

Pesticides and their effects on lowland farmland birds

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The science and policy of nature conservation in the UK owes much to the study of the effects of pesticides on lowland farmland birds. During the 1950s and 1960s, a developing appreciation of the effects of organochlorine pesticides on bird populations led ultimately to a wider understanding of the potentially enormous scale - in space and time - over which such pollutants were capable of acting. As sentinels warning of such threats, farmland birds were the stimulus for restrictions followed by bans on the chemicals responsible, leading to the subsequent recovery not only of affected bird populations but also of other species such as Otter *Lutra lutra*.

The story of the organochlorine insecticides provides an important case study of population recovery in response to regulatory action taken against certain classes of insecticides and their replacement with less persistent alternatives. The occurrence of the principal metabolites of DDT (DDE) and dieldrin (HEOD) in the livers of Sparrowhawks *Accipiter nisus* has declined over the past three decades in response to successive restrictions on the use of those insecticides, between 1961 when there was a voluntary ban on the use of dieldrin as a spring seed dressing, 1974 when autumn seed dressing was stopped, and from 1969 when DDT uses began to be phased out in the UK (Fig.1). The pattern of changes in eggshell thickness, one of the physiological consequences of exposure to DDT in Sparrowhawks, has shown a similar time trend (Fig. 1 Newton 1995). The corresponding population decline of this species, which occurred chiefly in the period 1957-1963, and recovery which began in the early 1970s and was largely complete by the mid-1980s, has been well documented (Newton 1986).

The story of the organochlorines has certain parallels with the present concerns over the indirect effects of pesticides and their role in the decline of farmland birds. At first, evidence for the role of the organochlorines in bird declines was entirely correlational; it was not until evidence for the mechanisms of organochlorine action started to accumulate that regulatory action was taken (Newton 1995). As is evident from many other contributions in this volume, birds continue to be indicators that all is not well within the farmland ecosystem but for many species the evidence for the role of the indirect effects of pesticides is purely correlational. This review examines the evidence for the extent of risks and impacts due to the direct and indirect effects on farmland birds of pesticide use in UK agriculture. The effects of other aspects of farming intensification are considered elsewhere in this volume. This is a broad subject area, and the review is of necessity selective - it does not, for example, deal with the

impacts of veterinary medicines, many of which fall into the same classes as some agricultural insecticides. However, it is possible to draw attention to those areas where concerns should focus and in the process to highlight significant gaps in understanding.

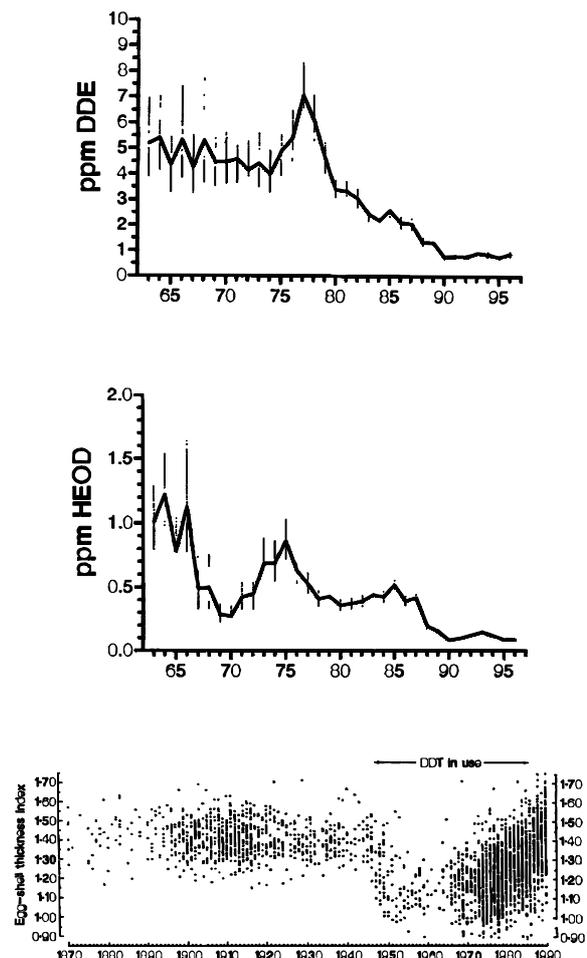


Figure 1. Trends in levels of organochlorine pesticides in livers of Sparrowhawks, and in eggshell thickness (reproduced from Newton 1995, with permission from *Journal of Animal Ecology*, and Newton *et al.* 1999).

EFFECTS OF PESTICIDES ON FARMLAND BIRDS

Pesticides can affect farmland birds in a number of different ways:

Direct lethal effects. Current perception is that direct effects do not now play a significant role in affecting bird populations in the UK (Campbell *et al.* 1997). The evidence for current impacts as well as risks of direct effects is examined in the first section of this review.

Direct sublethal effects. There is considerable evidence for the widespread exposure of birds to pesticides in the field. This review briefly considers the potential significance of such sublethal exposure for bird populations.

Indirect effects. The potential importance of the indirect effects of pesticides, both in the broader sense of their role in facilitating modern intensive farming methods, and the narrower sense of their impact on populations through effects on essential requirements such as habitat quality or food availability, are becoming more generally recognized (Campbell *et al.* 1997). This review briefly examines the evidence for their role in contributing to farmland bird declines. The review concludes by considering the ways in which various options for pesticide use within different farming systems might contribute towards the recovery of declining farmland bird populations.

Direct (lethal) effects - evidence from field observations

Although population recoveries of a number of species (such as Sparrowhawk and Stock Dove *Columba oenas*, Marchant *et al.* 1990) followed the restrictions on the use of organochlorine seed treatments, which had led to major mortality incidents in the 1950s and 1960s, the use of organophosphate seed treatments continued to result in bird mortalities during the 1970s and into the 1980s. Insecticide seed treatments pose a particular risk to birds owing to the attractiveness of grain as a food source and the high pesticide loading required to achieve adequate control of invertebrate pests. Evidence for avian mortality during this period and into the present is derived chiefly from monitoring carried out under the Wildlife Incidents Investigation Scheme (WIIS) of the Ministry of Agriculture, Fisheries & Food (MAFF). Under the scheme, mortality incidents of a range of domestic and wild vertebrates, as well as Honey Bees *Apis mellifera*, are investigated as part of the post-registration surveillance of pesticides used in agriculture. Fig. 2 shows the change in frequency of incidents attributable to different classes of insecticide seed treatment, as these were successively introduced. Although major incidents of avian mortality following the

introduction of the organophosphate (OP) seed treatments were relatively few, some had significant effects on certain vulnerable populations. Thus during 1971-1975 a series of incidents involving the use of the OP seed treatment carbophenothion resulted in mortality of Greylag Geese *Anser anser* and Pink-footed Geese *Anser brachyrhynchus*. In 1975, the numbers of Pink-footed Geese known to have died from eating such treated grain in the UK accounted for more than 1% of the world population of this species (Greig-Smith 1994). Such an unacceptable consequence of the use of an insecticide led to restrictions on its use in particularly sensitive locations and at times of year when grey geese were most vulnerable. It was replaced by an alternative organophosphate chlorfenvinphos, which posed a lower risk to geese although a greater risk to pigeons *Columba* spp. The predicted increase in pigeon mortality incidents is reflected in Fig. 2 (Greig-Smith 1994). In turn chlorfenvinphos was replaced during the mid-1980s by fonofos, which carries a lower risk to pigeons, resulting in around five incidents per year recorded during the late 1980s.

