A review of Indirect Effects of Pesticides on Birds and mitigating land-management practices

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Executive Summary

Many species of farmland bird have shown huge declines in numbers and range over the past four decades. These have been linked to agricultural intensification, which has taken the form of a suite of changes in farmland practice. One of these is increased use of pesticides. Concern has switched from the direct lethal or sublethal effects of pesticides on birds, such as declines in sparrowhawks due to decreased eggshell thickness resulting from use of organochlorine insecticide seed treatments in the 1950s and 1960s, to the indirect effects of pesticides.

These indirect effects act predominantly via reduction in food supplies. As well as reducing numbers of target invertebrates and weeds, insecticides and herbicides can reduce availability of non-target and beneficial species. This is predominantly due to insecticide use causing decreased abundance of insect food, and herbicide use causing decreased weed seed abundance, and decreased insect abundance, due to loss of host plants.

The grey partridge is still the only species for which population level indirect effects of pesticides been demonstrated. Insecticide and herbicide use have reduced chick survival, due to decreased invertebrate availability, to a sufficient extent to cause population declines. Immediate effects of pesticides on corn bunting insect food, and of insect food on chick mass and nest survival, have been found, but population modelling has not been conducted. Effects of breeding season insecticide applications on yellowhammer chick condition and brood survival have been shown, but population modelling suggests that these may have a relatively small effect on the overall population, with over-winter survival being the current limiting factor for this species. Evidence of effects for skylark remain ambiguous; organic farms support higher densities of skylarks, but also appear to be associated with higher nest failure rates. There is some evidence of effects of pesticides on chick food, number of nesting attempts, and chick diet and survival, but these often only occurred during poor weather. For barn swallow, pesticides do not seem to affect aerial invertebrate food or foraging behaviour, but as sample sizes are low further research would be needed to be sure this was not due to lack of power.

Most research has looked at effects of pesticides via breeding season insect abundance. However, a recent study showed that over-winter stubbles which followed on from a low-input spring cereal supported higher densities of cirl bunting, yellowhammer and reed bunting.

Amongst farmland birds, granivorous species have shown the worst declines. These species usually eat seed food in winter, but feed their chicks insect food in summer, when they themselves also eat invertebrates. Thus, discussion of food for farmland birds often focuses on summer insect food and winter seed food. However, two species, turtle dove and linnet, are both entirely granivorous, and the diet of both species has changed dramatically since the onset of agricultural intensification, switching from predominantly weed seeds to cereal seeds. For linnet, it seems that

oil seed rape seed availability has compensated for loss of weed seed in the landscape, as populations are now recovering. Turtle dove populations, however, continue to decline, with number of nesting attempts per pair having roughly halved since the onset of agricultural intensification. Investigation of effects of herbicides would be valuable for this species.

Another species that may be affected by pesticides, but for which research has not been conducted, is the yellow wagtail. The preferred nesting habitat of yellow wagtails switches through the breeding season from winter wheat to potatoes, which receive a relatively high number of pesticide applications.

Most research into the indirect effects of pesticides on farmland birds relates to short-term effects of pesticides, for example studies of the effects of breeding season insecticide applications on productivity, or of herbicide applications on the preceding cereal crop on use of an over-winter stubble. However, it should also be remembered that the widespread introduction of pesticide use is considered to have caused large-scale losses of seed and invertebrate food over time that will have affected many species of farmland bird. Such long-term effects are difficult to demonstrate, with the best evidence for them coming from the GWCT's study of the grey partridge on the Sussex Downs.

Effects of pesticides can be reduced by careful consideration of spraying practice, and also by providing alternative food rich habitats on the farm through agri-environment schemes. Options that may be particularly beneficial in reducing the effects of pesticides on farmland birds include conservation headlands, margins and buffer strips, wild bird seed mixtures, beetle banks, fallow plots, over-winter stubbles, skylark plots and undersown spring cereals. Specific options available in the different UK countries vary, and are discussed, along with uptake of the various options. Quantification of how much of different options is desirable is discussed, but has rarely been conducted, and further quantification of this, and of the ideal balance of different options would be valuable.

Contents

1. Introduction ... 6
  1.1. Farmland invertebrate declines ... 8
    1.1.1 Effects of insecticides on invertebrate abundance ... 10
    1.1.2 Effects of herbicides on invertebrate abundance ... 10
    1.1.3 Effects of fungicides on invertebrate abundance ... 11
  1.2 Farmland plant declines and effects of herbicides on plant and weed abundance ... 11
  1.3 Effects of pesticides on farmland birds ... 12
  1.4 Measures that may reduce the effects of pesticides on farmland birds ... 13
    1.4.1 Agri-environment schemes ... 13
    1.4.2 Set-aside ... 14
2. Methods ... 15
3. Results ... 15
  3.1 Species Accounts ... 15
    3.1.1 Grey Partridge ... 15
    3.1.2 Corn Bunting ... 17
    3.1.3 Skylark ... 19
    3.1.4 Yellowhammer ... 20
    3.1.5 Cirl Bunting ... 23
    3.1.6 Turtle Dove ... 25
    3.1.7 Linnet ... 25
    3.1.8 Yellow wagtail ... 26
    3.1.9 Barn Swallow ... 27
  3.2 Agri-environment prescriptions likely to compensate for effects of pesticides ... 28
    3.2.1 Conservation headlands ... 28
    3.2.2 Margins and Buffer Strips ... 29
    3.2.3 Uncropped margins ... 30
    3.2.4 Wild bird seed mixtures ... 31
    3.2.5 Beetle banks ... 31
    3.2.6 Fallow plots ... 31
    3.2.7 Over-wintered stubbles ... 32
    3.2.8 Skylark plots ... 33
    3.2.9 Undersown spring cereals ... 34
  3.3 Scale of management required ... 34
4. Discussion ... 36
  Further research ... 40
  Recommendations ... 40
Acknowledgements ... 41
References ... 41
Table 1. Uptake of agri-environment options likely to compensate for the effects of pesticides ... 55
  Scottish Rural Development Programme ... 58
Table 2. Effects of pesticides on farmland birds ... 62
Table 3. Quantification of the amount of some food rich habitats required to ... 63
achieve stable populations for farmland bird species ... 63
Appendix ... 66
Table 1. Scientific names for bird species included in text ... 66

1. Introduction

Many species of farmland bird have shown alarming declines in numbers and/or range over the past four decades (Baillie et al., 1997, Fuller et al., 1995, Marchant and Gregory, 1994, Siriwardena et al., 1998), and these declines have been attributed to changes in farming practice (Chamberlain et al., 2000). Agriculture has become increasingly intensive in the UK since the Second World War, and particularly since 1973, when the UK joined the EC (Donald et al., 2002). This intensification has taken place as a suite of changes in farming practice, such as loss of mixed farming, the switch from hay to silage, the switch from spring to autumn sowing of cereals and associated loss of over-winter stubbles, increased agrochemical input and loss of unfarmed structures such as hedgerows and ponds (Evans et al., 1995, O'Connor and Shrub, 1986). This review focuses on the effects of increases in pesticide applications on farmland bird populations.

Concerns over the effects of pesticides on birds first arose in the 1950s and 1960s, when use of organochlorine insecticide seed treatments (DDT and dieldrin) decreased eggshell thickness and thus productivity in sparrowhawks (see Appendix, Table 1 for scientific names of species in text), resulting in a population decline (Newton, 1995). Organochlorine seed treatments also caused a population decline in stock doves (see Burn, 2000 for review), but following restrictions in the use of these products, populations of both species have recovered (Newton, 1986, Marchant et al., 1990). Subsequently, in the 1970s and early 1980s, use of an organophosphate seed treatment led to mortalities of greylag and pink-footed geese, resulting in pink-footed geese fatalities in the UK in 1975 equivalent to over 1 % of the global population (Greig-Smith, 1994). The product has since been replaced. It is possible that the use of products such as rodenticides, molluscicides and insecticides may occasionally have unintentional side-effects on smaller passerine species. Such effects may be under-estimated due to the lower probability of finding casualties. The possible effects of rodenticides on bird species, particularly those that have close associations with humans, such as barn owls and red kites, require continued monitoring.

Sublethal effects, where pesticides affect the behaviour or physiology of individuals, are widely documented, however evidence of population level effects is rare (Burn, 2000). The decline of the sparrowhawk due to decreased eggshell thickness discussed above is one such example, and population level effects have also been found for white-tailed swallows following use of organophosphate insecticides during forestry operations in Canada, due to changes in parental behaviour (Busby et al., 1990). Sublethal effects of pesticides that inhibit the neurotransmitter hydrolysing enzyme acetylcholinesterase (AChE), have been found for many species, affecting a wide range of behavioural and physiological functions (for review see Burn, 2000). A study of tree sparrows at Boxworth found that, following application of an organophosphate aphicide, chicks were fed a higher proportion of aphids and fewer ground beetles. Exposed chicks exhibited depressed cholinesterase activity and had a slower development rate (Hart et al., 1992). However, the study could not show for certain that these changes were due to sublethal effects of the aphicide, rather than the dietary shift, and subsequent population trends were not monitored. Generally, recent studies of changes in reproduction or survival following sublethal exposure to AChE inhibitors have found few, if any, effects (Bennett, 1994, Grue et al., 1997). Thus, overall, there is little evidence that direct lethal and sub-lethal effects of pesticides are having significant population level impacts in the UK at present (Burn, 2000).

