

ASSESSING THE INDIRECT EFFECTS
OF PESTICIDES ON BIRDS
PN0925
Final report

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Central Science Laboratory
Game Conservancy Trust
Royal Society for the Protection of Birds
Department of Zoology, University of Oxford

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Technical Appendices

1. A review of the indirect effects of pesticides on farmland birds. Helen McKay (*CSL*).
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3. The indirect effect of pesticides on yellowhammer breeding performance and demography. I: The relationship between yellowhammer *Emberiza citrinella* breeding performance, arthropod abundance and insecticide applications on arable farmland. J.D. Hart, T.P. Milsom, G. Fisher, V. Wilkins, S. Moreby, A.W.A. Murray & P.A. Robertson (*CSL*).
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5. Invertebrate monitoring within the large-scale field study. John M. Holland & Barbara Smith (*Game Conservancy Trust*).
6. Seed monitoring within the large-scale field study. John M. Holland & Barbara Smith (*Game Conservancy Trust*).
7. Population responses by granivorous birds to the manipulation of seed abundance on arable farmland during winter. T.P. Milsom, J.D. Hart, D. Parrott, J. Allcock, J.D. Bishop, R. Walls, & G.V. Watola. (*CSL*).
8. Predicting effects of summer insecticides on yellowhammer population growth rate. Joe Crocker & Justin Hart (*CSL*).

Executive summary

1. The aims of this study were to investigate the possible effects of pesticides on the food supply of farmland birds and their **indirect effects** on the demography of individual bird species, and to develop a framework for a risk assessment of the indirect effects of pesticides to aid the registration process.
2. There are three possible routes by which indirect effects of pesticides might arise. (i) Insecticides may deplete or eliminate arthropod food supplies, which are exploited by adults and their dependent young during the breeding season and, in so doing, reduce breeding productivity (Type 1 effect). (ii) Herbicides may reduce the abundance of, or eliminate, non-crop plants that are hosts for arthropods taken as food by farmland birds during the breeding season, and thereby reduce breeding productivity (Type 2 effect). (iii) Herbicides may also deplete or eliminate weed species, which provide either green matter or seeds for herbivorous and seed-eating species respectively, thereby reducing survival of those birds that rely on those food supplies (Type 3 effect).
3. As indirect effects of pesticides are defined by their impact on populations, the reduction in either breeding productivity or survival must be of sufficient magnitude to reduce the population growth rate of the bird species affected. Indirect effects may be additive because some farmland bird species may be vulnerable to more than one type. To result in a population decline, indirect effects acting either singly, or in combination, must reduce the population growth rate below threshold required to maintain a closed population.
4. A framework was developed for a risk assessment of Type 1 effects. It incorporated deterministic relationships between pesticide and abundance of arthropod prey, between arthropod prey abundance and chick growth rates, and between chick growth rates and the probabilities of brood reduction nests and of nests producing fledged young. This stage of the framework was structured to allow for multiple nesting attempts in each season. The predicted output of young from all nesting attempts per territory was then fitted to equations, which incorporate published estimates of survival, to estimate population growth rate. The predictions are deterministic and do not allow for stochastic or density dependent effects. Further methodological developments are required before equivalent frameworks for Type 2 and Type 3 effects are practicable.
5. A large-scale field experiment was carried out to assess the relative magnitude of Type 1 and Type 3 effects on farmland birds that feed arthropods to their young in summer and which are seed-eaters in winter. This was done by introducing perturbations to food supplies in summer and winter and observing the birds' responses. In winter, seed densities were increased, by providing additional seeds, whereas in summer, arthropod abundances were depressed by applying extra insecticide. These treatments and their controls were applied to four 1km blocks of arable farmland at each of three sites, according to a factorial design. Type 1 effects on breeding productivity were demonstrated for yellowhammer on all three sites. There was some evidence that the provision of supplementary seed resulted in increases over-wintering numbers of yellowhammers but the effect was not consistent across all sites. The findings for other farmland bird species were inconclusive.
6. A combination of field investigations and reviews of other studies identified Grey Partridge *Perdix perdix*, Corn Bunting *Miliaria calandra* and Yellowhammer *Emberiza citrinella* as being at risk from Type 1 indirect effects. Red-backed shrike *Lanius collurio*, Skylark *Alauda arvensis*, Tree Sparrow *Passer montanus*, and Yellow Wagtail *Motacilla flava flavissima* were possibly at risk. Linnet *Carduelis cannabina* and Turtle Dove *Streptopelia turtur* were possibly at risk from Type 3 effects. Type 2 effects have, as yet, only been demonstrated for Grey Partridge. A previous review, by the Joint Nature Conservation Committee (JNCC), had listed Barn Swallow *Hirundo rustica* and Lapwing *Vanellus*

vanellus as being possibly at risk from indirect effects but recent autecological studies suggest that this is unlikely. The status of five species, Bullfinch *Pyrrhula pyrrhula*, Song Thrush *Turdus philomelos*, Reed Bunting *Emberiza schoeniclus*, Blackbird *Turdus merula* and Starling *Sturnus vulgaris*, which had been identified by the JNCC review as being possibly at risk, is unclear due to a lack of relevant data.

7. As many of the prey species taken by farmland birds during the breeding season are non-target arthropods within the terms of the draft Guidance Document on Terrestrial Ecotoxicology Under Council Directive 91/414/EEC, the testing protocol described in the Guidance Document applies. However, as most farmland bird species and young take a wide range of arthropod taxa, it is insufficient to rely upon the proxy organisms used in the Standard Tests. It is necessary to obtain data on an appropriate range of arthropods, at an appropriate time of year, to calculate species-specific chick food indices from crops treated with the candidate insecticide, and for appropriate controls. These data can only be obtained under field test conditions, which form part of the Higher-tier tests. Provided that the requisite field data are available, the framework enables predictions to be made of the likely population effects of particular insecticide products. A worked example for yellowhammer is presented.
8. Potential mitigation and compensation measures are considered. Buffer zones, in which pesticide inputs are reduced, comprise the most promising mitigation measure in a regulatory context. In principle, buffer zones would be effective for boundary species, such as yellowhammer and tree sparrow, though wider buffer zones than those designed to protect boundary vegetation may be needed to ensure sufficient arthropod-rich feeding habitat. Buffer zones would not be appropriate for open field species, such as skylark and lapwing. An alternative to mitigation is to provide alternative food resources in non-cropped habitats to compensate for losses in cropped areas treated with pesticides. These may be situated in field margins, within the cropped area, or they may be whole field measures. The value of wild bird food seed mixes to bird species that occupy field margins has been demonstrated by other studies. However, further work is required to develop effective measures for open field species, such as skylark and lapwing.

Introduction

Aims and Objectives

The aims of this study were to investigate the possible effects of pesticides on the food supply of farmland birds and their indirect effects on the demography of individual bird species, and to develop a framework for a risk assessment of the indirect effects of pesticides to aid the registration process.

The project had four principal objectives:

1. To develop a causal framework for the assessment of indirect effects of pesticides on farmland birds. Also to identify those species most at risk from indirect effects and the mechanisms by which they may be affected (*Work undertaken by RSPB, University of Oxford, GCT, CSL*).
2. To conduct large-scale replicated field experiments to manipulate food resources available to farmland birds to demonstrate the magnitude of any indirect effects (*Work undertaken by GCT, CSL*).
3. To examine the current risk assessment based regulatory procedures in relation to indirect effects of pesticides (*Work undertaken by CSL*).
4. To propose suitable risk management practices that may reduce indirect effects of pesticides on birds (*Work undertaken by CSL*).

The Main Report is structured around these objectives. Additional information is presented in a series of Technical Appendices, which are cross-referenced with the text of the Main Report.

Background

Pesticides have the potential to cause declines in farmland bird populations by depleting their food supplies. This may occur because organisms taken as food by farmland birds are either invertebrate pests or injurious weeds that are deliberately targeted by pesticides, or because non-target organisms that are also food items for birds are eliminated incidentally by pesticide applications. As these processes do not involve **direct** poisoning of birds due to the toxicity of the active ingredients in pesticides or their metabolites, they have been termed **indirect** effects (e.g. Newton, 1995; Burn, 2000). There are three possible routes by which indirect effects of pesticides might arise:

1. Insecticides may deplete or eliminate arthropod food supplies, which are exploited by adults and their dependent young during the breeding season and, in so doing, reduce breeding productivity.
2. Herbicides may reduce the abundance of, or eliminate, non-crop plants that are hosts for arthropods taken as food by farmland birds during the breeding season and, thereby, reduce breeding productivity.
3. Herbicides may also deplete or eliminate weed species, which provide either green matter or seeds for herbivorous and seed-eating species respectively, thereby reducing survival of those birds that rely on those food supplies.

As indirect effects are defined by their impact on populations, the reduction in either breeding productivity or survival must be of sufficient magnitude to reduce the population growth rate of the bird species affected. Indirect effects may be additive because some farmland bird species may be vulnerable to more than one type. In theory, seed-eating passerines, which rely upon arthropod food supplies in summer to provision their young, could be vulnerable to all three types of indirect effect. To result in a population decline, indirect effects acting either singly, or in combination, must reduce the population growth rate below threshold required to maintain a closed population.

The data requirements to demonstrate any of the three types of indirect effects in the field are formidable. Prior to the current project, these requirements had only been met for the Grey Partridge *Perdix perdix*, a quarry species, which is in serious decline in Britain (Aebischer & Ewald, 2004). The diagnosis that indirect effects of pesticides were drivers of the declines in grey partridge populations in Britain was only possible because the demography and behavioural ecology of the grey partridge had been the subject of an intensive and long-term study by the Game Conservancy Trust and others (Potts, 1986; Aebischer & Ewald, 2004). An empirically based population model, derived from counts made over many years, identified an increase in chick mortality as the principal driver of the population decline in the grey partridge. Mortality of grey partridge chicks was shown to be greatest in the first ten days of life when the birds rely on the availability of specific arthropod prey. The dependence of grey partridge chicks on arthropod prey was determined by a combination of post-mortem investigations of the crops contents of chicks (Ford *et al.*, 1938), faecal analysis (Green *et al.*, 1987) and chick growth experiments under laboratory conditions (Cross, 1966, cited in Potts, 1986). Dietary preferences, especially for plant bugs *Heteroptera* and sawfly larvae *Dolerus* spp., were identified from chick feeding trials (Vickerman & O'Bryan, 1979). It was subsequently demonstrated that it was possible to predict chick survival in a given breeding season from the abundance of key arthropod groups (Potts & Aebischer, 1991). The link with insecticides was established by a comparison of winter wheat fields sprayed with the aphicide demeton-s-methyl with a sample of fields that were not sprayed. It illustrated the capacity of demeton-s-methyl to decrease the abundance of arthropods taken as food by partridge chicks, and the knock on effect on the chicks' survival – a Type 1 indirect effect (Moreby & Potts, 1985). Direct effects of aphicides through poisoning were ruled out in a separate trial (Saint-Andre de la Roche & Douville de Granssu, 1982, cited in Potts, 1986). Subsequently, a replicated experiment, which entailed a comparison of plots receiving herbicide and fungicide applications with others left unsprayed, demonstrated a Type 2 indirect effect (Rands, 1985, 1986).

The findings of the grey partridge study combined with the discovery of widespread population declines in farmland birds in Britain (Fuller *et al.*, 1995) prompted the Joint Nature Conservation Committee (JNCC) to commission a review of the evidence of indirect effects of pesticides on the demography of all farmland bird species (Campbell *et al.*, 1997). The review concluded that there was convincing evidence of declines in the populations of 22 farmland bird species, and of declines in the abundance of their food, some of which coincided with pesticide use (Table 7.1, in Campbell *et al.* 1997). The reviewers noted temporal correlations between increases in the extent of pesticide applications and periods of rapid decline in many farmland bird species. It was suggested that the declines of at least 11 farmland bird species, in addition to the Grey Partridge, may have been driven, at least in part, by the indirect effects of pesticides (Table 1). Finally, the review stressed the need for further research to assess the relative importance of indirect effects of pesticides and other factors associated with agricultural intensification as drivers of population declines in farmland birds.

The JNCC review was hampered by the lack of knowledge of the demography of those species that were perceived to be at risk: for most species, it was not known whether their population declines were due to decreases in breeding productivity, survival or both of these variables. Such information was fundamental to the determination of the routes by which indirect effects of pesticides might act, and for the implementation of remedial measures. A second major gap in knowledge was the lack of information on the responses of individual farmland bird species to variations in their food supplies. Though it was undeniable that declines in the abundance of food supplies had occurred, it was not known whether the declines had been severe enough to affect food intake rates and, thereby, body condition, survival and breeding productivity.

These gaps in knowledge were addressed by the current project in two ways. In the absence of demographic information on the relative importance of over-winter survival and breeding

productivity as drivers of population change, a large-scale field experiment (Objective 2) was undertaken to assess the relative importance of summer and winter food supplies to the population dynamics of farmland birds. Secondly, a meta study of past and concurrent autecological studies of farmland birds was undertaken to collate information on the relationships between the breeding productivity of individual farmland bird species, their arthropod food supplies, and the application of pesticides during the breeding season (Objective 1).