The current situation reported under WIIS (Fletcher *et al.* 1994, 1995, 1996, 1997, 1998) records fewer avian

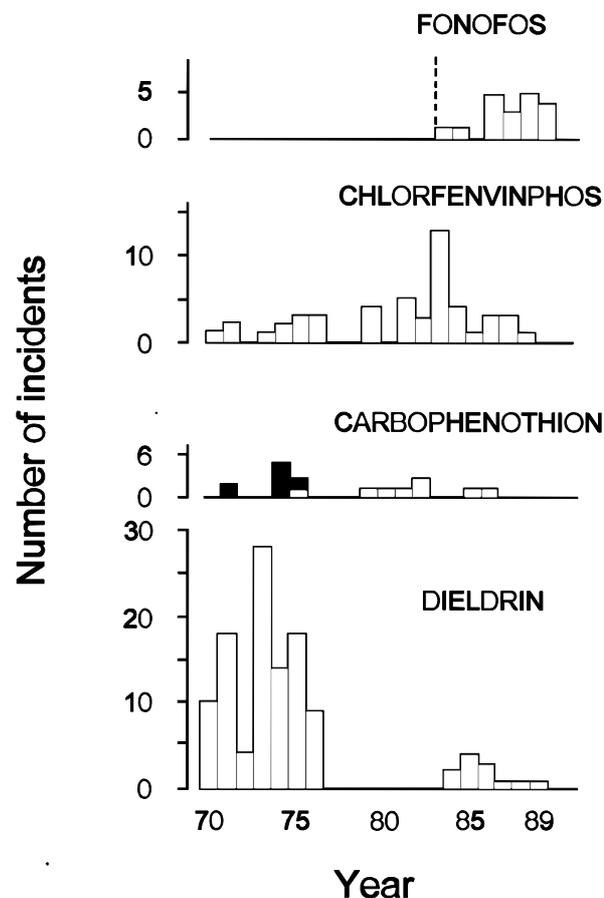


Figure 2. Numbers of incidents of wildlife poisoning due to insecticidal seed treatments in England and Wales from 1970 to 1989 (reprinted from Greig-Smith 1994, with permission from CRC Press, Boca Raton, Florida).

Table 1. Recent pesticide incidents involving birds (approved use, misuse, unspecified use), 1993-1997 (from Fletcher *et al.* 1994, 1995, 1996, 1997, 1998).

Pesticide type or use	1993	1994	1995	1996	1997
Organochlorine	2 (Merlin, corvid)	4 (Kestrel, Sparrowhawk, Starling, corvid)	3 (Kestrel, Sparrowhawk, Long-eared Owl)	1 (Kestrel)	1 (Grey Heron)
Organophosphate or carbamate seed treatment	3 (pigeon, gamebird)	3 (pigeon, gamebird)	3 (pigeon, gamebird)		
Other organophosphate	6 (Buzzard, Brent Goose, corvid, pigeon)	2 (corvid, passerines)	3 (pigeon, Red Kite, Starling)	1 (Marsh Harrier)	1 (corvid)
Other carbamate	3 (Buzzard, corvid)	1 (Peregrine)	1 (Buzzard)	1 (gamebird)	3 (Mallard, gamebird, gull)
Molluscicide	1 (Pheasant)	1 (Pheasant)			
Rodenticide	2 (Little Owl, Mallard)	3 (House Sparrow, Barn Owl, Kestrel)		1 (Goshawk)	

pesticide incidents attributable to seed treatments, although the approved use of agricultural pesticides does still lead to occasional incidents involving species of nature conservation concern (see Table 1). Incidents attributable to the deliberate abuse of pesticides in order to poison Corvidae etc. are not considered in this review. Such deliberate abuse incidents continue to be reported frequently and, together with other forms of persecution, are considered to have acted as a constraint on the spread of vulnerable raptors such as Hen Harrier *Circus cyaneus* and Red Kite *Milvus milvus* (RSPB/NCC 1991).

Of particular concern is evidence for probable secondary poisoning of birds of prey, possibly through eating contaminated earthworms following granular insecticide applications. Similar concerns have been noted in other parts of Europe, including France (Berny *et al.* 1997) and Switzerland (Jenni-Eiermann *et al.* 1996). Owing to the relatively short period when exposure is likely to occur, risks of population effects have been considered low. However, the occurrence of such incidents underlines the importance of WIIS in maintaining vigilance over such direct effects of pesticides.

One aspect of the direct effects of pesticide use where evidence suggests particularly close attention is needed is the use of rodenticides. Surveys carried out on behalf of the Joint Nature Conservation Committee (JNCC) by the Institute of Terrestrial Ecology (ITE) have shown an increase in the frequency of contamination of Barn Owls

Tyto alba by second-generation rodenticides. This has followed the increased use of such products as a result of the spread of warfarin resistance among rats *Rattus* spp., together, possibly, with more stringent hygiene requirements for grain storage (Table 2) (Newton *et al.* 1997a).

This monitoring is ongoing and at present the concentrations of residues recorded in livers of individual birds, together with symptomatic postmortem evidence, suggests that around 5% of contaminated Barn Owls (or less than 2% of the owls sent for analysis) may have ingested lethal doses of rodenticides. On this basis rodenticides are not thought currently to pose a risk to the Barn Owl population. However, it is not known what proportion of the contaminated population dies unrecovered, since lethally affected birds may die unobserved in roosting holes or nest sites. It is clear that Barn Owls appear more sensitive to second-generation rodenticides than do other bird species used in laboratory testing (Table 3), and lethal doses may approximate to ingestion of as few as 2 or 3 treated mice (Gray *et al.* 1994).

Given the wide range in avian susceptibility to rodenticides evident in Table 3, recent findings of Red Kites which have apparently been poisoned through the approved use or misuse (rather than abuse) of these products is of particular concern (Fletcher *et al.* 1999). Red Kites are likely to be particularly at risk of exposure to rodenticides, given their preference for carrion and

Table 2. Percentage of Barn Owls with livers containing residues of rodenticides, 1983-1996 (based on Newton *et al.* 1997a).

	1983-4	1985-6	1987-8	1989-90	1991-92	1993-4	1995-6
No. owls tested	20	76	64	141	162	139	109
% with residues	5	12	17	22	32	30	38

willingness to forage close to human habitation. Within one of the areas in England where attempts are being made to establish a breeding population from a small initial population of released birds, Brown Rats *Rattus norvegicus* are second only to Rabbits *Oryctolagus cuniculus* in frequency in the diet (Carter & Burn 2000.). This fact, taken together with the vulnerability of small populations, recent evidence (albeit limited) of exposure of this species in the field and of the spread of resistance of rats to second-generation rodenticides (and hence possibility of higher body burdens in a proportion of the prey of Red Kites) renders this instance of direct toxicity of particular concern to the nature conservation organisations and a challenge for the control of rats in regions with anticoagulant resistance.