In recent decades, concern has switched from these 'direct' i.e. lethal or sublethal effects of pesticides on survival or productivity to 'indirect effects', i.e. effects via loss or reduction in suitability of nesting habitat or reduction of summer or winter food resources. Whilst it is possible that widespread use of glyphosate to remove the green cover of set-aside may have implications for the breeding success of ground-nesting species, such as skylark, due to reduced nest-concealment, effects of pesticides on nesting habitat availability is generally considered relatively minimal. This review focuses on the effects of loss of summer and winter bird food.

Amongst farmland bird species, granivorous passerines have shown the worst declines (Fuller et al., 1995). However, most of these granivorous passerines are partially insectivorous, at least during the breeding season, when they also rely on invertebrates as a source of high-protein chick food (Baillie et al., 1997). There is also evidence that amongst these declining granivorous passerines, those that are more dependent on insects have shown significantly worse declines (Wilson et al., 1999). Thus, for declining farmland bird species, discussions of abundance of 'summer food' are often assumed to refer mainly to abundance of invertebrates, and 'winter food' to abundance of seeds. Declines in food abundance may affect farmland bird populations via productivity and/or survival.

Although grassland accounts for the largest area of agricultural land in the UK (64 % of all agricultural holdings by area, Defra, 2008), pesticide inputs on grassland are extremely low, with most permanent pasture and rough grazing receiving no pesticides at all (Pesticide Forum, 2006). Arable land accounts for 25 %, and winter wheat for 11 %, of UK farmland (Defra, 2008), and the most widespread effects of pesticide use are due to applications made to arable land. Horticulture only accounts for a very low proportion (1 %) of the farmed area in comparison to arable crops (Defra, 2008), but pesticide usage can be intense on some crops, particularly orchards, hops and some vegetables, driven in part by the requirement for cosmetically perfect produce with long shelf life (Pesticide Forum, 2006). However, studies of farmland birds on horticultural crops are lacking.

There has been increased reliance on pesticides in arable farming since the 1950s, with both total area treated and number of applications per year having increased since the 1950s (Campbell et al., 1997, Chamberlain et al., 1999). Although pesticide use can also be measured in terms of mass applied, this is not a suitable way to look at trends over time, as changes in active substances mean that the amount that needs to be applied has changed. Methods of application have improved over time, reducing factors such as drift into adjacent habitats and quantity of chemicals reaching watercourses, but the proportion of the cropped area sprayed has shown dramatic increases over the past fifty years. Widespread use of herbicides to control arable weeds began in the late 1940s, and areas sprayed in England and Wales have more than doubled since the late 1970s. Use of foliar fungicides to control mildew and rusts began in the mid-1970s, and increased steadily for 20 years such that the area sprayed is now similar to that for herbicides. There was some use of insecticide seed treatments and organochlorines in cereal crops in the 1940s, but widespread insecticide use to control aphids did not really begin until the late 1960s. Although at a lower level of application than herbicides and fungicides, insecticide use has increased markedly during the 1980s and 1990s, as has use of molluscicides. For a review of pesticide use, see Campbell et al., 1997. Pesticide use has also been associated with other changes in farming practices, for example by facilitating autumn sowing of cereals and removing the need to control diseases by traditional rotations, but these are not considered in detail here.

In more recent years, the area of arable crops treated with pesticide increased by 6 % between 1996 and 2006, although this can be partly explained by a 3 % increase in the area of arable crops grown during this period (Garthwaite et al., 2006). The remaining difference reflects an increase in average number of sprays applied from four in 1996 to over five in 2006 (Garthwaite et al., 2006). The weight of pesticides applied decreased by 38 % during this time, due to increased effectiveness or targeting of pesticides (Garthwaite et al., 2006). The area treated with insecticides decreased by 5 % during this period (weight applied decreased by 3 %), whilst area treated with herbicides increased by 9 % (but weight applied decreased by 10 %) and area treated with fungicides increased by 9 % (but weight applied decreased by 2 %) between 1996 and 2006 (Garthwaite et al., 2006).

Of the categories of pesticide, herbicides, insecticides, fungicides and molluscicides are considered to have had the most widespread effects on farmland birds and so are the main focus of this review. Rodenticides may affect food availability for birds of prey (e.g. barn owls), and avermectins (worming drugs) may reduce availability of dung-feeding invertebrates for birds such as chough and wagtails (Webb, 2004), but these effects are not considered here, as the review focuses on pesticides that are plant protection products.

Pesticides may affect birds in three main ways, referred to as type 1, 2 and 3 effects, defined below:

1. Reduced invertebrate abundance due to direct effects of insecticides.
2. Reduced invertebrate abundance due to indirect effects of herbicides via loss of host plants.
3. Reduced abundance of weed seeds due to direct effects of herbicides.

These effects could be additive, e.g. seed-eating passerines that feed their chicks on invertebrates may be susceptible to all types of effect.

1.1. Farmland invertebrate declines

A long-term study of invertebrate abundance on by the Game and Wildlife Conservation Trust (GWCT, formerly known as the Game Conservancy Trust) on over 100 cereal fields on the Sussex Downs found that, although different invertebrate groups have shown varying trends in recent decades, overall numbers of invertebrates, excluding Collembola, declined by about 50 % between 1970 and 1990 (Aebischer, 1991). They predicted that there has probably been a roughly 75 % decline in invertebrate abundance in cereal fields since the introduction of herbicides in the 1950s (Aebischer, 1991).

A review of research on farmland invertebrate abundance concluded that many species of invertebrates have been declining on farmland, with the exception of most aphid species (Sotherton and Self, 2000). Aphids do feature in the diet of some farmland bird species (Holland et al., 2006), but this may just represent the fact that they are more abundant, rather than selection for them. Their small size means that a high abundance may equate to little nutritional value, and for some species they do not represent a satisfactory substitute for preferred larger prey items, for example increased proportions of aphids in grey partridge diet affects chick growth and flight feather development (Borg and Toft, 2000). Other studies vary in their conclusions relating to trends in aphid numbers. Data from the Rothamsted insect survey, a network of suction traps in a range of habitats, suggests aphid populations have shown little marked change since the 1960s, with a few species having increased (Woiwod, 1991), whilst the Sussex Downs study, found that aphid numbers had decreased dramatically since the 1970s (Aebischer and Potts, 1990).

Data from a network of light traps have shown that macro Lepidoptera have decreased in numbers on farmland between the periods 1933-1950, and 1960-1989, with no similar decrease being found for woodland traps (Woiwod and Thomas, 1993). A study of ground beetle (Carabidae) diversity in a weedy arable plot in the Tyne Valley has also found a decreasing trend in species of ground beetles since 1981 (Luff, 1990), and similar Carabid declines have been shown across Europe (Luff and Woiwod, 1995, Kromp, 1999). Both common and localised butterfly species have decreased in frequency in pastoral north Wales between 1901 and 1997 (Cowley et al., 1999). Declines in butterfly populations have been reported elsewhere in Europe, although they seem less severe in southern Europe, which, despite pockets of intensive agriculture, generally has a less intensively managed landscape (Van Swaay, 1990, Pavlicek-van Beek et al., 1992). There have also been declines in many species of bumblebee *Bombus* species, particularly eastern and central England (Williams, 1986), and throughout Europe (Corbet et al., 1991).

Pesticide use is frequently suggested as the largest driver of these declines (Aebischer and Potts, 1990, Wilson et al., 1997, Moreby and Southway, 1999, Sotherton and Self, 2000). Other possible factors contributing to invertebrate declines are increased specialisation of farming, decreased undersowing, timing and depth of ploughing, and a reduction in the number of uncultivated field margins (Wilson et al., 1999, Sotherton and Self, 2000).

1.1.1 Effects of insecticides on invertebrate abundance

Many studies have demonstrated the potential impact that broad spectrum insecticides may have on non-target invertebrates, such as those eaten by birds, both in the short-term (Vickeram and Sunderland, 1977, Sotherton, 1990, Duffield and Aebischer, 1994,) and the long-term (Burn, 1989, Aebischer, 1990). The GWCT's Sussex study found that numbers of important chick food invertebrates at the site have declined since 1970 (Aebischer, 1990, Ewald and Aebischer, 1999), and these declines have correlated with increases in pesticide use (Ewald and Aebischer, 1999). Ewald and Aebischer (1999) found that densities of important non-target invertebrate taxa were lower in fields treated with insecticides, with spring/summer insecticide applications appearing to be the most damaging. Abundance of invertebrates also related to whether the field was sprayed the previous year (Ewald and Aebischer, 1999). This supported earlier work relating to long-term effects of pesticides on abundance of Symphata by Aebischer (1990). Aebischer (1990) found that, using predictions from models based on weather and proportion of undersown cereals, wide-scale insecticide treatments reduced numbers of sawfly to one-tenth of those predicted, and recovery was expected to take seven years.

Different active substances vary in how much they affect non-target invertebrates. For example, the carbamate insecticide pirimicarb has been found to have relatively little effect on non-target invertebrates that are important to farmland birds, whilst numbers in fields treated with either pyrethroid or organophosphate insecticides was much lower than in untreated fields, or fields treated with pirimicarb (Ewald and Aebischer, 1999). There was also some evidence that fields treated with organophosphates contained lower abundances of non-target invertebrates than those treated with pyrethroids.

