sawfly larvae (Hymenoptera: Tenthredinidae). In: *Proceedings of the BCPC Brighton Conference on Pests and Diseases*. pp. 999–1004.
- Sotherton, N.W., Moreby, S.J., 1988. The effects of foliar fungicides on beneficial arthropods in wheat fields. *Entomophaga* 33, 87–99.
- Stoate, C., Moreby, S.J., Szczur, J., 1998. Breeding ecology of farmland yellowhammers *Emberiza citrinella*. *Bird Study* 45, 109–121.
- Thomas, M.R., Garthwaite, D.G., Banham, A.R., 1997. Pesticide usage survey report 141: arable farm crops in Great Britain 1996. Ministry of Agriculture, Fisheries and Food, London.
- Welham, S., 1993. *Procedure GLMM. Genstat 5 Procedure Library Manual*. NAG, Oxford.
- Whitehead, R. (Ed.), 2000. *The UK Pesticide Guide 2000*. CABI, Wallingford.
- Wilson, J.D., Arroyo, B.E., Clark, S.C., 1996. The diet of bird species of lowland farmland: a literature review. Unpublished report to the Department of the Environment and Joint Nature Conservation Committee. University of Oxford and Royal Society for the Protection of Birds, Sandy.
- Wilson, J.D., Morris, A.J., Arroyo, B.E., Clark, S.C., Bradbury, R.B., 1999. A review of the abundance and diversity of invertebrate and plant foods of granivorous birds in northern Europe in relation to agricultural change. *Agric. Ecosys. Environ.* 75, 13–30.