Direct (lethal) effects - evidence of risk

Thus far, discussion of the direct effects of agricultural pesticides on farmland birds has concentrated on evidence from monitoring of larger species such as birds of prey, geese and pigeons. Mortality incidents involving smaller farmland birds such as songbirds are very infrequently reported, both in the UK and in North America (Mineau & Collins 1988, Baillie 1993). There have been instances of kills of flocks of finches Fringillidae, such as a mixed flock largely comprising Greenfinches *Carduelis chloris*, which probably died drinking from a puddle contaminated by the herbicide monochloroacetate in 1989 (Greig-Smith 1994). There are evidently fewer such incidents now than during the 1980s, when 73 incidents involving up to 100+ songbirds were recorded between 1980-1989 (Greig-Smith 1994). Much recent investigation has focused on the extent

to which such a decrease in recorded incidents may be attributable to under-reporting rather than a true reflection of the use of less hazardous products and improvements in application practices.

Several studies, both in the UK and elsewhere (Mineau & Collins 1988, Hart 1990) have shown that the reliability of finding dead birds is low, and that this is more marked for smaller species. In North America, for example, consideration of the low reporting frequency and the occurrence of incidents involving songbirds has led to the conclusion that the use of granular insecticides there during the 1980s had the potential to have affected population levels of farmland birds significantly (Mineau & Collins 1988). In the UK, the frequency of recovery of dead birds under the British Trust for Ornithology (BTO) ring-recovery scheme has been used to develop a reporting frequency index for birds, which can be used to weight the evidence for mortality incidents reported for different species under WIIS (Table 4 - Baillie 1993).

It has been proposed that regulatory action might be triggered where monitoring under WIIS indicated that 1% or more of a population has been affected (less if dealing with a rare species), or 10 or more incidents per annum are reported (Cooke 1990). Using the relative weighting proposed in Table 4, one would expect a single incident involving Tree Sparrows *Passer montanus* to trigger the same level of concern as, say, 10 incidents involving Lapwings *Vanellus vanellus*.

In summary, then, we cannot completely discount the possibility that substantial mortality of smaller species may be occurring on the basis of the lack of reported mortality incidents. So are there other sources of evidence that may be used to assess whether there is in fact a risk of serious

Table 3. Comparative toxicities of some second-generation anticoagulant rodenticides to Barn Owls and other test species (from Newton *et al.* 1990, 1997b, Tomlin 1997).

Rodenticide	Conditions of use	Acute oral LD ₅₀ rat (mg/kg)	Acute oral LD ₅₀ birds (mg/kg)	Range of lethal dose Barn Owl (mg/kg)
Brodifacoum	under cover only	0.27	4.5 ^a , 0.31 ^b	0.29 - 1.25
Difenacoum	outdoors	1.8-2.45	>50 ^a	0.11 - 0.17
Bromadiolone	outdoors	1.125	138 ^c	0.33 - 1.72
(a) Chicken	(b) Mallard	(c) Bobwhite Quail		

Table 4. Reporting probabilities (%) for a range of lowland farmland birds found dead (based on Baillie 1993).

Species	Reporting probability (%)
Mute Swan	16.75
Sparrowhawk	5.50
Lapwing	1.12
Woodpigeon	1.96
Barn Owl	8.18
Yellow Wagtail	0.20
Rook	2.45
House Sparrow	0.87
Tree Sparrow	0.15
Chaffinch	0.43
Yellowhammer	0.24

direct effects of pesticides? Because of the high risks of avian exposure, many of the investigations into relative risks of mortality in the field have concerned the effects of seed treatments, although granular formulations which also carry high acute risks have also been the subject of numerous studies (Mineau 1988, Best & Fischer 1992, de Leeuw & Luttick 1995).

Based on a simple measure of risk that is widely used during pesticide registration procedures, the toxicity:exposure ratio (TER), the frequency of occurrence of pesticide seed-treatment incidents involving birds shows only a limited degree of correspondence with the predicted risk (Table 5), although seed treatments such as lindane, with a relatively high TER, have among the lowest frequency of reported incidents.

However, the TERs reported in Table 5 do indicate that a bird feeding on seeds treated with fonofos, for example, is likely to be exposed to several lethal doses per day. This would suggest that mortality incidents should be more frequently recorded (notwithstanding the difficulties in finding corpses). That this is not the case is in part because the risks of exposure can be effectively managed by

efficient burial of treated seeds (Pascual *et al.* 1999). However, we are also beginning to develop a better understanding of the factors that moderate the risks that seed treatments pose for birds, which operate both at the individual level (such as avoidance), and at the population level (such as the proportion of the population feeding within treated areas).

The basis for the apparent lack of serious mortality incidents due to modern seed treatments appears to depend significantly on their generally low palatability (Hart *et al.* 1999). Thus, for example, Woodpigeons *Columba palumbus* feeding on newly drilled seeds avoid fields treated with fonofos (McKay *et al.* 1999). However, recent laboratory studies (Hart *et al.* 1999) have shown that various factors can affect seed palatability and the avoidance response of birds, and so affect the risk of a lethal dose being ingested. The key factor is the rate at which seed is eaten: at a sufficiently high rate of intake a lethal quantity of treated seed can be ingested before aversion sets in. The relationship between feeding rate and risk of a lethal effect has been intensively studied for pigeons feeding on fonofos-treated seed. Fig. 3 shows that a number of factors can in turn affect the rate of feeding. Feeding in flocks, for example, increases the rate of feeding by individuals. Hunger and limited time available for feeding also increase the rate of feeding and reduce the effectiveness of the avoidance response in pigeons sufficiently to cause mortality (Fig. 3). Risks of mortality due to seed treatments can therefore move from (relatively) low to high for larger species such as pigeons, especially if hungry, feeding in flocks, or given limited feeding time. The fact that very few mortality incidents, or even evidence of sublethal effects, have been observed in intensive studies of the Woodpigeon (Cooke 1988) suggests that the factors that increase risk are unlikely regularly to be a significant feature. However, for smaller species the risks may be greater, both in terms of rate of feeding required to reach a lethal dose, and rate of reaching a hunger threshold that overrides an avoidance response. In addition, the evidence for lack of effects on such species (from WIIS) is perhaps less reliable. These findings have important consequences

Table 5. Mean lethal doses (acute) and toxicity:exposure ratios (TERs) for birds for some cereal seed treatments.

Seed treatment	LD ₅₀ (mg/kg)	TER	Records of mortality incidents in UK
Dieldrin	4 ^a	~1	many
Lindane	120-130 ^a	1-10	single incident
Chlorfenvinphos	13.5-25.8 ^b , 107 ^c	0.1-1	moderate frequency
Carbofenthoion	29-35 ^d	0.1-1	low frequency
Fonofos	128 ^e , 13.3 ^b	0.11	low frequency

(a) Bobwhite Quail; (b) Pigeon; (c) Pheasant; (d) Canada Goose; (e) Mallard.