1.1.2 Effects of herbicides on invertebrate abundance

Southwood and Cross (1969) estimated that disappearance of host food plants associated with the widespread use of herbicides from the 1940s would have reduced invertebrate numbers in cereal crops by half. Aebischer (1991) found that numbers of invertebrates, excluding Collembola, declined by a further 50 % between 1970 and 1990 on cereal fields in Sussex.

At the GWCT Sussex study site, densities of Chrysomelidae have decreased since the 1970s, and it is suggested this is probably a result of loss of host plants due to herbicide use (Potts, 1986). Sotherton et al. (1993) found that densities of Heteroptera were higher in conservation headlands, which receive only selective herbicide applications. Ewald and Aebischer (1999) also found an effect of spring/summer herbicide use and Chrysomelidae and Curculionidae densities, but no effect on non-aphid Hemiptera. Generally, Ewald and Aebischer (1999) found few relationships between herbicide use and abundances of non-target invertebrates at the Sussex site, although this could be because the study started in 1970, after the introduction of widespread herbicide use.

1.1.3 Effects of fungicides on invertebrate abundance

Sotherton and Moreby (1988) found that the foliar fungicide pyrazophos had insecticidal properties, but this has since been withdrawn. Research at the GWCT Sussex study site implicated foliar fungicides in a recent reduction in density of fungivorous beetles (Aebischer and Potts, 1990) and invertebrate abundances tended to have negative relationships with fungicide applications, particularly those of Aranae and Opilione (Ewald and Aebischer, 1999). Numbers of Aranae and Opilione, both polyphagous predators, decreased over the course of the Sussex study (Aebischer, 1991), and this correlated with increased use of fungicides and decreased levels of mildew and rusts (Aebischer and Potts, 1990). Links have been found between abundances of fungicidal beetles and fungal food availability, and it is possible such a relationship also exists between Aranae and Opilione and their invertebrate prey, some of which could be fungivorous. However, although Ewald and Aebischer (1999) attempted to control for insecticide applications in their models, it was difficult to separate effects of insecticides and fungicide applications, due to a strong correlation between the two, and further investigation of effects of fungicides on invertebrates would be valuable.

1.2 Farmland plant declines and effects of herbicides on plant and weed abundance

A review of the changes in abundance of farmland plants by Sotherton and Self (2000) concluded that, with the exception of some species of grass weeds and common broad-leaved weeds, many species are declining. Farmland vascular plants in the UK underwent huge declines in range and abundance last century and, as a result, farmland now supports a large number of scarce and rare species. Farmland is also important for a number of species of lower plants (liverworts and mosses).

The three main sources of large-scale population trend data for plants are the *Atlas of the British Flora* (Perring and Walters, 1990), the *British Red Data Book (Vascular Plants)* (Perring and Farrell, 1983) and *Scarce Plants in Britain* (Stewart et al., 1994). These all show that all of the vascular plant species, and most of the bryophyte species, for which farmland is important have suffered range contractions during the latter part of the last century. The GWCT's Wildflower Project indicated declines in almost all scarce, and many formerly widespread, species (Wilson, 1993).

These declines are due to a number of factors, but an increase in herbicide applications is likely to have been important, along with use of inorganic fertilisers, improved seed-cleaning techniques and the switch from spring to autumn sowing of cereals (Sotherton and Self, 2000).

Ewald and Aebischer (1999) suggest that although the effect on weed seed availability of the widespread use of herbicides to control arable weeds has not been quantified, it is likely to have been considerable. Research at the GWCT Sussex study site found that the occurrence of important dietary broad-leaved weeds generally increased over the study period (Ewald and Aebischer, 1999). Timing of herbicide applications was the most important factor affecting occurrence of these species, with summer applications resulting in lower occurrences. These results differ from some of the studies of changes in weed occurrence presented previously for example in Campbell et al.'s (1997) previous review of the indirect effects of pesticides. However, some of the studies cited sampled weeds either before or during herbicide applications (e.g. Chancellor, 1985, Whitehead and Wright, 1989), which may not accurately reflect what is available to farmland birds. Fryer and Chancellor's (1970a, b) work was based on changes that took place before 1970, whereas Ewald and Aebischer's (1999) study only began in 1970, after the use of herbicides had already become widespread. It is also important to note that studies reviewed in this paragraph are looking at occurrence, and not abundance, of weed taxa, and that Ewald and Aebischer (1999) took samples in June. Ideally, to investigate trends in weed seed availability for farmland birds, winter densities of seeds would be taken, however this is very time-consuming. Abundance of broad-leaved weeds generally was sampled at Sussex, and this was negatively related to use of dicotyledon-specific herbicides, and abundance of grass weeds was negatively related to broad-spectrum herbicides (Ewald and Aebischer, 1999).

1.3 Effects of pesticides on farmland birds

Evidence of an indirect effect of pesticides on a farmland bird population first came from a large-scale 30-year study of grey partridge by the GWCT. The study was conducted at a 62 km^2 study site on the Sussex Downs, and found that reduced chick survival due to reductions in chick food invertebrates due to direct effects of insecticides, but particularly indirect effects of herbicides via loss of host plants, was causing population declines. Since this study, evidence for indirect effects on yellowhammer chick condition and brood reduction (Boatman et al., 2004, Morris et al., 2005, Hart et al., 2006) and corn bunting chick survival (Brickle et al., 2000), although whether these exert population level effects is questionable in the yellowhammer (J Crocker, unpublished data), and not assessed for corn bunting.

Effects of herbicide on weed seed resources (type 3 effects) are suspected to have contributed to the declines of seed-eating birds, but although effects on densities in winter have been shown (Bradbury et al., in press), population level effects are yet to be demonstrated, partly because of analytical difficulties (Boatman et al., 2004).

Evidence of indirect effects of pesticides on birds via their food resources can be investigated through autoecological or experimental studies. Experimental manipulations of pesticide use are ideal as they remove the effects of confounding factors, however they are also more expensive and hence relatively rare. Following investigation of the effects of pesticides on demographic rates (e.g. productivity and survival), modelling to investigate whether these are likely to be of sufficient magnitude to have population level effects is ideal, but currently has been done for just a few species.

It is recognised that pesticides are part of modern farming, that environmental concerns must be balanced with the need for farming to be profitable and efficient, and that there is a trade-off between intensity of cultivation and the area cultivated. There are also incidences where pesticide use is actually environmentally beneficial, for example, use of selective graminicides in grass margins can increase the abundance of beetles and other insects, native wild flowers and foraging birds by suppressing the dominance of grasses within the sward to allow conditions that promote germination, mobility and access to food. However, generally pesticide use has led to declines in food resources for farmland birds.

1.4 Measures that may reduce the effects of pesticides on farmland birds

Measures to help reduce effects of pesticides need to be financially desirable, and can include approaches such as Integrated Crop Management (see recommendations), and careful consideration of when and what to spray, but also measures funded by agri-environment schemes to provide alternative food rich habitats on the farm.

1.4.1 Agri-environment schemes

Agri-environment schemes, where farmers implement management techniques that are beneficial to biodiversity, in return for payments based on costs incurred or profits forgone in implementing these techniques, were first introduced into the United Kingdom in 1987. These require management over and above the cross-compliance standards of Good Agricultural and Environmental Condition that became a condition of qualifying for the Single Payment Scheme under Common Agricultural Policy reforms in 2003 (the Fischler reforms). These reforms meant that farming subsidies were no longer based on production ('decoupling'), and certain minimum environmental, welfare and safety standards were made necessary for subsidies to be received (cross-compliance), such as establishment of uncultivated 1 m protective zones on either side of ditches or hedgerows.

Some agri-environment options provide payments for reductions in pesticide inputs, and others will provide alternative food-rich habitats for farmland birds, and are thus likely to reduce the indirect effects of pesticides. Such options are discussed later in the review, and specific prescriptions for each country in the United Kingdom are presented in Table 1. Effectiveness of these agri-environment scheme options generally, and in compensating for the effects of pesticides on farmland birds, will obviously depend on them being implemented on a sufficient scale. It is often the choice of the participating farmer which options to implement, and this results in some options being very widely implemented, with others being chosen rarely (Boatman et al., 2007). Uptake of the different recommended options will be discussed, and where available, information on the scale of uptake to stabilise populations of different farmland bird populations is presented.

Agri-environment schemes differ between the different UK countries. The first agri-environment schemes to be introduced were Environmentally Sensitive Areas in 1987, when payments were made available to land owners within five areas considered to be of particularly high landscape, wildlife or historic value. Following this, the Countryside Stewardship Scheme was made available to farmers outside ESAs in England in 1991. This initially lacked arable options, but a pilot Arable Stewardship scheme was introduced two regions of England in 1998 to address this (Bradbury and Allen, 2003). Various arable agri-environment options were evaluated, and successful ones incorporated into the Countryside Stewardship Scheme in 2002. All of these schemes were competitive, and open to limited numbers of farmers. The ‘Curry Report’ recommended introduction of a scheme available to all farmers, with relatively simple yet effective management options (Policy Commission on the Future of Farming and Food, 2002). In 2005, the ESA and CS schemes in England were replaced by Environmental Stewardship (Grice et al., 2007), a two tier scheme encompassing Entry Level Stewardship (ELS), a ‘broad and shallow’ scheme open to all farmers, and Higher Level Stewardship (HLS), a competitive scheme with targeted payments for more demanding management aimed at priority species and habitats. The flexibility of ELS is popular, in that applicants can choose from list of over 50 options in order to meet a required points threshold, upon which entry is guaranteed. Initially, following its launch, ELS encountered a number of problems with the application procedure, preventing some farmers from being able to enter and causing widespread frustration, but this procedure has since been adjusted (Howie, 2007).