Since this project commenced in 1999, two important developments have occurred independently of the project but which have a significant bearing upon it. Analyses of the demography of farmland birds in Britain have begun to clarify the drivers of population change, while the decline of bird populations on farmland has also become a significant policy issue for Defra.

Table 1. Relationships between national population trends and demographic variables for farmland bird species in Britain that were provisionally identified as being at risk from the indirect effects of pesticides in the JNCC review (Campbell *et al.* 1997).

Species ¹	JNCC review ²	Breeding productivity ⁽³⁾	1 st year survival ⁽⁴⁾	Adult survival ⁽⁴⁾
Grey Partridge <i>Perdix perdix</i>	Y	+ ⁵	n.a. ⁵	n.a. ⁵
Tree Sparrow <i>Passer montanus</i>	P	-*	-	-
Turtle Dove <i>Streptopelia turtur</i>	P	-* ⁶	n.a.	n.a.
Bullfinch <i>Pyrrhula pyrrhula</i>	P	-	-	-
Song Thrush <i>Turdus philmelos</i>	P	n.a.	+ ⁷	- ⁷
Lapwing <i>Vanellus vanellus</i>	P	n.a.	(-) ⁸	(-) ⁸
Reed Bunting <i>Emberiza schoeniclus</i>	P	-	+	+
Skylark <i>Alauda arvensis</i>	P	-*	(+) ⁹	(+) ⁹
Linnet <i>Carduelis cannabina</i>	P	+	-	-
Barn Swallow <i>Hirundo rustica</i>	P	n.a.	-	-
Blackbird <i>Turdus merula</i>	P	n.a.	+	+
Starling <i>Sturnus vulgaris</i>	P	(-*)	(+)	(+)
Corn Bunting <i>Miliaria calandra</i>	(P)	-*	n.a.	n.a.
Spotted Flycatcher <i>Muscicapa striata</i>	(P)	-	+	?
Sand Martin <i>Riparia riparia</i>	(P)	n.a.	n.a.	n.a.
Mistle Thrush <i>Turdus viscivorus</i>	(P)	n.a.	-	+
Yellow Wagtail <i>Motacilla flava flavissima</i>	(P)	n.a.	n.a.	n.a.
Dunnock <i>Prunella modularis</i>	(P)	n.a.	-	-
Yellowhammer <i>Emberiza citrinella</i>	(P)	-*	(+)	(+)
Red-backed Shrike <i>Lanius collurio</i>	(P)	n.a.	n.a.	n.a.

Notes:

1. Species order as given in Table 7.1 of JNCC review (Campbell *et al.*, 1997).

2. : Quality of evidence for indirect effects (as given by Campbell *et al.*, 1997): Y = definite effect; P = possible effect; (P) qualified possible effect.

3. **Breeding productivity:** Measured from fledgling production per breeding attempt. Information source: Siriwardena *et al.* (2000), Siriwardena & Robinson (2002). + trend in breeding productivity matched population trend; - trend in breeding productivity did not match population decline, * breeding productivity increased over the period of population decline. n.a. = not analysed

4. **Survival:** Information sources: Siriwardena *et al.* (1998, 1999), Peach *et al.* (1999), Freeman & Crick (2003), Robinson *et al.* (2004). + variation in survival between years tracked population change or varied consistently between periods of population increase, stability and decrease; - variation in survival between years did not track population change or nor did it vary consistently between periods of population increase, stability and decrease;. n.a. = not analysed.

5. **Grey Partridge.** Inter-annual change in CBC counts positively correlated with grey partridge chick survival index (Aebischer & Ewald, 2004). Game Conservancy Trust Grey Partridge study ruled out over-winter survival as a significant driver of population change (Potts, 1986).

6. **Turtle Dove.** Yet Browne & Aebischer (2004) discovered that a reduction in the number of breeding attempts per season had occurred and that this was sufficient to have caused a population decline.

7. **Song Thrush.** Population multiplication rate between years was also significantly correlated with post-fledging survival (Robinson *et al.*, 2004).

8. **Lapwing.** No formal correlation analyses were carried out. Nonetheless, adult survival increased during the period of population decline; 1st winter survival fluctuated between years but did not show any long-term trend (Peach *et al.*, 1994).

9. **Skylark:** the only dataset describing temporal variation in the survival of skylarks is derived from a local study in Lancashire (Wolfendon & Peach, 2001).

The respective roles of survival and breeding productivity as drivers of population changes in seed-eating bird species have been provisionally determined from a series of analyses of long-term datasets amassed by the British Trust for Ornithology on annual breeding populations (Common Bird Census), breeding productivity (Nest Record Scheme) and survival (Ringing Scheme)(Table 1). As the analyses were correlative, diagnosis of cause and effect was subject to caveats. Agreement between trends in a particular demographic variable and population size was interpreted to mean that variation in the former was sufficient to have accounted for variation in the latter. Conversely, if the association between a particular demographic variable and trends in population size was not as predicted, or there was no association, that variable was rejected as a potential driver of population change.

The population declines of most seed eating species were not associated with decreases in breeding success (Table 1); the only exception was linnet *Carduelis cannabina* (Siriwardena *et al.*, 2000). In contrast, variation in survival rates between years appeared to be sufficient to explain the population trends in reed bunting (Peach *et al.*, 1999) and, according to Newton (2004) tree sparrow *Passer montanus* and yellowhammer *Emberiza citrinella* as well. Inter-annual variation in survival is probably the principal driver of population changes in two other seed-eaters which were not identified as being vulnerable to indirect effects of pesticides: goldfinch *Carduelis carduelis*, and house sparrow *Passer domesticus* (Siriwardena *et al.*, 1999). These analyses imply that if pesticides are impacting on the demography of seed-eating birds through their food supplies, it is more likely to occur during the non-breeding than the breeding season. The effects of herbicides on weed seed production may, therefore, be a key issue. Nonetheless, the results of the analyses should be interpreted with caution. The analyses of breeding success are restricted to breeding output per attempt and do not necessarily reflect the more significant measure of annual breeding output per pair. This is a particular problem for multi-brooded species, such as yellowhammer and skylark. Recent intensive autecological studies of corn bunting *Miliaria calandra* (Brickle *et al.*, 2000), yellowhammer (Bradbury *et al.*, 2000; CSL study) and turtle dove *Streptopelia turtur* (Browne & Aebischer, 2004), appear to contradict the findings of the demographic analyses. Siriwardena *et al.* (2000) interpreted their results to mean that the recent population declines in all three species had not been driven by a decline in breeding success, yet the autecological studies of corn bunting and yellowhammer have shown that that local breeding populations from the core ranges of both species probably do not produce sufficient young annually to be self-supporting. In the case of turtle dove, Browne & Aebischer (2004) discovered that a reduction in the number of breeding attempts per season had occurred and considered that this was sufficient to have caused a population decline. This component of breeding output would have been invisible in the Nest Record Scheme dataset.

The policy implications of the indirect effects of pesticides on farmland birds were broadened in 2000, when Defra committed itself to the following Public Service Agreement (PSA) target:

‘to care for our living heritage and preserve natural diversity by reversing the long term decline in the number of farmland birds in England by 2020, as measured annually against underlying trends’.

The Farmland Bird Index (FaBI) (Table 2) is a measure of progress towards the PSA target, and is one of a number of ‘headline indicators’ of the sustainable food and farming strategy of Defra (Gregory *et al.*, 2002a). Of the thirteen declining species in the index, it is noteworthy that eight were identified by the JNCC review as being potentially at risk to the indirect effects of pesticides (Campbell *et al.*, 1997).

Table 2. The constituent species of the Farmland Bird Index (FaBI), their long-term population trends in Britain, conservation status, and susceptibility to indirect effects of pesticides.

Species	Population trends 1970-1999 % change (1)	Conservation status: BAP & SOCC (2)	Indirect effects of pesticides (3)
Barn Owl <i>Tyto alba</i>	No data	-, A	N
Corn Bunting <i>Miliaria calandra</i>	-88	B, R	?
Goldfinch <i>Carduelis carduelis</i>	+23	-, G	N
Greenfinch <i>Carduelis chloris</i>	+13	-, G	N
Grey Partridge <i>Perdix perdix</i>	-86	B, R	Y
Jackdaw <i>Corvus monedula</i>	+88	-, G	?
Kestrel <i>Falco tinnunculus</i>	-14	-, A	?
Lapwing <i>Vanellus vanellus</i>	-40	-, A	P
Linnet <i>Carduelis cannabina</i>	-52	B, R	P
Reed Bunting <i>Emberiza schoeniclus</i>	-53	B, R	P
Rook <i>Corvus frugilegus</i>	+40	-, G	?
Skylark <i>Alauda arvensis</i>	-52	B, R	P
Starling <i>Sturnus vulgaris</i>	-71	-, R	P
Stock Dove <i>Columba oenas</i>	+127	-, A	N
Tree Sparrow <i>Passer montanus</i>	-95	B, R	P
Turtle Dove <i>Streptopelia turtur</i>	-71	B, R	P
Whitethroat <i>Sylvia communis</i>	-16	-, G	?
Woodpigeon <i>Columba palumbus</i>	+90	-, G	N
Yellowhammer <i>Emberiza citrinella</i>	-53	-, R	?
Yellow Wagtail <i>Motacilla flava flavissima</i>	-40	-, A	?

Notes:

1. Population trends: Gregory *et al.* (2002a).

2. Conservation status: BAP = Biodiversity Action Plan status (Anon, 1995; UK Biodiversity Group, 1998): B = Priority species status, - = no BAP. SOCC = Species of Conservation Concern (Gregory *et al.*, 2002b): R = Red list, A = Amber list, G = Green list;

3. Evidence of indirect effects: (Campbell *et al.*, 1997): Y=yes, P = possible effect, N=no; ? = equivocal evidence or not investigated

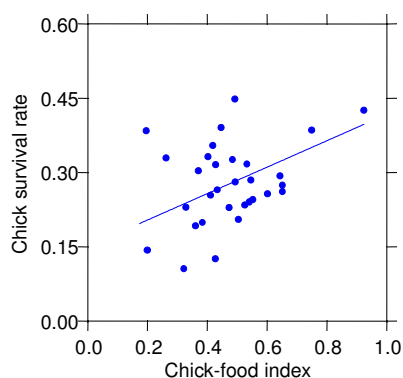
Objective 1.

Develop a causal framework for the assessment of indirect effects of pesticides on farmland birds.

As a minimum, a causal framework for the assessment of indirect effects of pesticides on birds requires links between a measure of food abundance and a key demographic variable, and between the same demographic variable and population growth rates. The relationship between food abundance and the demographic variable should be causal, and be based upon detailed autecological and behavioural studies of the farmland bird species concerned. The food abundance measure should also be in a form that allows comparison between plots or fields treated with a particular pesticide and those left as unsprayed controls. In principle, this simple framework is appropriate for the assessment of the effects of insecticides, herbicides or fungicides.

Two interlinked regression models from the grey partridge study provide a working example of what might be possible for other farmland bird species. In the first model, the survival rate of grey partridge chicks in a given year is regressed on an index of chick food abundance for the same year (Fig. 1, L.H. graph). The graph is an updated version of that shown in Potts & Aebischer (1991), which gives full details of the data sources. In the second model, the change in breeding density between Year t and Year $t+1$ is regressed on the chick survival rate for Year t (Fig. 1, R.H. graph). As chick survival is presented on the same scale in both regression models, it is possible to derive predictions of population change between one year and the next from initial data on chick food abundance. The approach offers scope for making simple deterministic predictions of population effects arising from pesticide treatments, provided that the requisite data on chick food abundance are available for treated and control plots.

Annual chick survival rate v. annual chick-food index
Sussex 1970 - 2000



Year-on-year change in spring pair density (CBC data) v.
chick survival (Partridge Count Scheme): 1962 - 1993

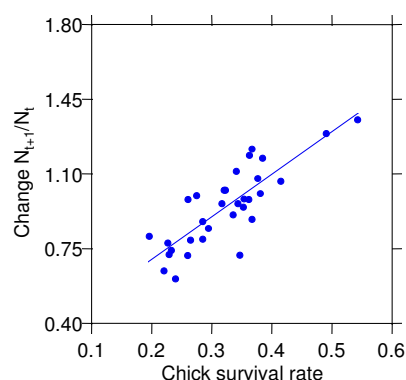


Fig.1. An example of food-resource based assessment framework for the Grey Partridge *Perdix perdix*. (Source: Robertson, Aebischer & Holland, *unpublished*).

A limitation of the method is that it does not allow for variability. Yet there is considerable evidence of variability in the response of grey partridge chick survival to a given level of food abundance (Fig. 1, LH Graph). In the regression of chick survival rate on chick food, the high chick food values lie very close to the fitted line but the residual errors of the low values of the chick-food index tend to be much greater. Thus, survival is good in years when the chick-food index is high but is much more variable when food abundance is low. Probabilistic risk assessment methods can be used to accommodate variability and these are considered under Objective 3 and in Technical Appendix 8.