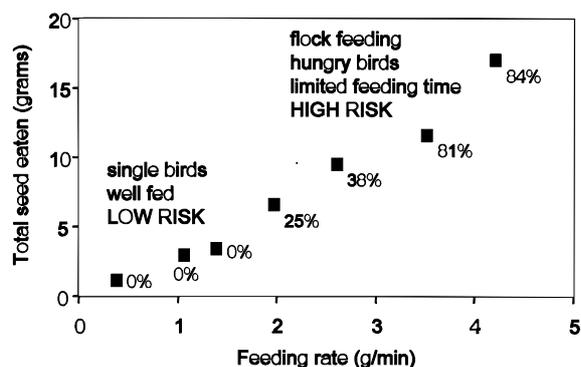


Figure 3. Effect of feeding rate on total seed eaten (and hence risk) for pigeons feeding on fonofos-treated wheat. Percentages show mortality in each treatment (reproduced from Hart *et al.* 1999, with permission from Birdlife South Africa).

for the assessment of risks to birds from the use of treated seeds in the field.

For smaller species such as Corn Bunting *Miliaria calandra*, which appear to feed extensively on newly drilled seeds in spring (Brickle 1999) and may need to feed rapidly to make up for loss in body weight overnight, particularly high-risk situations can be envisaged. It was, indeed, such a risk situation that, together with evidence of mortality in the field, led to the initial voluntary ban on the use of dieldrin for spring-drilled seeds. In situations of low or declining alternative food availability, spring-drilled seeds may become a more important food source. In addition to the likelihood that other seed sources are at their scarcest in spring, birds may be seeking high intake rates of particular food items then in order to reach breeding condition. Given the relative scarcity of spring-drilled cereals in many areas, single fields with treated seed may have the potential to affect birds from a wide area in the surrounding landscape (J.D. Wilson pers. comm.). The potential therefore exists for interactions between the direct effects of pesticides used as seed treatments, and indirect effects (described below) that affect food availability. It is likely that for individuals of such species a fine balance exists between mortality and survival depending on whether their exposure exceeds a lethal level before an avoidance response (or regurgitation, Pascual *et al.* 1999) takes place. Studies of the autecology of declining farmland birds that are currently underway should be of value in pesticide risk assessments by providing a better understanding of the extent and timing of food shortages for these species, and their possible significance for rates of feeding on treated seeds.

In addition to the response shown by individuals, a further major factor that affects the actual, as opposed to the predicted, risk of mortality due to pesticides is that birds do not feed exclusively within a localised, treated, area. A recent study of the risks to birds of insecticide use in orchards showed that much of the difference in bird numbers could be accounted for by differences in habitat

and in intensity of orchard management (Crocker *et al.* 1998a). The Tree Sparrow, for example, was among the ten most abundant species recorded in more traditionally managed orchards but was virtually absent from more modern orchards (Crocker *et al.* 1998a). Up to 25% of the variation in overall numbers of birds could be accounted for by differences in pesticide use. However, intensity of insecticide use is closely linked to the type of orchard management, so much if not all of this difference could in fact be due to other habitat and management factors. If pesticides do account for a significant proportion of the observed variation, this is probably due to indirect rather than direct effects since birds spend only a small proportion of their time within the treated area. A radio-tracking study of Blue Tits *Parus caeruleus* in orchards has shown that only a very small proportion of the birds that utilise the orchard spend sufficient time there to be exposed significantly to insecticide residues in their prey. Exposure of the population is therefore very much lower than simple observations would predict (Crocker *et al.* 1998b). Such studies have prompted the consideration of pesticide risk assessments based on the probability of exposure, rather than simple TERs, which should give a more realistic assessment of risks of unacceptable population effects (Crocker *et al.* 1998b).

Sublethal effects

Consideration has so far been given to the evidence for direct lethal effects, based either on field monitoring (principally through WIIS) or on risk assessments (often including field studies), particularly focused on seed treatments. However, sublethal effects can, through their effects on individual behaviour and physiology, have potential consequences for populations. This has been well documented, for example in the case of Sparrowhawks and DDT (Fig. 1). The consequences for farmland bird populations of sublethal effects associated with pesticide use therefore need to be considered. Three distinct bodies of evidence are important here. First, evidence for exposure to sublethal concentrations of pesticides in the field; second, evidence that sublethal exposures can carry a risk of significant physiological or behavioural effects; third, that such effects can have significance for populations in the field. Whilst there is abundant evidence for the first two of these, there is very little direct evidence that points to the operation of population consequences for birds due to the sublethal effects of modern pesticides, and none at all for farmland birds in the UK.

The pesticides that inhibit acetylcholinesterase (AChE) (chiefly the organophosphate and carbamate insecticides) can have acute effects at field dose rates, and their effects on enzyme activity can be used as a biomarker. As a result they have been intensively studied for the occurrence of

Table 6. Weights and wing lengths (\pm se) of 10-day-old Tree Sparrow nestlings subject to different levels of pesticide regimes, Boxworth, Cambs, 1987 (from Hart *et al.* 1992).

	Weight (g)	Wing length (mm)
Intensive pesticide treatment ($n = 10$)	16.0 \pm 1.0	39.9 \pm 2.5
Less intensive and integrated pesticide treatments ($n = 7$)	19.2 \pm 0.5	48.5 \pm 1.3

sublethal effects in birds. There is evidence for the depression of cholinesterase activity following field exposure to AChE inhibitors in a range of farmland species, including Swallows *Hirundo rustica* (Fossi *et al.* 1994) and Tree Sparrows (Thompson *et al.* 1988); frequent exposure to this group of insecticides either through overspraying, drift or contamination of prey items seems beyond doubt (Mineau 1991). Similarly, a large body of research has demonstrated that the AchE-inhibiting insecticides almost uniquely affect almost all physiological and behavioural functions, including thermoregulation, food consumption (via either pesticide-induced changes, or conditioned aversion causing unpalatability and limiting exposure as described above), reproductive activity (including sexual behaviour, clutch size, egg and embryo development) and other behavioural changes (including mobility/motility, migratory behaviour, territorial behaviour and care of young) (Grue *et al.* 1991, 1997). It has been suggested (Grue *et al.* 1997) that such physiological and behavioural effects have the potential for reducing the productivity and survival of individuals in the field, and experimental studies have demonstrated that such mechanisms may operate under field conditions. For example, effects on motility can in turn affect predation risk for House Sparrows *Passer domesticus* and Bobwhite Quails *Colinus virginianus* (Hunt *et al.* 1992, Hawkes *et al.* 1996); changes in ability to locate, capture and even manipulate food (such as dehusking seeds) reduces rates of feeding and leads to weight loss (Fryday *et al.* 1994); poorer thermoregulatory performance in American Kestrel *Falco sparverius* has resulted in mortality when exposed to cold temperatures (Rattner & Franson 1983). Similar responses have been shown in field or semi-field situations - for example changes in nestling care of Starlings *Sturnus vulgaris* (Grue *et al.* 1982) and in incubation behaviour of gulls *Larus* spp. (White *et al.* 1983).