In Wales, the ‘broad and shallow’ scheme is Tir Cynnal, also introduced in 2005. In order to qualify, farmers must have at least 5 % of semi-natural wildlife habitat on their holding, and if they do not they can choose from various habitat creation options. Tir Cynnal differs from ELS in that there is no list of management options, only one for habitat creation, where necessary. Tir Gofal has been available in Wales since 1999, and replaced the former ESA and Tir Cymen scheme, and this provides farmers with a list of management options, similar to those in ELS, to choose from. This scheme differs from ELS in that it is competitive rather than open to all farmers, however.

In Scotland, ESAs and the former Rural Stewardship Scheme (RSS) are in the process of being replaced by the Scottish Rural Development Programme (SRDP), which should be available later this year. Both schemes differ from the ELS in that they are competitive, but are similar in that they have a list of land management options to choose from.

Northern Ireland currently has the Environmentally Sensitive Area scheme and the Countryside Management Scheme, although options available in the two schemes are now identical. These are being replaced with the Northern Ireland Countryside Management Scheme (NICMS), to be launched in late spring/early summer of 2008. Again, these schemes are competitive rather than open to all.

The importance of evaluating agri-environment schemes was highlighted by Kleijn and Sutherland (2003). Creation of agri-environment schemes in the UK has been an iterative process, with lots of options and schemes having been evaluated and developed (see e.g. Ausden and Hirons, 2002, Bradbury and Stephens, 2004, reviews in Evans et al., 2002, Grice et al., 2004, Vickery et al., 2004). As well as evaluation of the effectiveness of measures, uptake on a sufficient scale of the right balance of options, correctly managed, is required and this is discussed later.

1.4.2 Set-aside

Set-aside was introduced in 1988 as a production control measure for arable crops, and became a condition of receipt of price support in 1992. Since then between 5 %

and 18 % of eligible arable land has been out of production each year. Although not introduced with the aim of enhancing biodiversity, there is evidence that set-aside supports increased abundances of weed seeds (Wilson, 1992, Draycott et al., 1997), broad-leaved plants (Wilson and Aebischer, 1995, Hansson and Fogelfors, 1998, Fowbert and Critchley, 1999) and invertebrates (Kennedy, 1992, Moreby and Aebischer, 1992, Sears, 1992), thus providing foraging habitat for a range of bird species, particularly in winter (Buckingham et al., 1999, Henderson et al., 2000, Henderson and Evans, 2002). The set-aside requirement for receipt of the Single Payment Scheme was fixed at 0 % as of 2007, greatly reducing the area of over-winter stubbles, and increasing the need to provide farmland birds with extra sources of winter seed food. The potential shortfall in food sources for farmland birds needs to be evaluated.

Mitigation for the indirect effects of pesticides requires better quantification of the relationships between pesticide application, abundance of food and effects on demographic rates e.g. productivity and over-winter survival. Such quantification has only been achieved for a few species. A review of the available information, and gaps in knowledge, for species for which indirect effects of pesticides are known or have been considered likely to occur is presented here. Likely effects of mitigating measures are also considered. Species on the lists suggested as possibly suffering indirect effects of pesticides by Campbell et al. (1997), Morris et al., (2001) and Boatman et al. (2004) are reviewed, plus any for which new evidence came to light during the course of the review.

2. Methods

Reviews were conducted for species suggested as possibly suffering indirect effects of pesticides by Campbell et al. (1997), Morris et al., (2001) and Boatman et al. (2004). Academic search engines were searched using the species' names, and relevant experts consulted.

3. Results

3.1 Species Accounts

Accounts of research into indirect effects of pesticides for a range of species is presented below. Research for these species is summarised in Table 2.

3.1.1 Grey Partridge

The grey partridge was the first species for which indirect effects of pesticides were demonstrated, and the only species for which population level effects have been shown. Grey partridges have been declining in Britain since 1945 (Potts, 1980), with the population declining by 88 % between 1970 and 2005, one of the largest of all farmland bird declines (Eaton et al., 2007). The GWCT conducted a long-term study of the grey partridge on 62 km^2 of the Sussex Downs. Blank et al. (1967) found that the key factor causing population changes in Hampshire was chick mortality, and went on to establish a relationship between chick survival and insect abundance (Southwood and Cross, 1969). Potts (1980) modelled data from several regions and found that invertebrate abundance in cereal crops in June was the main factor influencing chick survival. Rands (1985) conducted an experiment to investigate the effects of eliminating pesticides from a 6 m strip around the edge of cereal fields, the favoured foraging habitat of grey partridge broods (McCrow, 1980, Green, 1984), and the area containing highest abundances of invertebrates and weeds (Green, 1984). Mean brood size was significantly higher on plots with unsprayed headlands (sprayed = 2.15 +/- 0.52, unsprayed = 6.38 +/- 0.92). Assuming that the mean number of young hatching per successful nest is constant, chick survival rates were significantly higher on plots with unsprayed headlands. Abundance of chick food invertebrates was significantly higher in unsprayed headlands. Aebischer and Potts (1998) also found that intensive use of broad-spectrum insecticides over six summers in Sussex was associated with an approximately one-third reduction in chick survival.

Potts (1980) reported that average chick survival rate had declined in Britain since widespread application of herbicides began in the 1950s, and suggested this could be a major factor responsible for the decline of the grey partridge. Where headlands are sprayed, Rands (1985) found that brood sizes were insufficient to maintain the population. The GWCT's National Game Census revealed that mean chick survival rates declined from 49 % before the introduction of herbicides to 32 % once their use became widespread (Potts and Aebischer, 1995). However, Potts and Aebischer (1995) found that on a study site in Sussex where grey partridge densities had shown dramatic declines between 1968 and 1993, there was no corresponding decrease in chick survival. Models showed that a reduction in chick survival from 49 % to 32 % had little effect where predators were controlled, but caused population collapse where predator control was relaxed. They thus deduced that the result at the Sussex site was due to an early decrease in chick survival in the 1950s, which was initially mediated by a decrease in shooting, followed by relaxation of predator control in the 1960s and 1970s, leading populations to collapse (Potts and Aebischer, 1995). This situation has been mirrored throughout much of Britain, where predator control is not undertaken and numbers of predators have increased in the post-war years (Tapper, 1992).

This research has led to a number of farms and estates having stopped local declines of grey partridges, by compensating for the adverse impact of herbicides using techniques such as conservation headlands (Sotherton et al., 1993), although few have restored numbers. In 2002, the GWCT introduced a grey partridge recovery project at Royston, Hertfordshire, which aimed to restore grey partridge numbers (Aebischer et al., 2005). The demonstration area was 10 km^2 of arable farmland, and there was a similarly sized control area adjacent to this. A suite of management techniques was implemented, including predator control, year-round supplementary feeding, as well as tailored set-aside and agri-environment options (Aebischer, 2003). The project has been highly successful in increasing numbers of grey partridge, with spring densities increasing from 2.9 to 11.2 pairs per 100 ha, and autumn densities from 7.6 to 53.4 per 100 ha by 2005 on the demonstration farm, with no corresponding increases on the control farm (Aebischer et al., 2005). However, whilst the project demonstrates how to manage arable land in order to increase partridge populations, the value of individual agri-environment options to grey partridge cannot be evaluated, as all the management techniques were implemented simultaneously. Elsewhere, the population decline of the grey partridge continues, with a 37 % decline in numbers between 1994 and 2006 (Eaton et al., 2007).

3.1.2 Corn Bunting

The British corn bunting population declined by 89 % between 1970 and 2005 (Eaton et al., 2007), and this decline has frequently been attributed to decreased over-winter survival, possibly due to loss of stubbles (Donald et al., 1994, Donald and Evans, 1994, 1995, Siriwardena et al., 1999). However, no data are available on changes in survival. Nest record scheme data suggest that breeding success increased in parts of Britain during the population decline (Siriwardena et al., 1999), although problems during the breeding season may have contributed in other areas (Aebischer and Ward, 1997, Donald, 1997). A population collapse in Schleswig-Holstein, Germany, correlated with increased chick starvation (Busche, 1989). Aebischer and Ward (1997) found that density of breeding corn buntings on the Sussex Downs was higher in crops associated with low input farming than intensively managed crops and that it also positively correlated with Lepidoptera and Symphata larvae in cereal fields.