At first sight, the grey partridge framework appears to be an exemplar for other farmland species but, for a variety of reasons, it cannot be simply duplicated. In the first stage of the grey partridge framework, the effects of food supply on the development and survival of young are reduced to a simple bivariate relationship (Fig. 1. LH graph). It conceals several causal links that may need to be treated separately when the framework is applied to other species. In the case of the yellowhammer, it will be demonstrated that it is necessary to expand the first stage of the framework to include links between food abundance and chick growth rates, between chick growth rates and breeding productivity.

The method used to estimate the survival of young may also have to be adapted to the breeding biology of the species concerned. As the grey partridge is single brooded and produces precocial young, a count of broods when the chicks are c. 6 weeks old is sufficient to generate an estimate of annual breeding productivity. However, as farmland passerines produce altricial young, which they rear to fledging in the nest, the number of fledged young per nesting attempt is a more practical measure of breeding productivity, particularly as the fledglings of many passerine species become extremely difficult to monitor after leaving the nest. Moreover, as most farmland passerines are multi-brooded, information on the effects of food abundance on the outcome of each breeding attempt must be integrated over all attempts made during the breeding season.

Finally, the sampling methods used to derive chick-food indices for particular farmland species need to reflect the breeding biology of those species. Unlike grey partridge, the adults of farmland passerines become central place foragers (Stephens & Krebs, 1986) while provisioning dependent young in the nest and are confined to relatively small foraging ranges around their nests. To obtain representative samples of arthropods that are available to chicks in the nest, it is necessary to sample these foraging ranges, as Brickle *et al.* (2000) have done for the corn bunting, and to synchronise the sampling to the nestling period.

Turning now to the second stage of the model framework for the grey partridge, it has been possible to derive an empirically based regression equation that describes the relationship between chick survival in a given year and population changes in subsequent years (Fig.1, RH graph). Unfortunately, this elegant model solution is not directly transferable to other farmland bird species. An analysis of yellowhammer breeding data supplied by the BTO was undertaken to ascertain whether it was practicable to duplicate the grey partridge regression model and the result is shown in Fig.2. The regression of the change in yellowhammer numbers between successive breeding seasons on mean brood size at fledging is significant but the relationship is weak and its slope counter-intuitive. For this species, it will be shown that it is necessary to use survival estimates derived from the BTO ringing scheme (e.g. Siriwardena *et al.*, 2000; Bradbury *et al.*, 2000) to obtain predictions of population growth rates in subsequent years. The predictions are derived from average survival values and no allowance is made for density dependence between breeding productivity and subsequent survival rates.

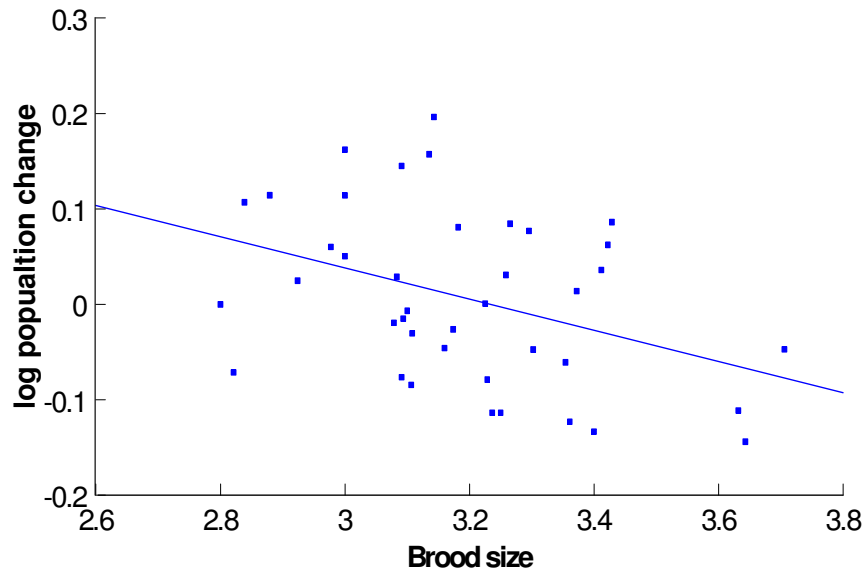


Fig.2. Breeding productivity per nesting attempt of yellowhammers as a predictor of year-to-year changes in breeding population ($r=-0.39$, $n=40$, $P<0.05$). The data are for selected counties in lowland England where cereals are the dominant crop. Mean brood size data are from the BTO Nest Record Scheme; population change data are from BTO Common Bird Census.

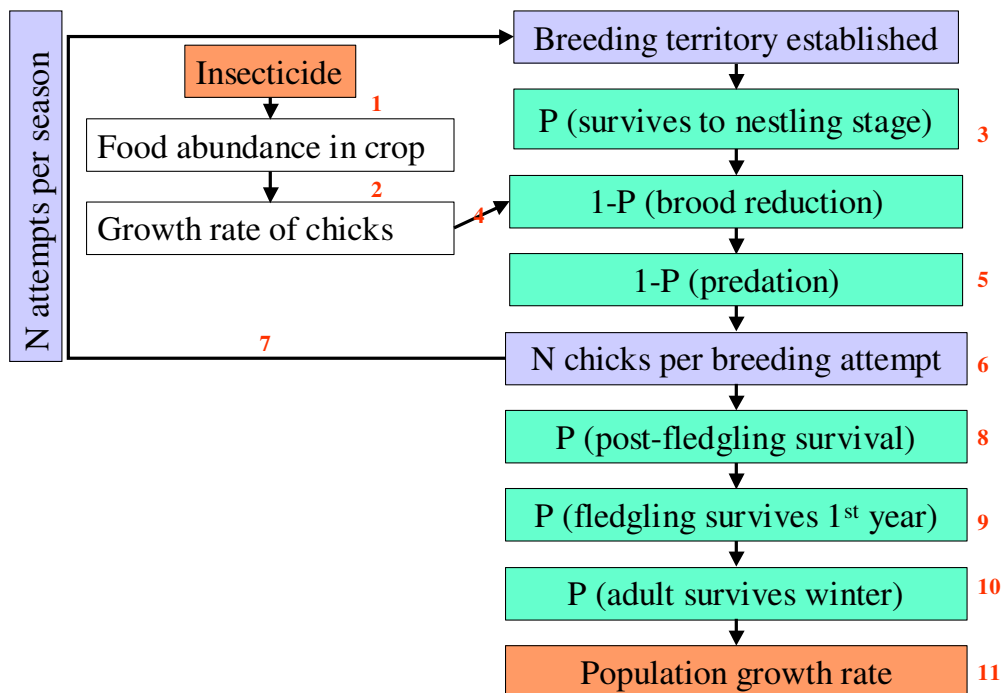


Fig. 3. A causal framework for the assessment of indirect effects of insecticides on the breeding productivity and population growth rates of yellowhammers. Numbers cross-refer with explanatory text in Technical Appendix 8.

A framework developed for yellowhammer is shown in Fig. 3. It incorporates stages that describe deterministic relationships between pesticide and abundance of arthropod prey, between arthropod prey abundance and chick growth rates, and between chick growth rates and the probabilities of nests fledging and of brood reduction. The equations describing the relationships between food and breeding productivity are derived from comparisons between nests, in contrast to those in the grey partridge framework, which were derived from comparison between annual averages of brood survival and food abundance across years (Fig. 1). The framework is structured to allow for

multiple nesting attempts in each season. The predicted output of young from all nesting attempts per territory is then fitted to equations that incorporate estimates of survival, extracted from the scientific literature, to estimate population growth rate between one year and the next. The framework is sufficiently generic to be applicable to all farmland passerines in the breeding season. A fully parameterised framework for yellowhammer is presented under Objective 3 of this report.

In comparison with the Type 1 indirect effect, the problems of modelling the indirect effects of herbicides on seed supplies and the survival of granivorous farmland birds during the non-breeding season (Type 3 effect) are far less tractable. At present, there is no ready solution because of the difficulties of relating herbicide inputs to seed production, and of relating over-winter survival of the birds to seed abundance. The development of methods to model the relationship between herbicides and seed production is covered in a concurrent PSD-funded project on wider biodiversity (Boatman *et al.*, 2004a) and will not be considered any further in this report. However, some comment is necessary on modelling methods to predict population effects from the responses of individual birds to changes in their food supplies. Recent developments of modelling theory have outstripped the collection of the field data required to parameterise them (Stephens *et al.*, 2003). In principle, depletion models based on the food requirements and foraging behaviour of individuals may be effective at predicting foraging patch use from the quantity of food available. To obtain an estimate of survival rates, it is necessary to integrate information from individual patches across all foraging patches used between breeding seasons. However, the models may incorporate major and probably unrealistic assumptions if the number, distribution and quality of the patches available to the birds are not known. The models may work well for over-wintering populations that remain within discrete and well studied areas during the non-breeding season, e.g. oystercatchers *Haematopus ostralegus* on a small estuary (Goss-Custard *et al.*, 1982, 1995), but are less satisfactory when the birds are mobile, and information on the number, distribution and quality of foraging patches is imperfect. Many farmland bird species are not only mobile during the non-breeding season but the average sizes of their home ranges (& hence the number and distribution of potential foraging patches) have been estimated in only a few cases (Calladine *et al.*, 2003; Haynes *et al.*, 2003).

Identify those species most at risk from indirect effects and the mechanisms by which they may be affected.

The meta-study of projects undertaken by RSPB and Oxford University (Technical Appendix 2) identified seven species, in addition to grey partridge, that are most probably at risk from indirect effects of pesticides (Table 3). Turtle dove and linnet were considered to be vulnerable to Type 3 indirect effects, whereas yellow wagtail *Motacilla flava flavissima*, red-backed shrike *Lanius collurio*, tree sparrow, corn bunting and yellowhammer were identified as being at risk from Type 1 effects. Autecological studies of barn swallow *Hirundo rustica*, house sparrow and lapwing *Vanellus vanellus* did not provide evidence of indirect effects from either insecticides or herbicides.

The quality of evidence for the mechanisms varied between species (Table 4). Most of the key linkages in the causal chain between insecticide and population growth rate were established for corn bunting and yellowhammer, and some have been verified by experiment.

In a combined analysis of data from eight arable farms, Morris (Technical Appendix 2) demonstrated four key relationships (Fig. 4): (1) timing of treatment of crops with insecticides and the abundance of arthropod prey taken by yellowhammers; (2) an avoidance of fields treated with insecticide by adult yellowhammers provisioning their young up until late June when alternative food supplies became available (ripening cereal grain); (3) frequency of insecticide applications and the condition of yellowhammer broods, and (4), a correlation between brood reduction and the

abundance of arthropod prey in adjacent crops (Fig. 4). Similarly, for corn bunting, Brickle *et al.* (2000) found that brood condition, and the probability of nest survival, were both correlated with the abundance of insect food close to the nest. The abundance of chick-food was negatively correlated with the number of insecticide applications (further details are given in Technical Appendix 1).

The evidence of Type 1 effects on skylark is based on a regression of brood condition on skylark chick food (Boatman *et al.*, 2004b), whereas that for red-backed shrike and tree sparrow was anecdotal (Technical Appendix 2).

There is some evidence that population dynamics of linnet and turtledove have been influenced by Type 3 indirect effects.

The decline in the linnet population coincided with reductions in the abundance of key weed species that provided the bulk of seeds for chicks, and preceded a dietary switch by linnets to the unripe seeds of oilseed rape and dandelions. The key weed species are known to be vulnerable to herbicides. Evidence for a demographic response by linnets to changes or reductions in the availability of seeds for chicks is circumstantial: linnet populations breeding in areas where oil seed rape and dandelions are scarce settle at low densities, breed later, suffer high levels of brood reduction and of complete brood starvation. Further details are given in Technical Appendix 2.

Table 3. List of farmland bird species for which there is currently evidence of indirect effects of pesticides

Species	Type 1 effect - insecticides on chick food	Type 2 effect - herbicides on arthropods	Type 3 effect - herbicides on seeds
Corn Bunting <i>Miliaria calandra</i>	+		
Grey Partridge <i>Perdix perdix</i>	+	+	
Linnet <i>Carduelis cannabina</i>			(+)
Red-backed Shrike <i>Lanius collurio</i>	(+)		
Skylark <i>Alauda arvensis</i>	(+) ¹		
Tree Sparrow <i>Passer montanus</i>	(+)		
Turtle Dove <i>Streptopelia turtur</i>			(+)
Yellowhammer <i>Emberiza citrinella</i>	+		
Yellow Wagtail <i>Motacilla flava flavissima</i>	(+)		

Notes: 1 – based on study not covered by the meta study in Appendix 2 but which is reported in Boatman *et al.* (2004b).

Table 4. Summary of state of knowledge of the impact of insecticides on the breeding behaviour, breeding productivity and demography of farmland bird species.