In a study of Tree Sparrows as part of the long-term study of effects of intensive pesticide use at Boxworth, England, chick development rate was significantly delayed in one year in areas that had been intensively treated with insecticides (Table 6). Following summer organophosphate aphicide applications in intensively treated fields, adult Tree Sparrows showed a marked change in feeding

preference, taking a higher proportion of Hemiptera (chiefly aphids) and smaller proportion of ground-dwelling Coleoptera (Hart *et al.* 1992). The contaminated aphid prey therefore formed a large part of the chick's diet immediately post-spraying, and blood cholinesterase activity in exposed chicks was significantly depressed. However, it was not possible to determine whether the observed differences in growth rates of chicks between intensively and less intensively treated areas were attributable to the direct effects of sublethal exposure or to a possible indirect effect of the changes in availability or nature of prey items taken. Nor, within the constraints of this experimental design, was it possible to observe any subsequent effect on Tree Sparrow abundance.

In fact, the majority of recent studies of changes in reproduction or survival of free-living wildlife following sublethal exposure to AChE inhibitors have found few, if any, effects (Bennett 1994, Grue *et al.* 1997). A measurable change in an avian population as a result of sublethal pesticide exposure following its field application has been shown in a study of White-throated Sparrows *Zonotrichia albicollis* in Canada (Busby *et al.* 1990). In this case, forestry spraying operations using the organophosphate insecticide fenitrothion resulted in a significant depression in brain cholinesterase activity in breeding adults. A range of behavioural changes was observed including reduced territorial activity, disruption of incubation behaviour and nest abandonment. In turn, this resulted in a significant reduction in fledging success and lower production of fledged young. Studies in similar forestry situations in the UK have not demonstrated similar population-related effects (Spray *et al.* 1987), but in this case the smaller size of areas treated possibly led to moderation of exposure risks as described for Blue Tits in orchards, above.

Conclusions - direct effects

In conclusion, the impact of both lethal and sublethal direct effects of pesticides on farmland birds cannot be discounted but there is little evidence that they are currently having significant population effects. In a few cases, such as rodenticides, concern over potential impacts is increasing; in most others, concern arises from

assessment of risks rather than from evidence of impacts on populations. It is clear that in the case of seed treatments, for example, the absence of serious effects is very dependent on risk management measures as well as on the physiological and behavioural status of the birds themselves. These factors may in turn be modified by the indirect effects of pesticide use.

Indirect effects of pesticides

Although the extent to which the indirect effects of pesticides have had an impact on farmland bird populations remains unclear, it is generally considered that indirect effects, rather than direct effects, pose the most serious risk to populations at present although other factors of agricultural intensification are also known to be important. A recent review (Campbell *et al.* 1997) examined three broad classes of evidence for the role of indirect effects of pesticides on farmland birds. For most declining farmland bird species available evidence was largely correlative, with temporal relationships between trends in pesticide use, changes in arthropod and plant food availability, and timings of population declines. Other classes of evidence concerned the existence of mechanisms linking the observed changes in abundance with pesticide

use, and experimental evidence for changes in populations or population parameters as a result of pesticide manipulation, although very little evidence from the latter two categories was available for most species.

Table 7, derived from Campbell *et al.* (1997), summarises the correlational evidence for those lowland farmland birds that are undergoing serious population declines. In the case of Linnet *Carduelis cannabina*, more recent analyses indicate a population recovery on farmland following a decline between the late 1960s and late 1980s (Siriwardena *et al.* 1998a); Siriwardena *et al.* (1998a) also raise doubts about the evidence for a recent population decline in Swallows. The conclusions of Campbell *et al.* (1997) in Table 7 have been modified accordingly. The final column in Table 7 summarises the overall picture for those species.

A more detailed analysis of the long-term changes in abundance of farmland bird populations and the timing of those changes has been carried out by Siriwardena *et al.* (1998a). This analysis has confirmed that, as a group, farmland “specialists” have undergone the greatest declines in abundance, and that it is probably the general intensification of agriculture that has been responsible. However, although a third of farmland species showed significant downturns at about the same period during the mid-1970s, the variation in timing of other downturns

Table 7. Categories of evidence in support of an indirect effect of pesticides. Evidence is possible (‘P’) or more firmly established (‘Y’) (based on Campbell *et al.* 1997).

	Is population declining?	Are diet items declining?	Does bird decline coincide with pesticide use?	Is there correlative evidence for an indirect effect?
Grey Partridge	Y	Y	?	Y
Tree Sparrow	Y	Y	Y	P
Turtle Dove	Y	Y	Y	P
Bullfinch	Y	Y	Y	P
Song Thrush	Y	Y	Y	P
Lapwing	Y	Y	Y	P
Reed Bunting	Y	Y	Y	P
Skylark	Y	Y	Y	P
Linnet	?	?	Y	P
Swallow	?	Y	?	P
Blackbird	Y	Y	Y	P
Starling	Y	Y	Y	P
Corn Bunting	Y	Y	?	(P)
Yellow Wagtail	Y	Y	?	(P)
Yellowhammer	Y	Y	?	(P)

The Grey Partridge is the only species for which evidence of an indirect effect of pesticides has been firmly established experimentally, but such an effect, operating via the effects of insecticides and/or herbicides on food availability, appears possible or even likely for a number of other species.

Table 8. Evidence for an indirect effect of pesticides on the Grey Partridge.

Effect	Source of evidence
Reduction in prey availability linked to pesticide use	experimental studies and long-term monitoring
Survival of chicks linked to invertebrate food availability	foraging preferences and caged experiments on survival rate
Population size affected by chick survival	simulation modelling and field monitoring
Experimental increase in productivity and population density	pesticide manipulation in conservation headlands

in populations and the range of trends found argue against a single element of farming intensification as an explanation for the recent declines in farmland birds. Different species are likely to have been affected by different factors associated with farming intensification.

The evidence that pesticide use can affect and has affected food availability over the long term has been supported by a recent re-analysis of data collected over a 30-year period within The Game Conservancy Trust's (GCT) Sussex study area (Ewald & Aebischer 1999). Again, evidence is correlative but this work has confirmed clear long-term declines in a range of invertebrates that form important components of the diet of many farmland birds (Potts 1986). On a spatial scale, population densities of invertebrate prey items were found to be inversely related to the intensity of use of broad-spectrum insecticides since the mid-1970s, with evidence for effects carrying over into successive years, but with little evidence for adverse effects arising from the use of more selective insecticides. Evidence from this analysis for the effects of herbicides on long-term changes in the frequency of occurrence of weed species over the period of the study is less convincing. However, the lack of quantitative estimates of individual weed species abundance, together with the fact that almost 100% of the area of study was receiving herbicide treatment prior to the start of the study, makes this result less surprising. Other evidence (Campbell *et al.* 1997, Wilson *et al.* 1999) points to significant changes in weed abundance in arable areas in recent decades, which may both reduce the availability of weed seeds for birds, especially in winter, as well as removing host plants for invertebrate prey (Aebischer 1990). Again, correlative evidence indicates a spatial relationship between intensity of herbicide use and indices of abundance for three bird species, Grey Partridge *Perdix perdix*, Skylark *Alauda arvensis* and Corn Bunting in the GCT Sussex study area - all three occurring less frequently in fields that had received recent herbicide treatments (Ewald & Aebischer 1999). More detailed analysis of regional changes of different types of pesticide use might throw more light on likely relationships between pesticides and population trends for individual bird species. Arguably, however, there is little to be gained from further correlational studies, and the focus now should turn on the one hand to understanding the mechanisms

behind declines and experimental manipulation of factors thought to be responsible, and on the other to developing pragmatic solutions.