Brickle et al. (2000) extended this study to investigate relationships between pesticide use, chick food abundance, foraging behaviour and breeding success. The study was conducted on 10 km^2 of the Sussex Downs study site between 1995 and 1997. Provisioning adults selected grassy margins more than any other habitat within their foraging range, and these contained at least eight times as many invertebrates as the poorest habitats (intensively managed grass and winter wheat). Preferences for spring barley, unintensified grass and set-aside, and avoidance of winter wheat and intensive grassland was also found. Foraging areas contained higher abundances of chick food invertebrates (the four main corn bunting chick-food groups: Opiliones, Lepidoptera larvae, Symphata larvae and Orthoptera) than non-foraging areas. Chick food density was also negatively correlated with number of herbicide, fungicide and insecticide applications. When analyses were restricted to cereal fields (which accounted for 46 % of the study area but contained approximately 70 % of nests) chick food invertebrate density was negatively correlated with just number of insecticide applications. Where chick food invertebrate abundance close to the nest was low, parents foraged at a greater distance from the nest and foraging trips were of longer duration. Chick weight was positively correlated with abundance of chick food invertebrates, and negatively correlated with duration of foraging trips. Probability of chick survival was negatively correlated with abundance of chick food invertebrates, it was suggested this was due to increased predation. Boatman et al. (2004) went on to quantify this relationship between chick survival and invertebrate food abundance, using an index of invertebrates from samples within 115 m (one third of the maximum observed foraging distance, as opposed to the maximum 346 m radius foraging range used in Brickle et al.'s (2000) study). Thus, it would seem that increased pesticide use causes declines in chick food invertebrates for corn bunting, with implications for chick mass, and chick survival.

Data from the BTO's nest record scheme has suggested that breeding success has increased during the period of population decline. However, nest survival rates found by Brickle et al. (2000) were much lower than those given in Crick (1997) from nest record scheme data. Brood sizes and daily survival rates at Brickle et al.'s (2000) study site were similar to those found by a study in the North Uists (Hartley and Shepherd, 1997). It is possible that nest survival rates have collapsed during the 1990s as they did for reed bunting (Peach et al., 1999). However, the nest record scheme data could be unrepresentative if corn buntings may have become concentrated in higher quality breeding sites as the population declined (Rodenhouse et al., 1997).

Population modelling was not conducted by Brickle et al. (2000), and so it can not be said whether the immediate effects of pesticides on chick mass and survival have population level impacts, whether this is the case will depend on other demographic rates (e.g. post-fledging and over-winter survival). It is possible that reduced breeding success due to increased pesticide use has contributed to declines, or hampered population recovery, in the corn bunting, but estimates of post-fledgling survival and population modelling are necessary to test whether observed breeding success is high enough to maintain a stable population. The observed effects on chick condition could also be affecting post-fledging survival, or survival of chicks to the following breeding season, as these factors are linked for many species of passerines (Mock and Parker, 1998).

Ewald et al. (2002) looked at variation in corn bunting breeding densities in relation to pesticide use and food availability across the whole 62 km^2 Sussex Downs study site for the same three-year period as Brickle et al.'s (2000) study. Corn bunting densities were positively related to a chick food index (derived for grey partridge), and to abundance of Symphata and Lepidoptera. There were negative relationships with number of pesticide applications, but these were not significant. Symphata and Lepidoptera appear to be particularly sensitive to the use of pesticides (Aebischer, 1990, Ewald and Aebischer, 1999).

A special scheme was launched in 2001 by the RSPB, along with SNH and the Farming and Wildlife Advisory Group targeted at the core of the corn bunting's Scottish range (Perkins et al., 2008). This included options for grass margins, beetle banks, conservation headlands, extensively managed unsprayed spring cereals and turnips, spring cropping, unharvested crops, delayed spraying and/or topping of set-aside, and supplementary feeding. All, except the last two, of these options, were also available under the Rural Stewardship Scheme. On farms that were part of this Farmland Bird Lifeline scheme, corn bunting numbers showed no change between 2002 and 2004, whilst on non-scheme farms they declined by 43 %, highlighting the benefits of these options for corn bunting populations (Perkins et al., 2008). Preferences shown by foraging adults for set-aside mean that the species may be adversely impacted by the large scale loss of set-aside from the agricultural landscape.

3.1.3 Skylark

There have been a number of studies into the indirect effects of pesticides on skylarks, with varying results. There is some indication of differences in skylark densities and breeding success between organic and conventional farms (Wilson et al., 1997, Chamberlain et al., 1999, Neumann et al., 2007, Piha et al., 2007). Wilson et al. (1997) compared organic and conventional farms in south east England found that territory densities of skylark in organic cereals could be more than twice those in pesticide treated crops. All ten cases of apparent brood starvation occurred in nests in winter cereals, and nine of these were intensively managed crops. It was suggested that these results were probably associated with greater invertebrate abundance due to lack of agrochemical application. This is supported by the fact that conventional winter cereals were avoided by breeding adults when foraging (J D Wilson, unpublished, cited in Wilson et al., 1997). Chamberlain et al. (1999) compared 22 pairs of organic and conventional farms in England and Wales, during three breeding seasons, two autumns and two winters, and found that density of breeding skylark was significantly greater on organic farms in one of the three breeding seasons.

However, it is not possible to separate the effects of pesticides from other differences between organic and conventional farms. Organic farms differ from conventional farms not just in restrictions on agrochemical inputs, but also due to measures to control crop pests, such as return to traditional grass-arable rotations, growing different crops or planting dense hedges to harbour natural predators of crop pests. Chamberlain et al. (1999) suggested that the observed differences in bird densities were probably mainly attributable to structural differences between the two farm types. Intensive mechanical weeding of organic crops in the Netherlands has been shown to result in high nest failure rates, due to increased accidental destruction of nests (Kragten et al. in press).

Wilson et al. (1997) also found that densities were higher on set-aside relative to other fields. Nest success was also higher on set-aside and Poulsen et al. (1998) found a strong preference for set-aside by adults provisioning young. Set-aside is also strongly favoured by foraging skylarks outside the breeding season (Wilson et al., 1996). As with the organic/conventional farm comparisons, these differences may reflect relative abundance of invertebrate food in set-aside fields due to the more diverse vegetation structure and lower pesticide inputs, but effects of pesticides are difficult to separate. Clutch sizes are also higher in set-aside than cereals, and chick diet differs: in set-aside, parents bring their chicks a large number of soft-bodied invertebrate larvae, which are particularly susceptible to pesticides, whilst in sprayed cereals chicks are fed a higher proportion of pesticide-tolerant ground beetles. Despite these differences, chick growth rate or condition did not differ between cereals and set-aside, and higher predation rates in set-aside mean that productivity from nesting attempts in cereals is higher (Donald et al., 2001).

Boatman et al. (2004) found no significant effects of pesticides on skylark chick condition or growth rate, although sample sizes were small (condition: 33 broods, growth rate: 13 broods). There was an effect of invertebrate abundance on chick condition, despite a sample size of only 11 nests. Boatman et al. (2004) suggest that more research is needed before firm conclusions can be drawn about effects of pesticide use on skylarks.

Odderskaer et al. (1997) is the only example of experimental manipulation (albeit on a small scale) to investigate the effects of pesticides on skylark. Effects of breeding season pesticide (herbicide and insecticide) applications in spring barley were investigated using four fields in Denmark. No effect was found on the number or size of skylark territories in spring cereals, distance flown by parents to collect food for chicks, or chick mass. Invertebrate food abundance was generally highest in unsprayed fields, with differences being greatest just after insecticide application, when it was approximately three times higher in unsprayed fields. However, parents in unsprayed fields consistently produced more chicks. This was due to a higher rate of nest survival and greater number of nesting attempts on unsprayed fields. Number of fledglings per field per year was reduced at 38 % in sprayed fields, and number of hatched eggs was reduced at 17 % in sprayed fields. Probability of clutch survival was lower in sprayed fields. Many pairs gave up breeding on sprayed fields late in the season, but continued breeding on unsprayed fields. However, these effects were less strong than those of the weather; and effects were only found in poor weather.

Nestling diet was dominated by Carabids (mean = 42 %), with Lepidoptera (19 %) and Heteroptera (7 %) being the only other groups to account for more than 5 % of the total average dry weight of faeces samples (Odderskaer et al., 1997). Nestling diet differed between sprayed and unsprayed fields: prior to and up to 14 days after insecticide spraying there was no difference, but after that Carabids made up a larger proportion of the diet in sprayed fields, and Lepidoptera were more frequent in the diet in unsprayed fields. Insects on the unsprayed fields are all relatively soft bodied compared to Carabids, and this and may affect their quality as food items (Carabids being particularly chitinous). Lepidopteran larvae are particularly sensitive to pesticide applications, unlike Carabids (Odderskaer et al., 1997).

The research conducted thus provides equivocal results, with some demonstrating an effect of pesticides on skylark densities or ecology, some demonstrating an effect only under certain conditions, e.g. bad weather, and other studies finding no effect.

3.1.4 Yellowhammer

Morris et al. (2005) found that yellowhammer invertebrate food was less abundant in cereal fields that had received insecticide applications during the breeding season. Adults avoided these fields when foraging early in the season, when chicks were predominantly fed invertebrates, but not late in the season, when chick diet was supplemented with cereal grain as it became available. Predicted foraging densities in fields that received no breeding season insecticide applications were nearly four times higher than in fields that did.

Chick condition had a weak positive relationship with insecticide applications up to one application, but a negative relationship with more than one application. A similar relationship was found between insecticide application and invertebrate abundance, with a general negative relationship, but fields with one insecticide application having a higher number of invertebrates than those receiving no applications. It was suggested that fields receiving no insecticide applications may not receive them due to inherently low invertebrate abundance associated with prevailing environmental conditions. Insecticides had the most significant and consistent effects on invertebrate abundance, with weaker, less consistent effects being found for herbicide or fungicide applications. Effects of timing of application were more important than those of number of applications.