Species	Effect on food	Effect on foraging behaviour	Effect on chick condition/ growth	Effect on brood size	Effect on chick survival	Effect on population growth rate
Corn Bunting <i>Miliaria calandra</i>	2	2	2		2	
Grey Partridge <i>Perdix perdix</i>	1			1	1	1
Red-backed Shrike <i>Lanius collurio</i>			3		3	
Skylark <i>Alauda arvensis</i>			3		X	
Tree Sparrow <i>Passer montanus</i>	3		?			
Yellowhammer <i>Emberiza citrinella</i>	1	2	2		1	1

Notes:

1 = Significant effect demonstrated by field experiment;

2 = Significant and consistent effect demonstrated by autecological studies;

3 = some effects apparent, but not consistently significant;

X = No significant effect demonstrated by autecological studies.

? inconclusive evidence.

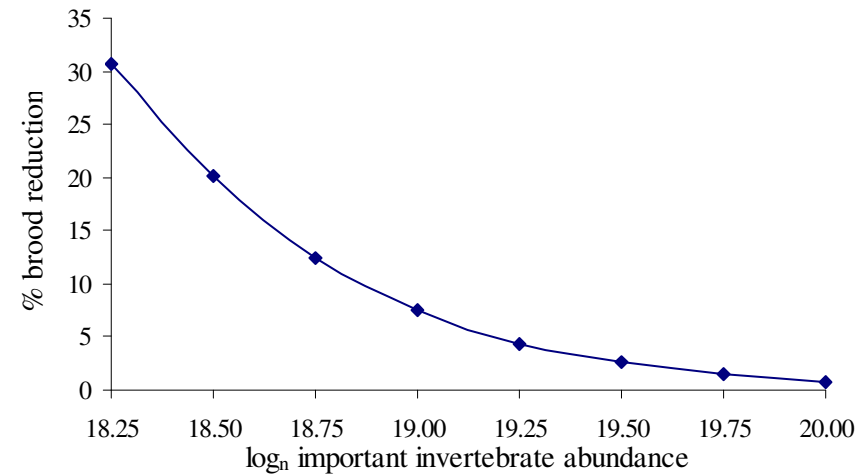
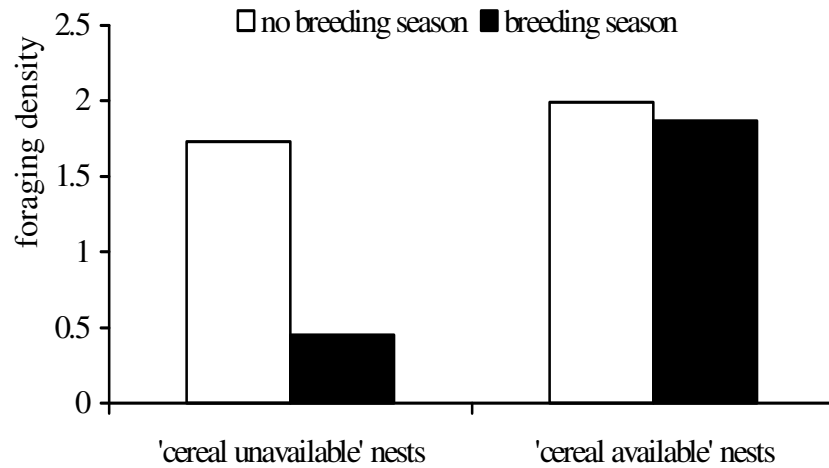
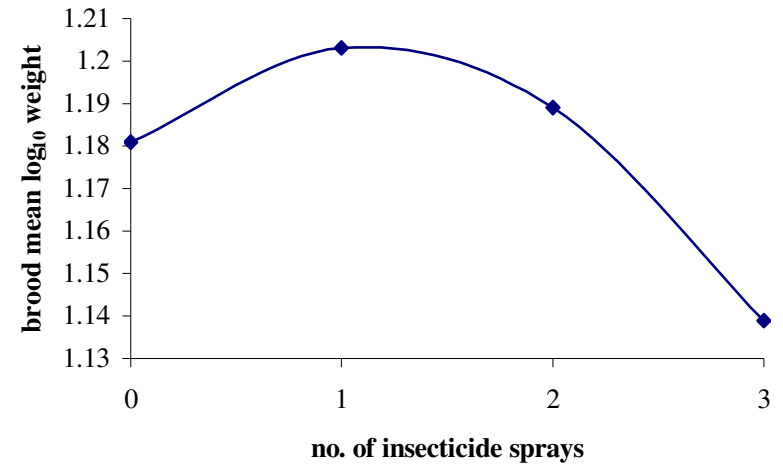
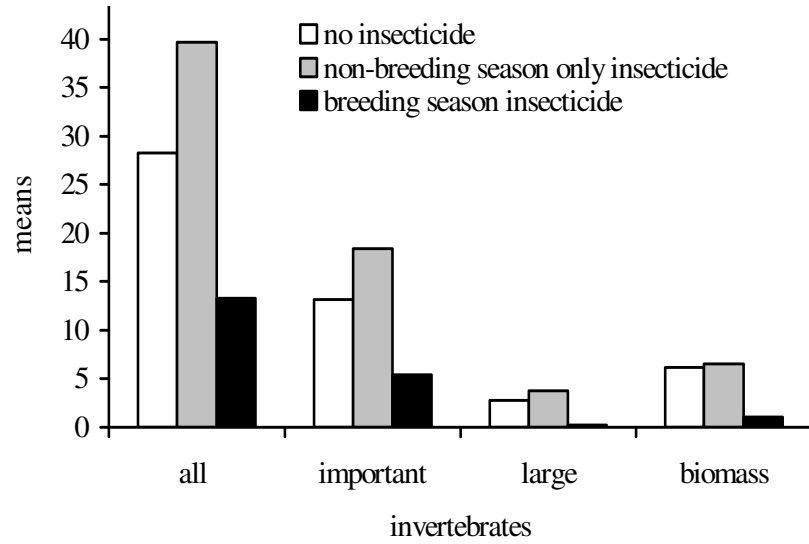


Fig. 4. Relationships between insecticides and arthropod prey (Top Left), insecticides and patch selection by foraging yellowhammers (Bottom Left), frequency of insecticide applications and yellowhammer brood condition (Top Right), and between incidence of brood reduction and abundance of key arthropod taxa. Sources (Morris, Technical Appendix 2). See Figs. 6, 9, 10, 12 and accompanying text in Technical Appendix 2 for further details.

Turtle Doves also feed extensively on weed seed supplies during the breeding season (Browne & Aebischer, 2003) and may ultimately rely upon them. Several of the weed species that feature extensively in the turtle dove diet (e.g. chickweed *Stellaria media*) have undergone widespread declines on farmland in Britain in recent years (Firbank & Smart, 2002), and are susceptible to the frequency of herbicide use (Ewald & Aebischer, 1999). It is probable that the reduction in the number of breeding attempts made each year may be responsible for the decline in turtle dove numbers (Browne & Aebischer, 2004). The shortening of the breeding season may be the result of food shortages later in the summer but further work will be required to confirm this.

The more complex herbicide pathway involving linkages between herbicides, invertebrate host plants, arthropod abundance, and survival of birds (Type 2 effect) has been established only for the grey partridge (Potts, 1986; Potts & Aebischer, 1991).

The current list of species considered to be at risk from the indirect effects of pesticides differs significantly from that compiled by the JNCC review (Campbell *et al.*, 1997). Indirect effects on two species in the ‘possible’ category on JNCC list, lapwing and barn swallow, have been provisionally ruled out, whereas they have been demonstrated for two species on the JNCC ‘qualified possible’ list, corn bunting and yellowhammer (Table 5). Finally, the position of nine species on the JNCC list, five in the ‘possible’ category and four in the ‘qualified possible’ category, is unclear because the requisite data are still lacking.

Table 5. Comparison of lists compiled by the JNCC review and the present study of species regarded as being at risk from the indirect effects of pesticides.

Species ¹	JNCC review ²	This study ³
Grey Partridge <i>Perdix perdix</i>	Y	Y
Tree Sparrow <i>Passer montanus</i>	P	(P)
Turtle Dove <i>Streptopelia turtur</i>	P	(P)
Bullfinch <i>Pyrrhula pyrrhula</i>	P	n.d.
Song Thrush <i>Turdus philmelos</i>	P	n.d.
Lapwing <i>Vanellus vanellus</i>	P	(N)
Reed Bunting <i>Emberiza schoeniclus</i>	P	n.d.
Skylark <i>Alauda arvensis</i>	P	P
Linnet <i>Carduelis cannabina</i>	P	(P)
Barn Swallow <i>Hirundo rustica</i>	P	(N)
Blackbird <i>Turdus merula</i>	P	n.d.
Starling <i>Sturnus vulgaris</i>	P	n.d.
Corn Bunting <i>Miliaria calandra</i>	(P)	Y
Spotted Flycatcher <i>Muscicapa striata</i>	(P)	n.d.
Sand Martin <i>Riparia riparia</i>	(P)	n.d.
Mistle Thrush <i>Turdus viscivorus</i>	(P)	n.d.
Yellow Wagtail <i>Motacilla flava flavissima</i>	(P)	(P)
Dunnock <i>Prunella modularis</i>	(P)	n.d.
Yellowhammer <i>Emberiza citrinella</i>	(P)	Y
Red-backed Shrike <i>Lanius collurio</i>	(P)	(P)

Notes:

1. Species order as given in Table 7.1 of JNCC review (Campbell *et al.*, 1997).

2: Quality of evidence for indirect effects (as given by Campbell *et al.*, 1997): Y = definite effect; P = possible effect; (P) qualified possible effect; - not on list.

3: Quality of evidence for indirect effects: Y = definite effect; P = possible effect; (P) circumstantial evidence for effect, (N) no evidence of effect from recent autecological studies; N = no effect, n.d. = no relevant data.

The greatest advances in our understanding of the indirect effects of pesticides have been made during the breeding season, yet the demographic analyses of Siriwardena *et al.* (1998, 1999, 2000) and Peach *et al.* (1999) highlighted importance of the decreases in survival amongst seed-eating birds, which may have arisen through food shortages during the non-breeding season. There is no doubt that the abundance of seeds from weeds has declined on arable farmland in Britain. Robinson & Sutherland (2002) estimated a c.90% decline in seed densities between 1900 and 2000, in the top 1cm of soils. This is the component of the seed bank in arable fields that would have been accessible to farmland birds. It is likely that an increase in the use of herbicides contributed to this decline (Newton, 1995; Ewald & Aebischer, 1999; Technical Appendix 1). However, the links between inputs of individual herbicides, seed densities in the seed bank, and the survival of seed-eating bird species have not been quantified partly because of methodological problems highlighted in a previous section of this report.

Objective 2.

Conduct large-scale replicated field experiments to manipulate food resources available to farmland birds to demonstrate the magnitude of any indirect effects

Aims of experiment

The experiment was designed to evaluate the relative importance of summer and winter food supplies on the demography of farmland birds that take arthropod prey during the breeding season but which are primarily seed-eaters at other times of year. This was done by introducing perturbations to the food supplies in summer and winter and observing the responses of farmland birds in the experimental areas.

Experimental design

Food supplies were manipulated by providing supplementary seed during winter, and by applying additional insecticides to depress the abundance of arthropod prey during summer. These treatments and their controls were assigned to blocks of arable farmland, each c. 1km² in extent, according to a factorial design (Fig. 5), which was replicated over three farms.

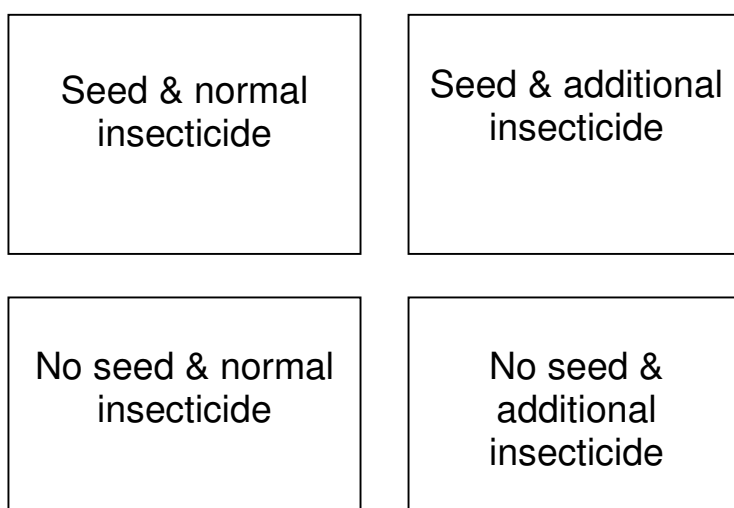


Fig.5. Factorial allocation of supplementary seed and increased insecticide treatments to the four 1 km² blocks on each farm.

The farms were situated in Hampshire, Lincolnshire and North Yorkshire respectively. As the entry of each farm to the experiment was staggered, the seed treatments were applied at the Hampshire, Lincoln and North Yorkshire sites during the winters of 1999/2000, 2000/1 and 2001/2 respectively (Technical Appendix 7). The supplementary insecticides were applied in the following summers in the second year experiment.