Indirect effects of pesticides - evidence from autecological studies

The most convincing evidence for the operation of indirect effects of pesticides at the population level is that for the Grey Partridge (Potts 1986). The various links between the mechanisms involved and observed declines in Grey Partridge populations are summarised in Table 8.

The importance of invertebrate prey availability for partridge chick survival has been established through caged experiments on foraging preference and on rates of chick survival with access to different levels of invertebrate prey, and field-scale correlations between chick survival rates and prey distribution (Potts 1986). Chick survival appears to be a key factor in determining partridge population levels, based on field evidence of the relationship between population size and breeding success, and simulation models that indicate that good chick survival leads to a higher equilibrium population level independent of the other mortality factors of shooting and predation (Potts 1986). Wide-scale declines in important invertebrate prey items correlate with pesticide use over the long term; however, other factors, including undersowing, have a major role together with herbicide use in affecting the abundance of important herbivorous invertebrate prey species (Aebischer 1990). The early decline in Grey Partridge numbers (during the 1950s and 1960s) correlates with an increase in herbicide use at that time, although changes in chick survival rates are less clearly correlated with herbicide use (Potts 1986). The most conclusive evidence derives from experimental manipulation of pesticide use in "conservation headlands". Reductions in the use of insecticides in summer, and of broad-leaf herbicides in relatively narrow (6 m) strips adjacent to field boundaries, have led to increases in weed and invertebrate densities in those areas (Moreby & Southway 1999), a significant increase in chick survival rates ($\times 1.68$), and subsequent increase in densities of Grey Partridges within the treatment area (Rands 1985).

No other farmland species has been studied so

intensively over such a long period as the Grey Partridge. Evidence for the role of indirect effects of pesticides on other bird species is therefore less readily available, and much evidence for the existence of likely or possible mechanisms for such effects is likely to become available from the autecological studies currently underway, many of which are described elsewhere in this volume.

Factors behind the decline in Corn Bunting population densities are discussed elsewhere; the available evidence does, however, support the view that pesticide use may play an important role in the dynamics of this species also (Brickle *et al.* in press). Nestling diet is comparable to that of Grey Partridge chicks (in common with Skylark, Cirl Bunting *Emberiza cirrus* and Yellowhammer *E. citrinella*). Areas within which adults foraged averaged fewer insecticide and fungicide applications, and had higher densities of chick-food items, although there was no apparent relationship between chick-food abundance and herbicide use. The survival of chicks in the nest and chick weight were both related in turn to the availability of food close to the nest, although in this species it has not been possible to investigate the relationship between chick weight at fledging and subsequent survival (Brickle *et al.* in press). Population modelling using limited existing data has indicated that the decline in the Sussex study area may be linked to low nest survival rates as well as to reduced frequency of breeding attempts during the year - the latter being less obviously related to factors associated with pesticide use. Winter food availability may, however, also be a limiting factor in this species (Brickle 1999).

In the case of the Skylark, which shares a similar diet at the nestling stage with other declining farmland bird species, clutch size is greater and chick body condition is improved in nests in set-aside fields compared with cereals or grassland (Donald & Vickery 2000). However, even if this difference is due to prey differences (which may in turn be linked to pesticide use), it appears unlikely that these are significant factors in determining the productivity and population size of this species. Predation plays a major role in reducing breeding success in set-aside, but the higher breeding density in that habitat, which is considered to be a function of vegetation structure rather than food abundance *per se*, probably compensates for the lower predation rate experienced in cereals (Donald & Vickery 2000). On present evidence it seems that population declines are unlikely to be due to reductions in chick-food availability. Indeed, recent evidence suggests that foraging locations may be selected on the basis of vegetation structure, even though these contain lower prey densities than the main crop (Odderskaer *et al.* 1997). However, the importance of differences in chick diet between set-aside and cereal crops has not been fully evaluated (Poulsen *et al.* 1998). Moreover, the strong preference of this species for feeding in cereal stubble in winter, where a greater

diversity of food exists, than in winter cereals, suggests that the significance of pesticide effects on food availability in winter cannot be discounted.

Preliminary studies on the Linnet (Moorcroft & Wilson 2000) suggest that breeding productivity of the species in Oxfordshire is currently sufficient to maintain population levels given recent estimates of survival rates made by Siriwardena *et al.* (1999). This finding accords with recent Common Birds Census trends, which show the species recovering from a long-term decline that ended in the late 1980s (Siriwardena *et al.* 1998a). That availability of food is an important factor determining breeding success is suggested by recent evidence that fledging brood size may be higher in pairs nesting closer to oilseed rape crops, and that oilseed rape now appears to be an important nestling food source (Moorcroft *et al.* 1997). Further evidence for a possible role of pesticides in restricting food availability for this species during the breeding season is not available, but it seems probable that oilseed rape has replaced weed seeds in the diet, many of which are likely to be controlled by herbicide use.

Evidence for a possible mechanism for the operation of indirect effects in the case of Tree Sparrow (Hart *et al.* 1992) was discussed previously, but there is little work on this species to show the population significance of any changes in reproductive performance and in particular in chick growth rates.

Work on the Yellowhammer, currently underway in the UK and described elsewhere in this volume, supports the view that many cropped habitats now contain suboptimal food sources and are therefore avoided. Winter wheat is apparently a preferred crop type on organically farmed areas, but often avoided elsewhere, although the habit of this species of taking unripe grain as food for nestlings complicates any interpretation here of the role of pesticides and invertebrate prey availability (Stoate *et al.* 1998). Studies in Denmark suggest that factors associated with organic crop production affect individual breeding performance, with greater clutch sizes in organic compared with conventionally grown crops (4.17 eggs per clutch compared with 3.64). Moreover, birds tended to forage further in conventional systems and there is some evidence for reduced use of fields following insecticide treatment (Petersen *et al.* 1995). However, the absence of any apparent link between pesticide use and breeding parameters other than clutch size, and the lack of an obvious mechanism for reduction in clutch size, throws some doubt on the primacy of any role of indirect effects of pesticides on this species during the breeding season. As with Skylark, its strong association with stubble fields in winter suggests that the significance of grain and weed-seed availability at that time, and consequently the potential role of herbicide use, should be considered. Some evidence based on ring-recovery data suggests that there

may have been a reduction in Yellowhammer survival rates during the recent period of population decline (Siriwardena *et al.* 1998b), consistent with a reduction in food availability over winter.