Probability of chick starvation was related to abundance of important yellowhammer chick food invertebrates. However, no relationship was found between probability of starvation and insecticide applications. This may have been due to the relatively small number of fields that received breeding season insecticide applications during the study period, due to low aphid abundances and restrictions due to foot and mouth.

To overcome this problem, Boatman et al. (2004) investigated this relationship experimentally by increasing summer insecticide inputs in a proportion of fields to vary the extent of spraying around individual nests. Probability of brood reduction was affected by the proportion of the foraging area that was sprayed with insecticide within the 20 days before hatching. A relationship between the abundance of important chick food invertebrates and levels of chick starvation was also found.

Hart et al. (2006) went on to find relationships between abundance of 'important' invertebrates and chick mass and condition and between chick mass and condition and brood reduction (due to starvation). Models also showed that insecticide applications within 20 days decreased invertebrate abundance to a level that could depress yellowhammer breeding productivity (Hart et al., 2006). Abundance of invertebrates was greater in boundary strips than in the crop from May until July. Chick food invertebrates were also more abundant in boundary strips during May and June, but by July had increased in the crop but not the boundary strips, so that in July chick food invertebrates made up a higher proportion of the total number of invertebrates in the crop than in the boundary strips. Chick food abundance was reduced in fields that had had insecticide applications within the last 20 days. The difference in abundance between sprayed and unsprayed fields increased as the season progressed, with chick food abundance in unsprayed areas increasing from mid-May to mid-July, whilst that in sprayed areas remained depressed. Thus, seasonal effects are important in terms of the indirect effects of pesticides on yellowhammers. Early in the season (May and June) there is more alternative foraging habitat, but by July chick food is no more abundant in the boundary strips than in the crop, and chick food invertebrates actually constituted a higher proportion of the total number of invertebrates in the crop than in the boundary strips by this time. Availability of alternative foraging habitat is also important.

The studies all found that timing of applications was key in determining their effect. Summer insecticide applications represent relatively small proportion of total insecticide use, and are generally only made when pests reach certain threshold values now, but their effects during critical periods are still important. For yellowhammer, these would probably be from March until late June/early July, after which nestlings are no longer solely reliant on invertebrate food, but for other species, insecticides may continue to be detrimental until crops are harvested.

To have population level effects any reduction in productivity or survival must be of sufficient magnitude to reduce the population growth rate of the species in question. Modelling of the consequences of changing food supply is often constrained by lack of data. However, Hart et al.'s (2006) work quantified relationships between insecticide applications, invertebrate abundance and yellowhammer productivity in a series of interlinked equations. These were used by J Crocker (unpublished data) to create models to predict the likely population effects of insecticide applications on yellowhammers.

J Crocker (unpublished data) used functions derived from Hart et al.'s (2006) field study, and published estimates of survival rates of yellowhammers (Siriwardena et al., 1998, Bradbury et al., 2000), to create deterministic models of yellowhammers at a study site in Norfolk. These investigated the effects of variation in chick food abundance, caused by summer insecticide, on population growth rates, and compared the importance of different demographic factors. These models indicated that short-term changes in chick food abundance caused by use of summer insecticides had only a small effect on population growth rate. Variation in over-winter survival of adults and first-year birds was a more important driver of population growth rate than mortality of nestlings and fledglings. Models assuming over-winter survival rates from periods of population decline suggested that increasing chick food abundance to the highest levels measured at the study site would not be sufficient to stabilise population growth, but that chick food abundance would need to increase by at least a factor of three to achieve a stable population. When over-winter survival rate from the early 1990s (when the population was in decline) was substituted with values from periods of stability or increase, the models predicted that the study population would increase under typical conditions of food abundance. Within breeding productivity, predation had more influence on population growth rate (explaining 14 % of variation) than did brood reduction through starvation (2 %). The models thus suggest that the effect on population growth rate of omitting summer insecticide are minimal. For example, the difference in population growth rate between 'no fields sprayed' and 'all fields sprayed' was 1 %. The model linking area of foraging territory sprayed with brood reduction from Boatman et al., 2004, suggested a similar magnitude of effect spraying all of the foraging territory, rather than spraying none of it, of about 2-3 %.

However, it should be remembered that the models only investigated the short-term effects of summer insecticides. They also only modelled influence of invertebrate food abundance on chick growth and survival in the nest. Breeding season insecticide applications could also be affecting survival of fledglings after they have left the nest, adult survival, or future reproductive output of first-year or adult birds. The models suggest that further research to give more precise estimates of over-winter survival of adults and first-years would improve the model predictions, as would investigations into post-fledging survival and brood predation rate. They also illustrate the importance of providing sufficient food resources throughout the year.

3.1.5 Cirl Bunting

The cirl bunting was once a common species in the UK, but populations collapsed both in numbers and range from the 1930s, and particularly from the 1970s (Sitter, 1982, 1985), such that by 1989 the population was reduced to just 120 territories (Evans, 1992). Cirl buntings require both unimproved grassland and weedy winter stubbles, for summer invertebrate and winter seed food respectively (Evans and Smith, 1994, Evans et al., 1997), and as such have suffered from intensification of grassland, and the switch from spring to autumn sowing of cereals, and associated loss of over-winter stubbles. Stubbles that do remain increasingly follow on from cereals with high-input herbicide regimes, and so contain fewer weed seeds.

In 1994, a Special Project for cirl buntings was introduced within Countryside Stewardship, restricted to target areas, mainly in south and east Devon. The project included options for low-intensity grassland, over-winter stubbles following low-input barley, and grass margins. This project was highly successful, with numbers of breeding cirl bunting increasing 83 % on agreement land compared to just 2 % on adjacent land between 1992 and 1998, and the increase on agreement land was part of a regional increase, rather than a redistribution of birds away from non-agreement land (Peach et al., 2001). In 1998, 22 % of all UK cirl buntings were breeding on land under CSS agreements in Devon, with a further 16 % of birds within 0.5 km of land under stewardship. The UK cirl bunting population subsequently increased to 697 pairs in 2003 (Wotton et al., 2004), again with a differential increase on agreement land. Countryside Stewardship grass margins had to be maintained without the use of pesticides or herbicides, and the over-winter stubbles were preceded by low-input spring barley. Options providing grass margins, and the weedy stubbles tended to gain additional cirl buntings. However, as a number of options are available under stewardship it is not clear to what extent restrictions on pesticide inputs led to the increases observed on project land.

Recently, use of stubbles provided under the Special Project and conventional stubbles was compared, in order to see whether the less expensive conventional Countryside Stewardship stubbles (OS1 and OS2 arable options) would be equally valuable to wintering cirl buntings (Bradbury et al., in press). Special Project stubbles allow use of fungicides, growth regulators and specified herbicides to control grass weeds, but prohibit the use of insecticides and herbicides to control broad-leaved weeds. Densities of birds were compared over 186 stubble fields during the winter of

2003/04, plant surveys were also conducted on these fields, and weed samples taken from a subset of fields.

Cirl bunting numbers were positively correlated with broad-leaved weed seed density. Special Project stubbles supported higher weed seed density, driven mainly by increased broad-leaved weed seeds (although they did not support more broad-leaved weeds), and were selected over conventional stubbles by cirl bunting, as well as yellowhammer and reed bunting. Linnet and meadow pipit also selected the Special Project stubbles over conventional wheat stubbles, but not conventional barley stubbles. In the case of meadow pipit, the only insectivore in this list, this is likely to be due to higher invertebrate abundance in Special Project stubbles, either due to the prohibition of insecticide use or restrictions on herbicide use in the preceding crop, although invertebrate abundance was not measured (Bradbury et al., in press).

Although the study does not necessarily imply that conventional stubbles are of no benefit to cirl buntings, which was not tested, the study indicates that Special Project stubbles are likely to be the best means of conserving this species, and also provide additional benefits for other species. Countryside Stewardship did not have prescriptions for stubbles preceded by low-input spring cereals, however such options are now available within Environmental Stewardship. Payments for prohibition of pre-harvest desiccant, post-harvest herbicide and a reduced herbicide regime in the preceding crop are available under Higher Level Scheme option HF15, whilst payments for just prohibition of pre-harvest desiccant and post-harvest herbicide are available under Entry Level Scheme option EF6. Special Project stubbles currently deliver 3.9 cirl buntings per 10 hectares, compared to 1.7 for conventional barley stubble and 0.9 for conventional wheat stubbles, and it is assumed that similar results will be seen under the HLS option (Bradbury et al., in press). How much additional benefit the ELS option provides compared to conventional stubbles depends on the relative importance of pre-harvest desiccant and reduced herbicides in determining weed and seed density in stubbles, which is as yet unquantified. There is also an option for winter stubbles following an unsprayed crop under Tir Gofal in Wales.