Methods

Insecticide applications

Additional insecticides were applied in summer, by agreement with the landowners, in two of the four blocks on each farm. The supplementary applications were timed to coincide with the nestling phase of the yellowhammer to achieve maximum effect. In most cases, pyrethroid products were used (cypermethrin, deltamethrin, lambda cyhalothrin) but an organophosphate (chlorpyrifos) and a carbamate (pirimicarb) were applied to some crops on two farms.

Table 6. Two-way analysis of variance of the mean number of insecticide applications per field per block in relation to insecticide treatment (TREAT) and farm (FARM). Dependent variable = mean n sprays per field per block.

Source	Type III Sum of Squares	d.f.	Mean Square	F	Sig.
Corrected model	8.959 ⁽¹⁾	5	1.792	47.792	<0.001
Intercept	13.632	1	13.632	363.601	<0.001
FARM	1.527	2	0.763	20.358	0.002
TREAT	5.267	1	5.267	140.481	<0.001
FARM*TREAT	2.166	2	1.083	28.880	0.001
Error	0.225	6	3.749E-02		
Total	22.816	12			
Corrected Total	9.184	11			

Notes:

(1) $R^2 = 0.976$ (Adjusted $R^2 = 0.955$)

Table 7. Marginal mean estimates of the number of insecticide applications per field per block in relation to farm. Dependent Variable: mean n sprays per field per block.

Farm	Mean	Std Error	95% Confidence Interval	
			Lower Bound	Upper Bound
Hants	0.642	0.097	0.406	0.879
Lincs	1.040	0.097	0.803	1.277
Yorks	1.515	0.097	1.278	1.752

Table 8. Marginal mean estimates of the number of insecticide applications per field per block in relation to treatment. Dependent Variable: mean n sprays per field per block.

Summer insecticide regime	Mean	Std Error	95% Confidence Interval	
			Lower Bound	Upper Bound
Normal	0.403	0.079	0.210	0.597
Elevated	1.728	0.079	1.535	1.922

The estimated mean number of applications per field per block in the blocks receiving the additional insecticides was approximately four times higher than that in the control blocks (Table 8) but there were also significant differences in the means between farms (Tables, 6,7). The mean number of applications per field per block in the blocks receiving the elevated insecticide inputs in the second year of the experiment at each farm was also significantly higher than the equivalent mean in the pre-treatment year (paired t-test: $t=5.72$, $d.f.=5$, $P<0.01$), whereas there was no difference between years in the control blocks (paired t-test: $t=0.402$, $d.f.=5$, NS).

Seed manipulations

A mixture of seeds of different weights was applied to two of the four blocks at each farm (Fig. 5). In all cases, the constituent seeds were mixed in equal quantities by weight. A combination of cracked maize, linseed and soya or barley was used at the Hampshire site. At the sites in Lincolnshire and Yorkshire, this mix was replaced by cracked maize, sorghum and millet, to accommodate crop hygiene requirements. The seed mix was broadcast across the surface of each field at a rate of 36Kg ha⁻¹ on three occasions between the end of November and early March the following year. Full details are given in Technical Appendix 7

Monitoring

The effects of the food manipulations were monitored by taking regular samples of seeds and arthropods, using a field-by-field sampling plan (Full details are given in Technical Appendices 5 and 6). The responses of the birds were monitored by undertaking counts of breeding birds, using a mapping method (Bibby *et al.*, 1992), over three breeding seasons (pre-treatment, treatment, and post-treatment) at each farm, and by carrying repeated field-by-field counts of all bird species between November and March. The winter counts were synchronised with the seed applications (Technical Appendix 7).

The yellowhammer was the subject of a more intensive study. The species was chosen for a number of reasons. During the breeding season, adult yellowhammers become central place foragers (Stephens & Krebs, 1986) while provisioning their young with arthropod prey. The birds forage extensively in crops but rarely select foraging patches more than 200m from their nests (Morris; Technical Appendix 2). They are, therefore, likely to be vulnerable to reductions in prey abundance induced by insecticide applications. As yellowhammers are seed-eaters outside the breeding season, they might also be expected to respond to the provision of supplementary seed during winter. Yellowhammers were sufficiently numerous in our study areas to provide adequate replication of observations at the block scale. The species is currently on the Red List of Species of Conservation Concern because it is currently undergoing a serious population decline nationally and is one of the species selected for the Farmland Bird Index (Table 2). During the breeding season, yellowhammer nests were monitored to gather information on chick growth and condition, and nesting success. In a sample of nests from the Yorkshire site, arthropod prey within 100m of the nest were also sampled to generate estimates of food abundance during the nestling period (Technical Appendix 3).

Analyses

The effects of the insecticide and seed treatments are analysed separately because it proved impracticable to consider them together.

Insecticides

Analysis of variance (ANOVA) was used to examine differences in settling densities between the breeding season when the insecticide manipulations were carried out and the following season. The difference in densities (n territories km⁻²) between years was the dependent variable and was presented as the log difference between Year t and Year t+1: $\ln(\text{density per block}_{t+1}/\text{density per block}_t)$. The spray treatment and site were fitted as factors, and the settling density per block in the treatment year as a covariate. The ANOVA was repeated for each of the five most abundant species: blackbird *Turdus merula*, dunnock *Prunella modularis*, skylark *Alauda arvensis*, whitethroat *Sylvia communis* and yellowhammer. All take invertebrate prey during the summer and foraged in the cropped portions of fields where the insecticides were applied.

No significant treatment effects were detected for blackbird, whitethroat and yellowhammer. In the case of dunnock and skylark, however, the mean log difference in settling densities did differ significantly between the treated and control blocks but the direction of the difference was not as predicted. Further investigations are required to interpret these provisional results.

The analyses then focussed on yellowhammer breeding productivity, particularly partial losses of nestlings from nests (brood reduction); it was assumed that these losses were due to starvation arising from the inability of the adults to find sufficient food. If the whole brood perished through repeated partial losses of young, the brood was assumed to have starved (Technical Appendix 3).

An initial analysis showed that the probability of brood reduction in a given nest was not related either to the insecticide treatment assigned to the block in which the nest was situated, or to the farm containing the block (Technical Appendix 4). As the extent of insecticide applications around each nest varied considerably within blocks, and it was not related to the experimental designation of the block, subsequent analyses were focussed on differences between nests (Technical Appendix 4).

As Morris (Technical Appendix 2) found that 90% of foraging trips by adult yellowhammers were made to patches within 200m of the nest, the extent of spraying was related to circular foraging ranges, each centred on a nest and with a 200m radius. One would expect the probability of brood reduction to increase as the proportion of the foraging range that receives insecticide increases. As local populations of arthropods tend to recover from insecticide applications within 3-4 weeks (e.g. Duffield & Aebischer, 1994), the effects of the insecticide are likely to be most severe when the applications were made either during or 3-4 weeks before nestling period. These predictions were tested by fitting variables describing the extent and timing of insecticide applications to binomial Generalised Linear Models (GLM) (with logit link and Bernoulli/Binomial errors) of the probability of brood reduction. Further details are given in Technical Appendix 4. Analyses were conducted on a nest-by-nest basis using continuous covariates to describe the extent and timing of the spray applications: proportion of the 200m radius foraging area sprayed ≤ 20 days before hatching, and proportion of the 200m radius foraging area sprayed > 20 days before hatching. A finer time division was explored but found to be unnecessary (Technical Appendix 4). The fit of the models was tested using the Hosmer-Le Cessie omnibus lack of fit test for a binary logistic model.

Seeds

The analyses addressed three potential effects of the provision of supplementary seed: (i) a short-term response by the birds following an application; (ii) a medium-term effect on the densities of birds remaining in each block at the end of the winter; and (iii) a carry-over effect on the densities at which birds settled during the breeding season following the seed applications. The analyses were restricted to five species of farmland bird which are partially or wholly granivorous during the winter: woodpigeon *Columba palumbus*, and four passerines, chaffinch *Fringilla coelebs*, linnet, skylark and yellowhammer. All five species forage for seeds on the ground and might be expected to respond to the provision of supplementary seed broadcast on the field surface. Further details are given in Technical Appendix 7.

The medium-term effects of the seed treatment were assessed by comparing changes in the densities per block of each species between seed treatments (SEED) and experimental site (FARM) using Analysis of Variance. The dependent variable was the log difference between the densities of a given species per block in February-March and the preceding November: $\ln(((\text{density}_{\text{Feb/Mar}}+1)/(\text{density}_{\text{Nov}}+1)))$. An interaction term between SEED and FARM was fitted. The ANOVA also included the density of birds per block in November as a covariate to allow for the possibility that the changes in densities over-winter were density-dependent.

Results

Insecticides

Effects on yellowhammer breeding productivity

An initial GLM showed a statistically significant effect of the proportion of the 200m radius foraging area sprayed ≤ 20 days before hatching on the probability of brood reduction in yellowhammer nests (Technical Appendix 4). The non-significant terms in the GLM were subsequently dropped and the model predictions made using only the significant term (Tables 9, 10, 11; Fig. 6). No lack of fit was detected (Hosmer-Le Cessie test: $Z=-0.65$, $p>0.05$). The inclusion of insecticide sprays applied after the hatching date but before the second weighing of broods did not improve the fit of the model. The probability of brood reduction quadrupled as the proportion of the foraging area sprayed increased from 0 to 1.

Table 9. Analysis of Deviance table for (reduced) logistic model⁽¹⁾ of brood reduction in yellowhammers (ignoring block structure and including only the significant covariate). Includes only years where extra spray treatments were applied and nests where at least one chick fledged.

Source of Variation	d.f.	Deviance	Mean deviance	Approx . ratio	F prob.
Prop ⁿ . sprayed ≤ 20 days from hatching	1	8.859	8.859	8.86	0.003
Residual	62	76.777	1.238		
Total	63	85.635	1.359		

Notes: 1: Generalised Linear Model (logit link, Bernoulli/Binomial errors) was fitted in GenStat® to brood reduction (0/1 binary indicator) using continuous covariates for insecticide. Site was treated as a 3-level factor.

Table 10. Grouped predicted values for probabilities of brood reduction in yellowhammers (model fitted without site terms and using only proportion of the 200m radius foraging area sprayed ≤ 20 days before hatching).

Prop ⁿ . sprayed ≤ 20 days from hatching	Probability of brood reduction	S.E.
0.00	0.26	0.067
0.25	0.40	0.066
0.50	0.56	0.091
0.75	0.71	0.114
1.00	0.82	0.111

Table 11. Parameter estimates for the model of the relationship between the probability of yellowhammer brood reduction and the proportion of the foraging range that had received insecticides within 20 days of hatching. N.B. model does not include Site term.

Term	Estimate	s.e.	t ₆₃	Antilog of estimate
Constant (<i>a</i>)	-1.068	0.353	-3.02	0.3438
Slope (<i>b</i>) for Prop ⁿ . sprayed ≤ 20 days from hatching	2.599	0.928	2.80	13.45

Notes:

To calculate brood reduction probability, *p*, at *x* = proportion sprayed ≤ 20 days from hatching, use:

$$p = \frac{e^{(a+b.x)}}{1 + e^{(a+b.x)}}.$$

Logistic model for brood reduction in treated years for fledged nests

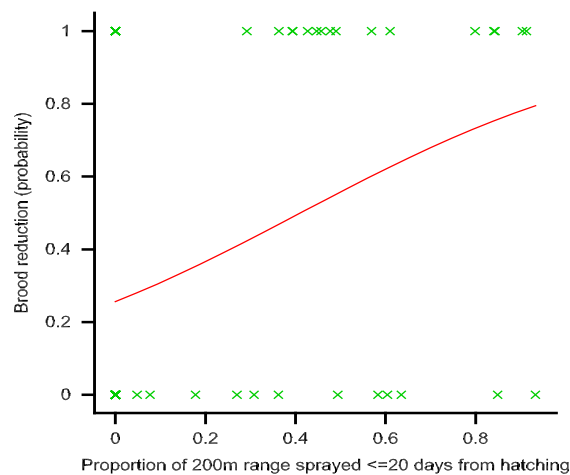


Fig. 6. Relationship between the extent and timing of insecticide application in the foraging ranges of yellowhammers provisioning dependent young in the nest, the probability of brood reduction. Observed values and predicted line shown. Extent of spraying measured as the proportion of a standardised foraging range (200 m radius) that received insecticides within 20 days of the date of hatching.

Seed experiment

Seed abundances

The densities of seed on the field surface declined between November and March on the farms in Hampshire and Lincolnshire but not in Yorkshire (Technical Appendix 6). At all three farms, seed densities in the blocks receiving supplementary seed were lower, in-mid winter, relative to those in the control blocks (Technical Appendix 6).