A number of other species are currently the subject of studies that tend to illustrate again the importance of recent changes in food availability on foraging behaviour and number of broods raised, for example Turtle Dove *Streptopelia turtur* (Browne *et al.* 1999) and Song Thrush *Turdus philomelos* (Thomson & Cotton 2000). In others, breeding success has apparently not been adversely affected and factors other than summer food availability are thought to be important (e.g. House Sparrow, Hole 1998). Overall, the detailed autecological studies tend to support the conclusion of Siriwardena *et al.* (1998a) that a range of factors associated with farming intensification are responsible for declines. Changes in breeding success that may be linked to food availability and in turn to pesticide use (along with other factors) are evident for a number of species. We still lack in most cases a clear understanding of whether observed changes in population parameters are likely to be key factors in determining population size. Further population modelling along the lines of that developed for Grey Partridge and Corn Bunting is needed, and one focus of the present tranche of autecological studies should be to provide the relevant population dynamics data to enable such models to be developed.

Indirect effects of pesticides - evidence from field experiments

As mentioned above, the most compelling evidence for the role of the indirect effects of pesticides in the decline of the Grey Partridge comes from studies of its population recoveries in conservation headlands, achieved by manipulating pesticide use alone within an otherwise conventionally grown cereal crop. There have been disappointingly few studies of the significance of conservation headlands for populations of other farmland bird species. Pheasant chick survival is apparently greater in conservation headlands (Rands 1986), Yellowhammers are known to use such areas preferentially for foraging (Stoate *et al.* 1998) and numbers of visits made by Blue-headed Wagtails *Motacilla flava*, although not by Skylarks and Meadow Pipits *Anthus pratensis*, were increased in unsprayed crop edges (de Snoo *et al.* 1994). However, other hedgerow-dependent bird species appear to be adversely affected by the habitat conditions created by conservation headlands, and numbers of Greenfinches, Robins *Erithacus rubecula* and Song Thrushes were lower in hedges adjacent to such habitats (Green *et al.* 1994).

Other recent large-scale field studies that provide evidence for the possible significance of the indirect effects of pesticides for birds fall into one of three categories:

comparisons of organic with conventional farming systems, experimental comparison of different farming systems where pesticide effects alone are investigated (for example the MAFF Boxworth Project and SCARAB (Seeking Confirmation About Results at Boxworth) project), and farming system studies, where pesticide use is only one of many variables manipulated and so their effects cannot be considered in isolation (for example the LINK Integrated Farming Systems (IFS) project and similar studies in the Netherlands and Germany).

A number of studies have provided evidence that organic farms tend to hold higher densities of birds than conventional farms, including paired-farm comparisons in the UK (Chamberlain *et al.* 1999) and in Denmark (Petersen *et al.* 1995). However, organic farming practices differ not only in crop management but also in the physical management of the farm (e.g. size of fields and hedge structure). Where non-crop variables such as availability of non-crop habitat and proximity to woodland have been controlled for, some major differences in bird densities remain. For example, density of breeding Skylarks was higher on organic farms (Wilson *et al.* 1997), and a number of bird species show strong or regular associations with organic management, particularly seed-eaters such as Tree Sparrow, Bullfinch *Pyrrhula pyrrhula*, Greenfinch and Reed Bunting *Emberiza schoeniclus* (Chamberlain *et al.* 1999). Similarly, the number of weed species and weed densities have been shown to be higher in organic crops, as have densities of certain invertebrates (although this has been less clearly the case for gamebird chick foods, Moreby *et al.* 1994). However, as Chamberlain *et al.* (1999) have warned, it is impossible to draw any inferences about the likely impacts on bird populations on organic farms of the absence of synthetic pesticides and artificial fertilisers alone, given the sometimes large differences in associated environmental variables. Such an understanding can only be achieved through large-scale experimental manipulation of those inputs alone.

The MAFF Boxworth Project (Greig-Smith *et al.* 1992) and the subsequent SCARAB study (described in Cooper 1990) both provide large-scale studies of contrasting pesticide regimes. The Boxworth Project carried out at a single site in East Anglia clearly showed major effects on invertebrate densities, including species now known to form an important part of the diet of several declining farmland bird species, under high ('insurance') pesticide regimes (Vickerman 1992). Similar effects were not seen for weed densities, and it has been suggested that this is because the lower levels of herbicide use achievable under 'integrated' and 'supervised' farming systems, together with a weed flora that had 'adapted' to a high-input regime, were not sufficient to enable any significant change in weed densities to be measurable (Marshall 1992). Similarly, few consistent differences in invertebrate

numbers were found between 'integrated' and 'supervised' (roughly equating to modern 'conventional') treatments in the MAFF Boxworth Project (Vickerman 1992). The scale of the Boxworth study was insufficiently large to overcome possible interference between treatments and confidently to determine effects on densities of breeding birds.

In the SCARAB study, carried out on 4-16 ha fields at a range of sites, average pesticide reductions of 50% were achieved between conventional and reduced-input treatments. Again, the scale of these trials was too small to expect measurable differences in bird densities, but responses of invertebrate and weed densities to differences in pesticide use were measured. Long-term effects on invertebrates were evident only for Collembola - a very poorly dispersive group - and whilst many other arthropod taxa were temporarily adversely affected by insecticide applications, their numbers subsequently recovered (Frampton in press). Although weed densities in the parallel TALISMAN (Towards a Low Input System for Managing Agrochemicals and Nutrients) study were consistently lower in fields that had received full rather than a reduced herbicide input, there was great variability in the weed densities between fields. By the end of the six-year experiment, summer weed densities in conventionally-treated fields (standard rotation) ranged from about 7-65 m², those in reduced-input (alternative rotation) areas ranged from 44-309 m² (Jones 1997). There is insufficient information on the quantitative feeding requirement of farmland birds from which to judge whether such differences are likely to have any ecological significance for bird populations. However, it should be borne in mind that even a temporary reduction or elimination of a prey type (such as occurred in the SCARAB trials after insecticide applications) at a critical time of year has potential significance for bird feeding requirements. Such critical periods are likely to include short-term reductions in invertebrate prey availability during brood rearing (especially for nidifugous species, or when foraging distances are curtailed by the need to provision nestlings) or during the post-fledging period when juveniles are inexperienced foragers. Similarly, short-term impacts may be important when the time available for foraging is restricted (e.g. short daylength in winter) or when weed densities are reduced at a time of year when seed return is most significantly affected. Absence of any demonstrable long-term differences in arthropod abundance in these studies is not of itself evidence for the absence of a mechanism for the indirect effects of pesticides.