There have been no studies specifically evaluating effects of chick food abundance on cirl bunting productivity, or of pesticides on the breeding population of cirl buntings. Evans et al. (1997) demonstrated that late season nests were twice as likely to produce fledglings as early ones, due to lower rates of starvation and predation, and that chicks from late nest had higher growth and survival rates. Chicks were fed a higher proportion of Orthoptera late in the season, and thus is was suggested that low invertebrate abundance may be limiting the population. The fact that cirl bunting populations increased more on land under Countryside Stewardship agreements, which included provision of grass margins, a rich source of summer food, and weedy stubbles, a rich source of winter food, imply that there may be an effect of pesticides on the size of the breeding population, but this has not been explicitly demonstrated. Preference for stubbles following low-input cereal regimes in winter has been demonstrated, but whether this translates into an increased breeding population is not clear. However, it is possible that it will; a recent paper by Gillings et al. (2005) demonstrated that availability of a winter food resource (over-winter stubbles) affected breeding population trends for a suite of species.

3.1.6 Turtle Dove

Turtle doves nest in scrubby habitats associated with woodland edge or farmland (Mason and Macdonald, 2000, Browne and Aebischer, 2004, Browne et al., 2005). The British turtle dove population declined by 83 % between 1970 and 2005 (Eaton et al., 2007). Early analysis of nest record data suggested that breeding success per attempt was not responsible for these declines (Siriwardena et al., 2000, Browne et al., 2005). However, comparison of data after the declines (1998-2000) with that before the declines, in the 1960s, showed that turtle doves now have a shorter breeding season, and consequently produce about half the number of young than they did previously (Browne and Aebischer, 2004). Number of nesting attempts has dropped from 2.9 +/- 0.1 in the 1960s to 1.5 +/- 0.1 between 1998 and 2000 (Browne and Aebischer, 2002). A simulation model suggested that this reduction in productivity would result in a population decline of about 17 % per year (Browne and Aebischer, 2004). Overall breeding success has fallen from 2.1 (+/- 0.3) chicks fledged per pair in the 1960s to 1.3 (+/- 0.2) by 1998-2000 (Browne and Aebischer, 2002).

Turtle doves feed solely on seeds. Radio-tracking work found that turtle doves foraged predominantly at ‘man-made’ sites, such as spilt grain, animal feed and grain stores, and were rarely recorded feeding at ‘natural’ sites (Browne and Aebischer, 2003). Analysis of diet found that wheat and rape seeds accounted for 61 % of all seed eaten on average, in contrast to the situation in the 1960s, when weed seeds accounted for over 90 % of those eaten, with wheat and rape seeds making up only 5 % (Browne and Aebischer, 2003).

Analysis of nest record data from 1965 to 1995 suggests that breeding densities on farmland and woodland have fallen in proportion to loss of nesting, rather than feeding, habitat. However, the dramatic reduction in number of young fledged per pair, and the dietary shift from weed to crop seeds observed since the onset of agricultural intensification, would suggest that loss of weed seed food is probably adversely affecting turtle dove populations. Effects of herbicides on turtle doves have not been directly investigated, but would be of value, and turtle doves are likely to benefit from provision of seed rich habitat in the agricultural landscape.

3.1.7 Linnet

A similar situation is found for another exclusively granivorous bird, the linnet. The British breeding population of linnets showed a severe decline during the 1970s and 1980s, followed by partial recovery during the 1990s (Siriwardena et al., 1998). Siriwardena et al. (2000) suggested that a reduction in fledglings produced per breeding attempt may have been an important driver of the initial declines. However, models show that recent levels of breeding productivity are sufficient to maintain or increase the population, which is in accordance with recent increases in abundance recorded on Common Birds Census (CBC) plots (Moorcroft and Wilson, 2000). Chick starvation rates are low, and suggest that breeding productivity is not limited by food supply (Moorcroft and Wilson, 2000).

However, as for the turtle dove, nestling diet has changed dramatically since the onset of agricultural intensification, with many traditional food plants being lost from, or found at lower levels, in nestling diet since the 1960s, probably due to decreased weed abundance, with part-ripe oil seed rape now forming the predominant chick food source (Moorcroft et al., 2006). Thus, although effects of herbicide use have not been directly investigated for the linnet, populations are likely to have been affected by decreased weed seed abundance. As recent population trends show, availability of oil seed rape seeds have compensated for this loss of weed seeds, and allowed the population to recover. However, the population remains vulnerable to changes in cropping patterns.

3.1.8 Yellow wagtail

The British yellow wagtail population has declined by 65 % since 1970 (Eaton et al., 2007), during which time the range has also contracted towards core population centres in southern and eastern England (Chamberlain and Fuller, 2001). Declines have been greatest in the pastoral west, and modern strongholds of the species tend to occur in arable dominated regions (Gibbons et al 1993, Chamberlain and Fuller, 2001, Gilroy, 2006).

Yellow wagtails are entirely insectivorous. Gilroy (2006) conducted an intensive autoecological study of an arable-breeding population of yellow wagtails in eastern England. It was found that most nesting attempts early in the season (May) occurred in autumn-sown cereal fields (May), but cereals were avoided once they reached their full height and potato crops were strongly favoured for later broods, in June and July. Occupancy rates were also relatively high in field beans and pea crops, particularly in June and July. This pattern broadly concurs with other published crop associations for the species (Stiebel, 1997, Mason and Macdonald, 2000). Preferred foraging habitats included wheat and potato crops, as well as field margins including ditches and tracks, with foraging preferences also shifting away from wheat and towards potato crops as the season progressed (Gilroy, 2006).

The fact that potatoes are a preferred nesting and foraging habitat late in the season means that the yellow wagtail is potentially susceptible to the effects of indirect effects of pesticides, as they receive a high number of applications (Pesticide Forum, 2006). Gilroy's (2006) study found that starvation in the nest was rare, and chick condition was not related to the availability of preferred foraging habitats. The factor suggested as most likely to be limiting the population was choice of nesting habitat, which influenced likelihood of predation, in that preferences for field beans and for nesting close to tramlines in winter cereals led to high predation rates (Morris and Gilroy, in review). However, the variation in management intensity within Gilroy's (2006) study area was minimal, and it is suggested that a study over an area encompassing a broad range of farm management styles, or experimental manipulation of chemical applications, would be of value in determining effects of pesticides on this species.

3.1.9 Barn Swallow

The long-term population trend of barn swallows in the UK is uncertain (Evans, 2001). Although CBC data indicates that populations have increased since the 1960s, anecdotal evidence suggest that populations have declined, with localised population declines, at least partly in response to loss of nest sites, being documented (Noble et al., 2000). The discrepancy could be due to the fact that CBC mainly monitors populations breeding on farms, whilst, at least in the past, many swallow populations bred in villages (Evans, 2001). Data also suggest that decreases have been greater in arable areas (Evans, 2001). However, Breeding Bird Survey (BBS) data demonstrates that, since 1994 at least, the population has increased in the UK as a whole (Evans, 2001).

A study of breeding barn swallows on three mixed farms in Oxfordshire between 1998 and 2000 found no significant short-term effects of pesticides on food taxa (Evans, 2001). Other studies suggest that invertebrate taxa on which swallows feed are negatively affected by pesticide use (Campbell et al., 1997, Ewald and Aebischer, 1999, Wilson et al., 1999). Morris et al. (2001) suggest that the lack of observed effect could be as in mixed farming areas aerial invertebrates are able to disperse rapidly from unsprayed areas such as permanent pasture, and that any effects of pesticides on aerial invertebrates over the arable fields are thus very short-lived. However, it should be noted that low aphid numbers meant that there was only one breeding season insecticide application during the study, and so these results could also be just due to lack of power. There was also little evidence for differences in aerial invertebrate communities of organic and conventional farms (Evans, 2001). Again, this is likely to be due to the ability of aerial invertebrates to disperse over large areas and so recover rapidly from any disturbance events, such as pesticide applications.

Evans (2001) also found no significant effects of pesticides on presence of foraging swallows over fields, however as before the sample sizes were small. Morris et al. (2001) suggested that, despite low sample sizes, the fact that arable is used little for foraging compared to pasture (Evans, 2001) means that reductions in swallow numbers due to pesticide applications is unlikely to be a major issue in areas where pastoral farming persists. However, pesticide use could, potentially, have contributed to arable fields becoming unsuitable foraging habitat. Despite the lack of effects of pesticides on swallow food resources found by Evans (2001), a study in Spain found that use of a biological control agent decreased abundance of invertebrates to levels that reduced barn swallow breeding success (Cabello de Alba, 2002). Thus, evidence of effects of pesticides on barn swallows is lacking, further study to determine whether there genuinely is no effect or whether the lack of effect was due to lack of power would be of interest, although as swallow populations in the UK are currently increasing this is probably not a high priority.

3.2 Agri-environment prescriptions likely to compensate for effects of pesticides

Options from agri-environment schemes considered likely to compensate for the effects of pesticides are discussed below. Although regular weed control by both herbicides and mechanical methods has been found to have negative effects on field use by granivorous birds (Buckingham *et al*., 2006), pesticide use on grassland is still relatively minimal (Vickery *et al*., 2001, Pesticides Forum, 2006), and so options discussed relate to prescriptions for arable farmland. Arable pockets within predominantly pastoral areas are beneficial to farmland birds (Robinson et al., 2001, Vickery et al., 2001, Buckingham et al., 2004), and current cropping prices may present an opportunity to increase these. This is likely to be beneficial as long as their management is such that they remain food-rich. However, this is not discussed further here. Likewise, only options that are likely to increase abundance of winter or breeding season food resources are discussed, and not those relating only to provision of nesting habitat. Obviously, agri-environment prescriptions on grassland, or relating to nest site provision, can benefit species of farmland birds affected by pesticides, however here we are considering options likely to directly compensate for the effects of pesticides, namely reduced food abundance on arable land.