Short-term responses by birds to seed provision

Four of the five seed-eating species selected for analysis showed positive responses to the provision of seed at one or more sites. Three types of positive response were theoretically possible but, in the event, the birds exhibited only one, Type ii, which entailed an influx of birds from outside the experimental area into the seeded blocks but no change in numbers in the un-seeded blocks (Table 12). Woodpigeons exhibited Type ii responses at all three farms and following two consecutive seed applications at the Lincolnshire site. Chaffinch, linnets and yellowhammers also responded positively to the provision of seed but only at some sites (Table 12). Skylarks showed no consistent response to seed provision. Full details are given in Technical Appendix 7.

Table 12. Types of short-term responses¹ exhibited by five farmland bird species to seed applications at the three experimental sites in Hampshire, Lincolnshire & Yorkshire.

Species	Types of short-term responses to the provision of seed		
	Hants. 1 st application	Lincoln 1 st & 2 nd applications	Yorks. 3 rd application
Woodpigeon <i>Columba palumbus</i>	ii	ii, ii	ii
Chaffinch <i>Fringilla coelebs</i>	ii	iv	iv
Linnets <i>Carduelis cannabina</i>	iv	iv, ii	v
Skylark <i>Alauda arvensis</i>	iv	iv, iv	v
Yellowhammer <i>Emberiza citrinella</i>	v	ii, ii	V

Notes:

1. Types of response:

- (i) a redistribution of birds from the un-seeded to the seeded blocks following seed application but no overall change in numbers in the four blocks;
- (ii) an influx of birds from outside the experimental area but no change in numbers in the un-seeded blocks;
- (iii) an influx of birds from outside the experimental area and a redistribution of birds from the un-seeded to the seeded blocks;
- (iv) no consistent change in numbers, pre- and post-seed application;
- (v) no net change in numbers in either the seeded or un-seeded blocks.

Over-winter effects of seed provision on bird densities

Of the four species that had shown short-term responses to the provision of supplementary seed, only yellowhammer showed any evidence of an over-winter effect (Table 13). The interaction term between FARM and SEED was significant but the main effect was not. Over-winter trends in densities per block also differed significantly between farms and were negatively correlated with the densities of yellowhammers present in November prior to the first seed application. The density-dependence of the over-winter changes is shown in Fig. 7. On the un-seeded blocks, yellowhammer density increased over winter only when the November density was very low. The scatter of data points suggests that, for a given starting density in November, increases in over-winter densities were more likely in blocks that received supplementary seed than in those that had not. However, this pattern was not general to all farms because the co-ordinates for the two seeded blocks from the

Hampshire farm lay in the lower left-hand quadrant of the graph, which was the source of the significant interaction term between FARM and SEED.

Table 13. Summary statistics from ANOVAs of the relationship between log difference in densities per block^a, between the beginning (November) and end (late February/early March) of the treatment period, experimental site (FARM), and the provision of supplementary seed (SEED), with pre-treatment densities per block as a covariate.

	F values and significance levels ^b				
	Chaffinch	Linnet	Skylark	Woodpigeon	Yellowhammer
Intercept	7.90 *	0.002 NS	36.23**	1.48 NS	9.49 *
Pre-treatment density	6.07 (*)	0.19 NS	10.53*	1.75 NS	8.97 *
FARM	2.71 NS	0.17 NS	13.72*	1.06 NS	31.10 **
FEED	3.70 NS	0.09 NS	0.16 NS	2.08 NS	2.13 NS
FARM*FEED	2.78 NS	0.08 NS	4.98 NS	0.1 NS	14.27 *
Adjusted R ²	0.63	0.42	0.91	0.05	0.91

Notes:

^a $\ln \left(\frac{\text{density}_{\text{Feb/Mar}} + 1}{\text{density}_{\text{Nov}} + 1} \right)$;

^b NS = Not significant at 10% level; (*) $P < 0.1 > 0.05$; * $P < 0.05$; ** $P < 0.01$; *** $P < 0.001$;

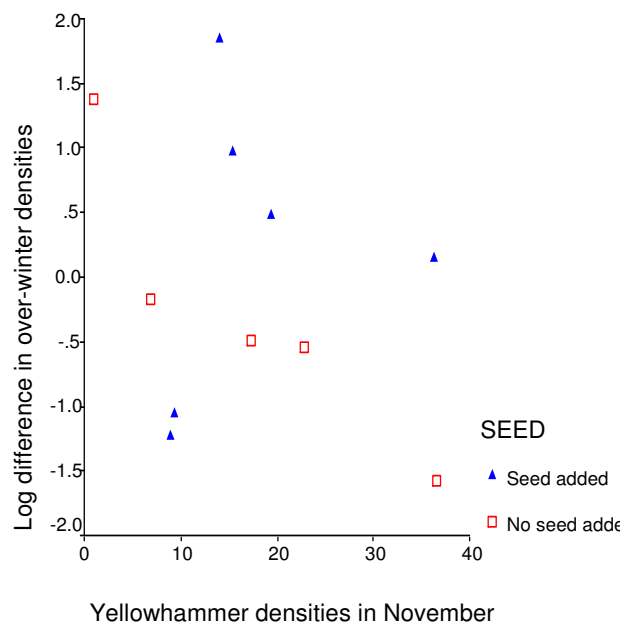


Fig. 7. Relationship between initial densities (n birds km⁻²) of yellowhammers per block prior to first seed application and subsequent over-winter changes in densities, expressed as the log difference between the initial and final densities: $\ln \left(\frac{\text{density}_{\text{Feb/Mar}} + 1}{\text{density}_{\text{Nov}} + 1} \right)$.

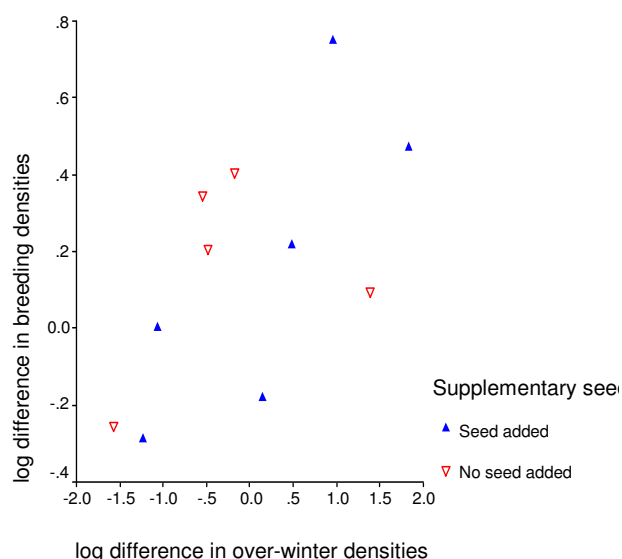


Fig. 8. Relationship between log difference in yellowhammer densities per block between November and March and log difference in settling densities per block in breeding seasons before and after seed applications. Correlation coefficient: $r=0.62$, $n=11$, $P<0.05$.

Breeding season effects

Densities of breeding yellowhammers increased in seven blocks, remained the same in one, and decreased in three in the breeding season following seed application. These changes were positively correlated with the over-winter changes in yellowhammer densities (Fig.8) but there was no consistent link with the provision of seed. There was no obvious clustering of the seeded and unseeded blocks, which would have been expected if there had been a seed treatment effect, whereas there was some evidence of a grouping of blocks by farm.

Discussion

Insecticides

The ‘proportion foraging area sprayed’ model for yellowhammer contains no information on the causal chain between pesticide input and brood reduction in the nest. This chain, which comprises linkages between insecticides and arthropod abundance, arthropod abundance and brood condition or growth rates, and between brood condition or growth rates and breeding output per nest, is treated as a ‘black box’. An intensive study at the Yorkshire site, in the summers of 2002 and 2003, confirmed the findings of the ‘proportion foraging area sprayed’ model by quantifying the linkages within the ‘black box’ (Technical Appendix 3). These form the basis of the risk assessment framework described under Objective 3 and in Technical Appendix 8.

The brood reduction model does not, in itself, give an indication of the likely population effects. However, a preliminary assessment of the likely population effects of varying the extent of insecticide applications around yellowhammer nests was made using the equation from the ‘proportion foraging area sprayed’ model, data on clutch and brood sizes from the Yorkshire site, and survival estimates from the scientific literature. Full details are given in Technical Appendix 8. With no spraying, the Population Growth Rate (PGR) had a value of 0.98 but, when the whole of the foraging range was sprayed, the growth rate decreased to between 0.9 and 0.95 depending upon the model specification. In all cases, the rate lay below the critical value (PGR=1) required to maintain a closed population.

Seeds

The supplementary seed was not recorded from most of the seed samples, even though the birds had clearly detected its presence. This may have been because the seed rain from the seed-spreader was so diffuse that the number of seeds expected to fall in the sampling areas was very small, or because the supplementary seeds had been severely depleted by birds before sampling was undertaken (Technical Appendix 6).

In-mid winter, the densities of seeds in the treated blocks were lower than those in the controls, which was unexpected. As this difference was not apparent on the first seed counts at the Hampshire and Yorkshire sites, it suggests that the decline in seed bank was linked to the seed treatment. The most likely explanation is that the increased foraging pressure from birds attracted to the supplementary food depleted the existing seed bank.

Overall Discussion.

As manipulations of the food supplies of birds on farmland are seldom carried out at the scale attempted in this project, some comments on the practicability of the experiment are appropriate.

The 1km² blocks were sufficiently large to detect effects on over-wintering numbers in the less mobile species, e.g. yellowhammer, but they were too small for woodpigeon, which has a large home range (Haynes *et al.*, 2003). Greater replication would have been desirable, to absorb the variability in the birds' responses between farms. The staggered entry of farms would permit the introduction of additional sites to achieve this.

Large blocks were selected in the hope being able to detect changes in settling densities in breeding birds between one season and the next but this proved not to be the case. In some cases, the lack of change may have been due to the buffering effect of immigration. The nest monitoring approach used for the yellowhammer proved to be workable, if labour intensive, and would be appropriate for investigating the demographic consequences for other farmland passerines arising from food supply manipulation during the breeding season.

Objective 3.

Examine the current risk assessment based regulatory procedures in relation to indirect effects of pesticides.

Background

As many of the prey species taken by farmland birds are non-target arthropods within the terms of the draft Guidance Document on Terrestrial Ecotoxicology Under Council Directive 91/414/EEC (European Commission, 2002), the testing protocol described in the Guidance Document applies. However, given the range of arthropod taxa taken by farmland birds, it is insufficient to rely upon the proxy organisms used in the Standard Tests. It is necessary to obtain data on an appropriate range of arthropods, at an appropriate time of year, to calculate species-specific chick food indices from crops treated with the candidate insecticide, and for appropriate controls. These data can only be obtained under field test conditions, which form part of the Higher-tier tests (European Commission, 2002).

Application of arthropod data from field tests to the risk assessment framework

Provided that the requisite field data are available, the frameworks devised for grey partridge (Fig. 1) and yellowhammer (Fig. 3) enable predictions of the likely population effects of particular insecticide products. A worked example of the newly developed framework for yellowhammer is presented here. It has been used to predict population growth rates under two extremes of arthropod abundance. The lower value is consistent with that observed from plots that had been recently treated with pyrethroid insecticides at the Yorkshire site.

Risk assessment framework for yellowhammer

Framework structure: an outline

The framework (Fig. 3) comprises a series of inter-linked deterministic model equations, which are shown in Box 1. Single values are fed into the model at the start to obtain single predictions of population growth rates. This version of the framework does not accommodate variability nor does it allow for density dependence. Equations 1 to 3 have been parameterised using data collected from the Yorkshire site in 2002 and 2003 (Technical Appendix 3). Full details of the treatment of those data and the parameterisation of the equations are given in Technical Appendix 8.

Equation 1 describes the relationship between the abundance of selected arthropod taxa (the yellowhammer chick food index) within the foraging ranges of nests and the growth rate of yellowhammer chicks, which are expressed as mean growth rates per brood. Predicted values of brood growth rates are then fitted to **Equation 2**, which describes the relationship between the probability of brood reduction and brood growth rates. **Equation 3** takes the predicted probability of brood reduction and multiplies it to the other terms shown in Box 1 to obtain an estimate of the number of fledglings per breeding attempt. **Equation 4** integrates this estimate across all breeding attempts to derive an estimate for the breeding season. The estimated number of fledgling per breeding season is then fitted to **Equation 5**, which includes terms for survival which are taken from the scientific literature, to derive a value for the number of individuals still surviving at the start of the following breeding season. In **Equation 6**, the number surviving in Year $t+1$ is then divided by the number of individuals in Year t to derive the population growth rate.

Box 1: Model equations used to populate the risk assessment framework for yellowhammer.

Equation 1.

$$\text{Brood Growth Rate} = m \times \text{Chick Food Index} + c$$

where

Chick Food Index = Number of Aranea, Diptera, Coleoptera, Hemiptera and Lepidoptera larvae per 0.5m² DVac sample

m= slope of linear regression

c= intercept

Equation 2

$$\text{Log} \left(\frac{p}{1-p} \right) = m \times \text{Brood Growth Rate} + c$$

where

p = Probability of Brood Reduction

m=slope of logistic regression

c= intercept

Equation 3

$$N_{\text{Fledglings}} = P(\text{Hatching}) \times (P(\text{Brood Red}) \times (1 - P(\text{Pred})) \times N_{\text{B+P}} + (1 - P(\text{Brood Red}) \times (1 - P(\text{Pred})) \times N_{\text{B-P}})$$

where

N Fledglings = Number of chicks that fledged per breeding attempt

P(Hatching) = probability brood surviving through egg-laying and incubation to hatching

P(Brood Red) = Probability that one or more chicks starve

P(Pred)=Probability that the whole brood is killed by predators

N_{B+P} = Number fledglings in nests that have suffered Brood Reduction but have not been Predated.