A further problem with large-scale long-term studies is that without a well-documented understanding of the ecological status of such trial areas prior to introducing different treatment regimes, it is difficult to interpret the ecological significance of any population response (or its

absence). In the SCARAB project, as in other farming studies such as the LINK IFS studies, the practice has been to contrast a continuation of a 'conventional' pesticide regime with a reduced-input regime. It may be unlikely, depending on the length of time that fields received 'conventional' treatment prior to the study, that any further declines in populations will be observed, although recovery might be expected where pesticide treatments are relaxed. Evidence suggests that recovery from long-term decline is likely to be very slow (Aebischer 1990, Frampton in press) and will depend, amongst other things, on the practices in surrounding farmland. On this basis, it is perhaps not surprising that SCARAB found relatively few long-term consequences for arthropods when a reduced-pesticide regime was imposed on a 'conventional' background, whereas the Boxworth project found widespread effects when a more intensive pesticide regime than conventional was introduced. As a result, neither study is able to shed much light on the possible historic role of indirect effects of pesticides in the decline of farmland birds.

Other evidence for the possible role of pesticides may be derived from the recently completed LINK IFS project. In this case, pesticide effects cannot be separated from other treatment differences. These aim to show the environmental, economic and practical consequences of adopting an IFS approach (based on a range of management practices including variations in crop rotations, cultivar selection and husbandry practices) compared with conventional management of arable crops (Fisher 1998). In an initial analysis of this major programme, Holland *et al.* (1998) have concluded that there was no indication that invertebrate numbers and diversity increased in the integrated compared to conventional crops over time. They also point out that insecticide use was infrequent in both treatment regimes. In certain arthropod groups, for example the money spiders Linyphiidae, consistently lower numbers were found in conventional compared with integrated winter wheat crops. Such an effect was not found for other groups such as Carabidae. Holland *et al.* (1998) conclude that further measures - beyond those introduced under the IFS regime - will be needed if arthropods are to be encouraged within arable fields. However, the focus of that analysis was on arthropods as natural enemies of crop pests, rather than potential bird prey items, and information on other invertebrate groups is not considered there.

CONCLUSION - FUTURE REQUIREMENTS

It is clear that, although there has been a reduction in the frequency of incidents of pesticide poisoning of birds, the

risks arising from the direct effects of pesticide use cannot be ignored. Present methods of surveillance are biased towards larger species, and the nature of the interaction between indirect effects (hunger) and risks of direct lethal effects, particularly from treated seeds, is becoming apparent. In order to refine present risk assessments, we need to know whether risks are likely to be significantly greater than predicted (and the present autecological studies, especially of feeding requirements overwinter may help here) or less than predicted because of the type of factors indicated in the orchard studies outlined previously. There are certainly some specific areas where particular vigilance over the risks of direct effects is necessary, and the particular risks associated with rodenticide use have been highlighted here.

Evidence is mounting for the existence of mechanisms for the operation of indirect effects of pesticides in several declining farmland bird species in addition to the Grey Partridge, although it is clear that for some species other factors linked to intensification are likely to be of greater importance. At this stage, it appears that three main issues need to be tackled:

- (a) What further evidence is needed to identify the role of indirect effects? The present array of autecological studies will fill many of the gaps in understanding the possible mechanisms for the action of indirect effects. There is a need now for the further development of models to demonstrate and enable the testing of the likely population responses to changes in, for example, breeding parameters. Experimental studies to test the effects of pesticide manipulation are unlikely to be difficult to carry out at the scale required to investigate population responses of farmland bird species. However, studies based on pesticide reductions in conservation headlands may show the way for such investigations, having yielded data on changes in breeding parameters both rapidly and over a manageable scale of investigation.
- (b) If pesticides are a major factor in farmland bird declines through their indirect effects, how could or should this be taken into account in the regulatory process? Indirect effects may operate through effects on prey that may be either target or non-target species of the pesticide applications. In the case of target organisms, at least, requirements to reduce the severity of indirect effects are likely to conflict with efficacy requirements for the pesticide. Ways in which such indirect effects might be tackled through the regulatory process for pesticides are currently the subject of a MAFF research contract. It is likely that any approach to bringing indirect effects within the regulatory regime would reinforce the need for extending the monitoring of pesticide effects (under WIIS) beyond the present limited range of vertebrates and bees, to include important food resources including other invertebrates and non-crop flora.
- (c) What part should manipulation of pesticide use play in reversing declines of farmland birds? In most cases it seems probable that indirect effects are likely to be only one of several factors that have contributed to farmland bird declines. In the case of the Skylark, for example, crop cultivation timing and variety are crucial factors, and in the case of the Corn Bunting other attributes of mixed farming seem to be important.

Approaches towards reversing such declines are described in more detail elsewhere in this volume, but there are broadly three main ways in which the pesticide component can be addressed:

1. Adoption of low-intensity mixed farming systems, characterised by an organic farming approach. The potential benefits of this approach have been discussed previously in this paper.
2. Greater uptake of “artificially designed” systems to reduce pesticide use within targeted areas of the farm. These usually lie outside the main production function of the farming system and generally rely on some form of financial support. Examples are the pilot Arable Stewardship scheme, conservation headland, gamebird cover, certain tiers of some Environmentally Sensitive Area schemes, and set-aside.
3. Integrated farming systems, which claim to deliver environmental as well as economic benefits within conventional farming systems. Such approaches appear to be compatible with the agronomic and economic requirements of conventional farming systems, and form a key part of recommendations for achieving pesticide minimisation from the Pesticides Forum (Anon. 1999). However it is far from clear whether they will deliver the levels of increase in biodiversity (including invertebrates and non-crop biomass) required to reverse farmland bird declines.

The absence of measurable effects in many studies of integrated or reduced pesticide farming systems may be attributed to problems of scale when dealing with relatively mobile groups such as many invertebrates of arable systems. If, however, these are real effects, then one conclusion must be that IFS systems appear unlikely to offer real opportunities for recovery of farmland invertebrates unless other factors are incorporated in addition to manipulating the cropping regime and husbandry techniques. An alternative explanation is that many of such experimental studies are now carried out against a background of many years of intensive production to which within-field communities have become adapted, with the result that there are few

reservoirs from which recolonisation of 'reduced-input' systems can take place (e.g. Marshall 1992, Fuller 2000). This is not universally the case, however, as evidenced by the rapid increase in biodiversity in conservation headlands (at least on certain soil types) within a single cropping season (Rands 1985).

Several studies of IFS or reduced-input systems have reported increases in weed densities and biomass where herbicide inputs have been reduced (e.g. Jones 1997). In the case of the LINK IFS trials, changes in weed density at one of the study sites were considered to have resulted in increases in the densities of some invertebrate taxa (Holland *et al.* 1998). However, over the relatively short term of such studies, increases in weed densities have been variable and often fairly modest. Clearly, weed densities will be required to remain within agronomically acceptable limits within IFS, and it is not clear whether the weed levels that can be achieved (and are likely to be tolerated) within IFS are likely to approach those required by farmland birds. Wilson (1998) has drawn attention to the need for research to determine whether there are agronomically acceptable levels at which flowering and seeding weed genera favoured by farmland birds can co-exist alongside arable production, and to identify crop husbandry techniques to facilitate this co-existence of weeds and crop plants. If the adoption of IFS is to play a role in reversing declines of farmland bird species, further evidence is needed of ways of implementing such systems to enhance invertebrate densities, and there is an urgent need for studies of the levels of weed abundance required to attract and support populations of farmland birds.

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