N_{B-P} = Number fledglings in nests that have suffered neither Brood Reduction nor Predation

Equation 4

$$N_{\text{Fledglings per season}} = N_{\text{Fledglings per Breeding attempt}} \times N_{\text{Breeding attempts per season}}$$

Equation 5

$$N_{t+1} = N_{\text{adults}} \times P(\text{adult survival}) + N_{\text{fledglings}} \times P(\text{post-fledgling survival}) \times P(\text{first-year survival})$$

where

N_{t+1} = population available for breeding next year

P(adult survival) = probability of adult surviving one year

P(post-fledgling survival) =probability that yellowhammer fledglings will survive first 14 days into dependence

P(first-year survival) = probability that this year's newly independent birds will survive their first year.

Equation 6

$$\text{Population Growth Rate} = \frac{N_{t+1}}{N_t}$$

where

N_{t+1} = Population at the start of next year

N_t = Population at the start of current year

Note on data sources

Two sets of model predictions are shown. The first set uses parameter estimates derived from the 23 territories in the Yorkshire site for which data on arthropod prey abundance were available. The second set of predictions is derived from all nests recorded from the Yorkshire site, for which data on breeding success were available. Estimation of the number of fledglings per season depends heavily upon an accurate determination of the number of breeding attempts made in each territory. Two sets of predictions are given: the first based on the observed mean value of 1.91 breeding attempts per territory; the second is derived from the breeding rule devised by Bradbury *et al.* (2000) from an autecological study of yellowhammers in Oxfordshire.

Framework predictions

Within the range of values measured at the Yorkshire site, increasing food abundance from the minimum (15) to the maximum (200), increased the number of fledglings in a season by less than one chick per pair of breeding birds (Figs. 9, 11). The increase depended upon which value had been chosen for the number of breeding attempts per territory and it was slightly larger when the Bradbury rule had been applied. If we extrapolate beyond the range of food densities measured at the Yorkshire site, we find the relationship becomes asymptotic. Increasing food abundance beyond about 5 times mean levels at the Yorkshire site leads to little further increase in fledglings or Population Growth Rate (PGR)(Figs. 10, 12). Though the difference in chick production is modest, its significance becomes apparent from the population growth curves; in three out of four cases, the asymptote of the curve does not lie above the critical threshold (PGR=1) for the maintenance of a closed population. The only exception was the curve predicted from the sample of 23 nests, using the Bradbury rule.

Discussion

The field-by-field counts of arthropods (Technical Appendix 5) suggested that the Yorkshire site supported relatively low densities of insects. At the Hampshire site, there were 2-3 times as many chick food items in both years of the study, while the Lincoln site, in its first year, had 3-4 times the density at the Yorkshire site. If it is assumed that, under conventional farming methods, insect numbers will rarely be more than four times those found at the Yorkshire site, then we might expect an additional 0.64 yellowhammer chicks per breeding pair on insect rich farms. Though this may appear a relatively small improvement, it would probably be sufficient to bring the population to a self-sustaining level. Indeed the relationship between chick food abundance and population growth rate suggests that a three-fold increase in insect abundance, if it could be fully converted into brood growth, would be enough to bring the population to a stable level.

It is also noticeable in Figs. 9 and 11 that by extrapolating the food abundance values to zero, the number of fledged chicks is still positive. This may be because the slope of the model is incorrectly estimated for values near zero. An alternative explanation is that by sampling randomly from crops within foraging range of yellowhammer nests, the experimenter may have missed important sources of food that yellowhammers did not. Yellowhammers may have found very local or very transient insect concentrations within the crops that the experimenter failed to sample. A third possibility is that the foraging adults may have found food in non-crop habitats, such as hedgerows, field margins or set-aside.

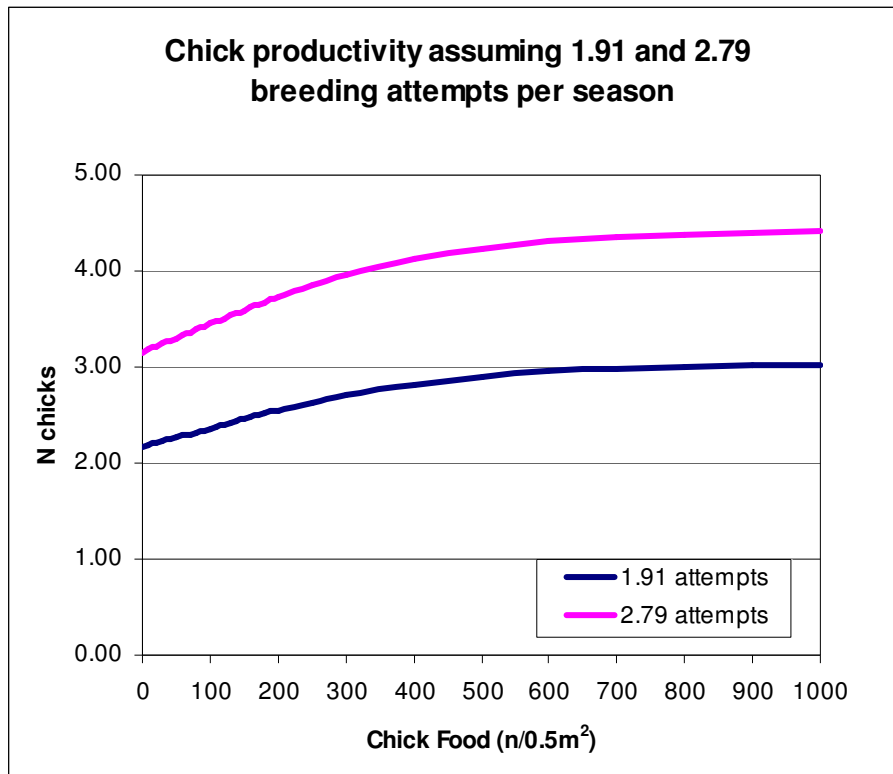


Fig.9. Fledgling productivity on 23 territories where food abundance was measured, comparing 2.78 breeding attempts per pair (based on breeding rule suggested by Bradbury *et al.*, 2000) and 1.91 breeding attempts per territory (empirical results from the Yorkshire site).

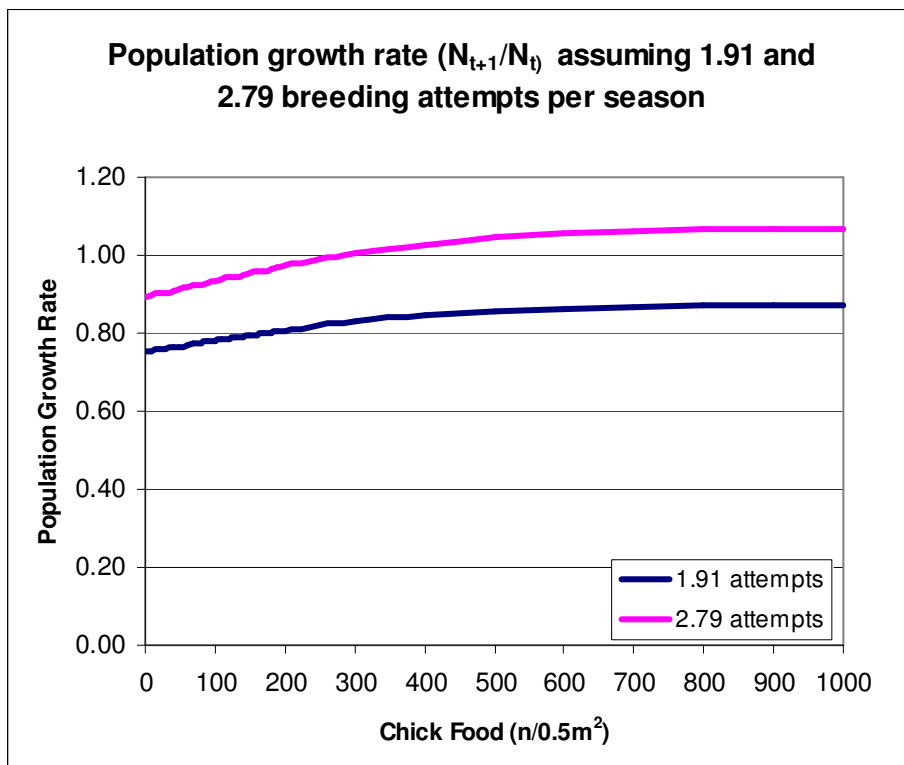


Fig.10. Population Growth Rate (N_{t+1}/N_t) on 23 territories where food abundance was measured, comparing 2.78 breeding attempts per pair (based on breeding rule suggested by Bradbury *et al.*, 2000) and 1.91 breeding attempts per territory (empirical data, Yorkshire).

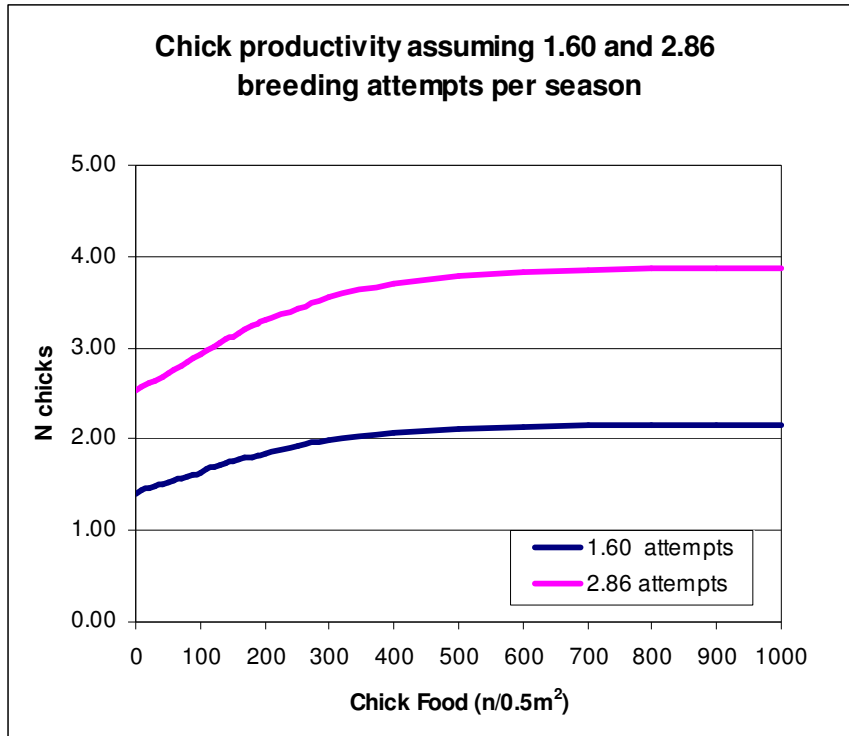


Fig. 11. Fledgling productivity on 98 yellowhammer territories, comparing 2.86 breeding attempts per pair (based on breeding rule suggested by Bradbury *et al.*, 2000) and 1.60 breeding attempts per territory (empirical results from the Yorkshire site).

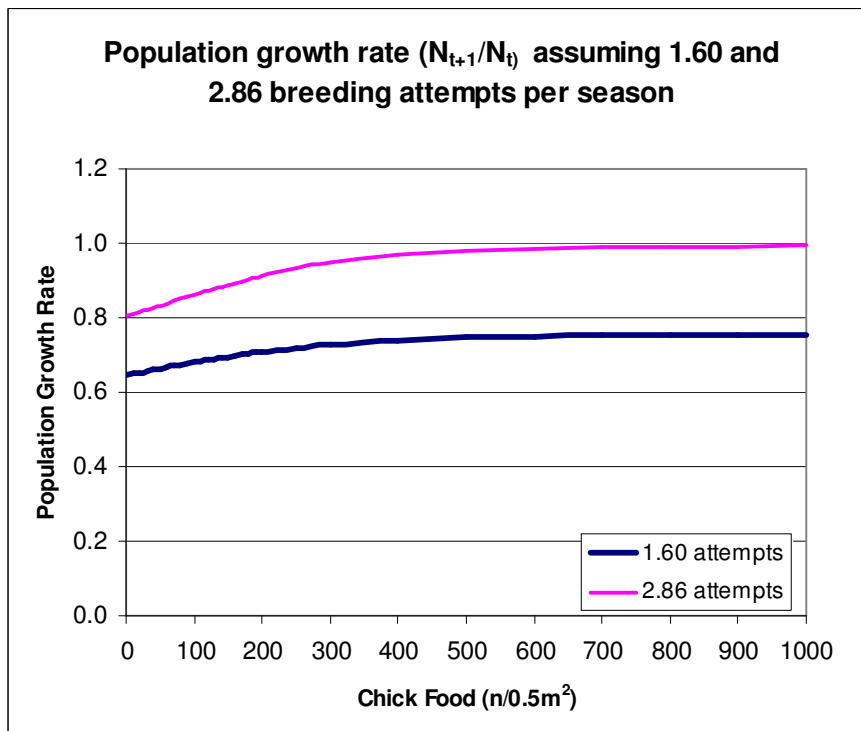


Fig. 12. Population Growth Rate (N_{t+1}/N_t) on 98 yellowhammer territories comparing 2.86 breeding attempts per pair (based on breeding rule suggested by Bradbury *et al.*, 2000) and 1.60 breeding attempts per territory (empirical results from the Yorkshire site).

Objective 4.

Propose suitable risk management practices that may reduce indirect effects of pesticides on birds.

Introduction

In principle, pesticides may disrupt the food chain of farmland birds at several stages but the risk assessment framework, which has been parameterised using data from the field experiment, has focussed on insecticides applied to arable crops in summer, and their effects on arthropods taken by adult yellowhammers provisioning their young in the nest. The food chain models, and the risk assessment framework derived from them, may be used to identify the range of arthropod abundances within which adverse impacts on breeding productivity are likely. When it is linked with data on the efficacy of individual insecticides, the foregoing approach permits an assessment of the likely impact of a given insecticide on the breeding productivity of those farmland bird species for which food chain models are now available. This approach is compatible with the current pesticide screening process because it facilitates a product-by-product assessment. In the case of the grey partridge, the requisite data exist to estimate an effect on population growth (G.C. Smith, in University of East Anglia *et al.*, 2002). However, for other species, data on age-specific survival rates are incomplete, so at present it would only be possible to estimate the upper and lower bounds of a population response, using a plausible range of survival rate values in cases where precise values are lacking.

For the purposes of insecticide screening, the food chain assessment assumes the worst case, which is that the depression of arthropod abundance that follows spraying occurs throughout the foraging range of adults from an individual nest, or nests of a notional population, and that no alternative arthropod food supplies are accessible. It is unlikely that this assumption would hold in practice because applications of a given insecticide across a farmland landscape are rarely synchronous, and field boundaries, which may provide alternative feeding habitats for some farmland bird species (Perkins *et al.*, 2002; Vickery *et al.*, 2002), are not treated, though they may be subject to spray drift (However, the field data were collected from fields with field boundary vegetation, so the availability of food from this source is factored into the food chain model). Moreover, the extent of insecticide applications is of fundamental importance to the outcome of a nesting attempt, as the ‘proportion foraging area sprayed’ models for yellowhammer have demonstrated.

Risk management to offset the effects of summer insecticide application could take one of two forms. Application could be restricted so as to reduce the impact within the crop (“mitigation”), or alternative resources could be provided (compensation). Compensatory measures may be in or outside the crop.

Mitigation

The ‘proportion foraging area sprayed’ model of brood reduction in yellowhammers illustrates that there is scope for mitigating the effects of summer insecticide applications by restricting the extent of their application. This could be achieved in several ways: (i) by minimising application events, (ii) by reducing the dose, (iii) by changing the timing of application and, (iv), by using unsprayed buffer zones.

Minimising application events

The avoidance of prophylactic or “insurance” spraying, by spraying only according to need and in response to thresholds, is widely promoted as part of integrated crop management (ICM) programmes, along with the encouragement of natural enemies and other measures to reduce the

probability of pest populations reaching damaging levels. Whilst the promotion of ICM is a valuable approach to minimising pesticide use, it is not suitable for use in a regulatory context because it needs to take account of site-specific factors, and so a prescriptive approach is not readily achieved.

Dose reduction

Insecticide dose is determined by the susceptibility of the target species. Where there is scope for reducing the dose, this is often done for economic reasons. However, this is not a suitable mitigation approach for regulatory use, since the reduction in dose needs to be determined on a case-by-case basis. Even where reduced doses are used, benefits will only accrue where non-target species are less susceptible than the target pest to the active ingredient concerned.

Changing timing of application

Diurnal adjustments to application timing have been used in an attempt to reduce exposure of non-target species, e.g. recommendations to spray pyrethroids in the early morning or late evening, to avoid effects on honeybees. However, it is unlikely that such a method could be used to reduce effects on chick food invertebrates, because of the range of taxa concerned. Indeed, some groups such as dipteran adults may be more vulnerable at such times because they are likely to be inactive, resting on the crop and so more at risk of exposure. The spraying date is generally determined by the need to respond to a pest outbreak, and opportunities for changing are likely to be limited. In any case, insecticide applications at any time during the breeding season are likely to affect breeding productivity of susceptible bird species to some extent.

Buffer zones

Buffer zones are amenable for use in a regulatory context and are already required for certain products. In 2003, buffer zones six metres wide from the water's edge were required next to watercourses for about 92 products, though for most of these, the width of the zone could be modified through a Local Environment Risk Assessment for Pesticides (LERAP) (Burn, 2003). However, only six pesticides (all insecticides) currently have statutory buffer zones to protect terrestrial habitats (Burn, 2003).

Unsprayed field margins next to suitable nesting habitat could be used to mitigate effects of insecticides (e.g. Snoo, 1995) for boundary-nesting species such as yellowhammer and tree sparrow, which feed in field margins (Perkins *et al.*, 2002, Vickery *et al.*, 2002). The clearest example of the potential benefits to be gained from restricting insecticide applications along the margins of fields is provided by a replicated field experiment undertaken by Rands (1985). He demonstrated that the abundance of arthropods taken as food by grey partridge chicks was higher in unsprayed than sprayed headlands of cereal fields. Moreover, partridge broods were larger on average in fields where unsprayed headlands were available. Not all farmland birds restrict their foraging to field margins, however. For example, foraging observations carried out during the current study and by others (e.g. Stoate, Moreby & Szczur, 1998) indicate that crops as well as field margins are also important foraging habitats for yellowhammers, and wider buffer zones than those designed to protect boundary vegetation may be needed to ensure sufficient invertebrate-rich feeding habitat. Buffer zones are not suitable for open field species such as lapwing and skylark.

Compensation

An alternative to mitigation is to provide alternative food resources in non-cropped habitats. Such habitats would need to be within the normal foraging range from suitable nesting habitat for the

species concerned, have a vegetation structure suitable for foraging and contain adequate densities of food items.

A range of managed habitats may be considered, many of which are (or are soon likely to be) eligible for financial support through agri-environment schemes. It is convenient to consider these in three groups: habitats that are normally created in field margins, those which are located in the main cropped area, and whole field measures.

Field margin habitats

Vickery *et al.* (2002) recently reviewed six margin options in terms of their value as foraging habitats for birds: grass margins, grass/wildflower margins, naturally regenerated (set-aside) margins, uncropped wildlife strips, game cover crops and conservation headlands. They concluded that grass/wildflower strips, uncropped wildlife strips and naturally regenerated rotational set-aside strips, followed by conservation headlands, would provide the most abundant summer food supplies, whilst game cover crops and naturally regenerated rotational set-aside strips would provide seeds over winter.

In-crop measures

In crop measures include beetle banks and skylark scrapes. Conservation headlands are also located in the crop, but cover the outer few metres next to the field boundary.

Beetle banks are grassy ridges created across field centres and were developed to provide additional overwintering habitat for predatory beetles and spiders, which are natural enemies of crop pests (Thomas *et al.*, 1991). However, they are also a potential source of invertebrates important in chick diet (Thomas *et al.*, 2001).

Undrilled patches are small, bare areas created in cereal fields by lifting the drill for a short distance as the crop is sown. They are currently the subject of a Defra LINK research project (Sustainable Arable Farming For an Improved Environment; SAFFIE). Early results have shown that, in fields with undrilled patches, there were more skylark territories that were maintained for longer, there were fewer nest failures, and more chicks were produced per nesting attempt (Morris *et al.*, 2004). As very few skylark nests were found in the patches, their main advantage appears to be in providing improved foraging habitat. Current work aims to further elucidate the mechanisms involved. Only a very small area is involved; patches are around 16-20m², at a density of four per hectare.

Whole field measures

Whole field measures include naturally regenerated set-aside, spring crops and associated overwintered stubbles. Naturally regenerated set-aside covers a large area and is potentially a major source of food supplies (e.g. Moreby & Aebischer, 1992), as well as providing nesting habitat for open field species. Generally, overwinter stubbles are followed by a further fallow period over the spring and summer, providing continuity of habitat and freedom from disturbance for the best part of a year. Unfortunately, this potential is generally not realised, for two reasons. Firstly, seed densities are low in the majority of stubbles, due to the efficiency of weed control in the previous crop. A recent study showed that most stubble fields supported no birds at all, and only a small number held high densities of birds. Pesticide use on the preceding crop was a major factor in determining this variation between fields (British Trust for Ornithology, 2002). Secondly, most rotational set-aside is sprayed with glyphosate, in April or May, to prevent weeds from seeding and to clear the ground prior to cultivations for the following crop. The destruction of the vegetation

early in the nesting season renders nesting birds vulnerable to predation, and also reduces the density of invertebrates by removing their habitat and food plants.

Spring crops are usually more open than winter crops, providing higher quality nesting habitat for species such as skylark (Donald & Vickery, 2000) and lapwing (Shrubb, 1990). Spring crops generally contain a higher diversity of weed species. Also, a number of species known to be important in bird diet are spring-germinating and are, therefore, more abundant in spring crops (Marshall *et al.*, 2003). Finally, spring crops are preceded by a stubble, though this may not persist for the whole winter as stubbles are often ploughed early to allow the creation of a “frost tilth”. Furthermore, as noted above, the value of many stubbles is limited because of weed control measures in the previous crop.

Conclusion

There is a range of measures available to mitigate, or compensate for, the effects of pesticides on bird food supplies. Information is available for some on invertebrate and seed densities, but not always for chick food groups, and often from a limited number of sites. Further work is also needed on the amount and location of such measures needed to provide the level of mitigation or compensation required. This will be affected by the foraging habits of the bird species concerned.

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List of scientific papers produced from project

Status of scientific papers produced or in preparation from Project PN0925.

Author/title/journal	Peer reviewed	Status
Boatman, N.D., Brickle, N.W., Hart, J.D., Milsom, T.P., Morris, A.J., Murray, A.W.A., Murray, K.A., Robertson, P.A. (2004). Evidence for the indirect effects of pesticides on farmland birds. <i>Ibis</i> , 146 (Supplement 2), 131-143.	Yes	Published
Crocker, D.R., Hart, J.D. & Milsom, T.P. (in prep). The indirect effect of pesticides on yellowhammers. II: Predicting the effects of summer insecticides on yellowhammer population growth rate. To be submitted to <i>Journal of Applied Ecology</i> .	Yes	In prep.
Hart J.D., Milsom T.P., Fisher G., Wilkins V., Moreby, S., Murray A.W.A. & Robertson, P.A. (under review). The indirect effect of pesticides on yellowhammers. I: The relationship between breeding performance, arthropod abundance and insecticide applications on arable farmland. Submitted to <i>Journal of Applied Ecology</i> .	Yes	Under review
Hart, J.D., Murray, A.W.A., Milsom, T.P., Parrott, D., Allcock, J., Watola, G.V., Bishop, J.D., Robertson, P.A., Holland, J.M., Bowyer, A., Birkett, T. & Begbie, M. (2002). The abundance of farmland birds within arable fields in relation to seed density. <i>Aspects of Applied Biology</i> , 67 (Birds and Agriculture), 221-228.	No	Published
Holland, J.M., Southway, S., Ewald, J.A., Birkett, T., Begbie, M., Hart, J., Parrott, D. & Allcock, J. (2002). Invertebrate chick food for farmland birds: spatial and temporal variation in different crops. <i>Aspects of Applied Biology</i> , 67 (Birds & Agriculture), 27-34.	No	Published
Morris, A.J., Bradbury, R.B. & Wilson, J.D. (2002a). Determinants of patch selection by yellowhammers <i>Emberiza citrinella</i> foraging in cereal crops. <i>Aspects of Applied Biology</i> , 67 (Birds & Agriculture), 43-50.	No	Published
Morris, A.J., Bradbury, R.B. & Wilson, J.D. (2002b). Indirect effects of pesticides on breeding yellowhammers <i>Emberiza citrinella</i> . The BCPC conference - Pests & Diseases 2002: 965-970.	No	Published
Morris, A.J., Wilson, J.D., Whittingham, M.J. & Bradbury, R.B. (in press). Evidence for indirect effects of pesticides on breeding yellowhammers <i>Emberiza citrinella</i> . <i>Agriculture Ecosystems & Environment</i> .	Yes	In press
Robinson, R.A., Hart, J.D., Holland, J.M. & Parrott, D. (2004) Habitat use by seed-eating birds: a scale-dependent approach. <i>Ibis</i> , 146 (Supplement 2), 87-98.	Yes	Published.