In this sense, the aim is different to that of Glass et al. (2006) who selected a suite of options likely to mitigate for effects of pesticides on three key species of farmland bird, the grey partridge, yellowhammer and corn bunting, in that Glass et al. (2006) discussed all options likely to be needed to benefit populations of these species, including nesting habitat.

Options are discussed in broad terms below, specific agri-environment options for each of the UK countries are listed in Table 1, along with figures on their uptake. A review by Ewing et al. (2008) of the benefits of the Entry Level Scheme to farmland birds was of great help when writing this review, and has been drawn on to a large extent in the following section.

3.2.1 Conservation headlands

Conservation headlands are cropped margins varying in width, at the boundary of cereal fields, that receive no inputs of insecticide and only restricted applications of selective herbicides and fungicides. They have been shown to support higher densities of important chick food invertebrates (see Frampton and Dorne, 2007, for review) and bird food plants (Critchley et al., 2004) than conventional field boundaries.

Grey partridge brood sizes and chick survival rates are higher in cereal fields with conservation headlands, due to increased abundance of chick food invertebrates (Rands, 1985). Whitethroat, greenfinch and yellowhammer are positively associated with boundary habitats next to conservation headlands (Stevens and Bradbury, 2006). De Snoo et al. (1994) found that yellow wagtail (but not skylark) used unsprayed margins significantly more than sprayed margins, although Gilroy (2006) found field edges were not used very much by foraging yellow wagtail. This could be due to differences in the other foraging habitat available.

Options for unharvested conservation headlands are likely to provide a food source for farmland birds throughout the year, rather than just during the breeding season, as there will be spilt grain and seeding arable plants in the unharvested crop during the winter. A review of ES options in 2007 has recommended that the option for conservation headlands which does not prohibit fertiliser use should be withdrawn, leaving just the low-input options, as application of fertilisers to conservation headlands both causes weed problems, and reduces their value to birds.

3.2.2 Margins and Buffer Strips

Grass margins and buffer strips

There are a number of agri-environment options for creation of field margins or buffer strips with restricted herbicide inputs, and these include options for margins sown with flower mixtures (Table 1). These create grassy strips between the crop and adjacent boundary features. Field margins and buffer strips support high abundances of invertebrates, and thus provide valuable foraging habitat for farmland birds, as well as buffering field edge habitats, e.g. hedgerows, from agrochemical drift.

In 1994, a Special Project for cirl buntings was introduced under Countryside Stewardship, which included prescriptions for grass margins. The scheme had clear benefits, with cirl bunting numbers increasing by 83 % on project land compared to just 2 % on non-project land, between 1992 and 1999 (Peach et al., 2001). Although the Special Project contained a suite of prescriptions, there was strong evidence that agreements containing grass margins gained additional cirl buntings, and that 6 m wide margins were more beneficial than 2 m margins (Peach et al., 2001).

Margin width also influences density of yellowhammers (Bradbury et al., 2000), and occupancy by whitethroats (Stoate and Szczur, 2001). Margins are also likely to provide nesting and foraging habitat for grey partridge (Aebischer et al., 1994), corn bunting (Wilson et al., 2007), reed bunting (Brickle and Peach, 2004) and chaffinch (Stevens and Bradbury, 2006). Despite these clear benefits to farmland birds, Marshall et al. (2006) found that there was no significant difference in abundance of seven species of farmland bird on fields with and without unsown margins, although it is possible the study was at an insufficient spatial scale to demonstrate an effect. Although most evidence for benefits of grass margins relates to the breeding season, a study of grey partridge in Scotland found them to be strongly associated with grass margins in winter (Hancock and Wilson, 2002).

Most margins are mown annually in late summer, and over the years can become dense and rank with little bare ground. This can prevent germination of annuals and movement of surface-dwelling insects, and also limit access for foraging birds. Preference for grass margins by foraging yellowhammers decreases through the breeding season, and manipulations of margin height showed that this was due to decreasing accessibility of invertebrates within margins later in the season (Douglas et al., in prep.). The Sustainable Arable Farming For an Improved Envrionment (SAFFIE) project investigated new management techniques for grass margins, including scarification and graminicide use. Scarification as with a power harrow in spring to create 60 % bare ground across margins. Applications of graminicide were made in spring to suppress more vigorous grass species. These aimed to reduce competitive grasses and increase bare ground, resulting in greater biodiversity benefits. Both methods supported higher densities of birds. Field margins that are cut in autumn support higher densities than those that remain uncut (Henderson et al., 2007), probably due to increased accessibility of the food resources.

Wildflower and grass mixtures on 6 m margins, and pollen and nectar flower mixtures, are designed to provide suitable foraging habitats for key insect pollinators, such as bumblebees and butterflies, but may also provide benefits and other invertebrate groups (Pywell et al., 2007), and for farmland birds. Skylarks in Switzerland have shown preference for nesting and foraging in wildflower margins, and this is likely to be due to the greater invertebrate abundance (Weibel, 1998, 1999). A study of skylark (Weibel, 1999) found that although clutch size was higher and feather growth rate in fledglings faster where wildflower margins were provided. However, a potential problem associated with field margins is that they can constitute a predator trap due to higher rates of nest predation in and around margins (e.g. Weibel, 1999; Morris and Gilroy, in review).

Buner et al. (2005) found that territory locations of grey partridge were associated with distribution of 'ecologically enhanced areas' (e.g. wildflower strips and hedges), and that wildflower field margins were strongly preferred, particularly in summer, but with associations also being found in autumn and winter.

Options to create buffer strips in other areas of the farm e.g. next to in-field ponds, watercourses, arable trees, as well as field corner options and those limiting cultivation and pesticide input next to hedgerows and ditches are likely to provide similar benefits to field margins (Table 1).

3.2.3 Uncropped margins

Uncropped margins are predominantly aimed at benefiting rare arable weeds, but support higher abundances of both weeds (Critchley et al., 2004) and invertebrates (ADAS, 2001, Pywell et al., 2007). They can thus provide valuable resources to farmland birds, particularly during the breeding season (Vickery et al., 2002). For example, distribution of reed buntings is related to presence of margins (Stevens and Bradbury, 2006). Uncropped margins are often characterised by sparse vegetation cover and greater area of bare soil (Meek et al., 2002), which will also increase accessibility of food resources.

However, it should be noted that uncropped margins may not be appropriate for all soil types; on heavier clay soils they can lead to growth of pernicious grass weeds such as black grass, which are agricultural pests and also of little value as bird food.

3.2.4 Wild bird seed mixtures

Agri-environment options for cultivation of seed-bearing crops, such as kale, quinoa and various cereals, with restricted herbicide applications, aim to provide seed rich habitats for farmland birds, particularly in winter. As well as containing a high abundance of crop seeds, these mixtures also support high abundances of arable weeds (Pywell et al., 2007) and invertebrates (Moreby and Southway, 2002, Pywell et al., 2007)

Winter abundances of a suite of farmland birds are higher when wild bird seed mixtures are present (Stoate et al., 2003), and there is also evidence that these options are beneficial in summer; foraging densities of up to 80 times those in conventional crops have been recorded (Parish and Sotherton, 2004), and kale and cereal based wild bird crops are preferred habitats for skylark when provisioning young (Murray et al., 2002).

Despite these apparent benefits, a study of in France found no effect of provision of wild bird crops on grey partridge reproductive success (Bro et al., 2004), and there is also evidence that wild bird crops can constitute predator traps, with a study of grey partridge in France finding lower over-winter survival on plots with wild bird strips.

3.2.5 Beetle banks

Beetle banks are grass strips that receive no insecticide, and restricted herbicide, applications. They are usually about 2 m wide, and created on a raised bank through the middle of an arable field. Their aim is to provide over-wintering habitat for beneficial predatory invertebrates (Thomas et al., 1992), which then colonise cereal crops during spring and summer, and prey on cereal pests, such as aphids.

Beetle banks do support high densities of polyphagous predators especially beetles and spiders, during the breeding season (Thomas et al, 1991, 1992, Collins et al., 2003) and also in winter (Collins et al., 2003). This would suggest that they could provide foraging opportunities for farmland birds. However, Stevens and Bradbury (2006) found no benefits of beetle banks for corn bunting, lapwing, skylark or yellow wagtail. Murray et al. (2002) found foraging preference for beetle banks in skylarks (but not yellowhammers) in Leicestershire. This relatively weak evidence for beetle banks providing foraging opportunities is probably because of poor accessibility due to their tall, dense vegetation structure. However, the lack of result is still somewhat surprising as it might be expected that the interface between the crop and beetle banks would be a good source of invertebrates for farmland birds.

3.2.6 Fallow plots

This option is for plots of at least 2 ha, managed by cultivating in spring to produce a rough fallow, and retained without the use of pesticides or fertilisers. Although designed to provide breeding and foraging sites for ground-nesting farmland birds such as lapwing and stone curlew, fallow plots have also been found to benefit more common arable plants and some invertebrates (e.g. bumble bees), and so are likely to be of benefit to farmland birds generally (Fisher and Anderson, 2006).

3.2.7 Over-wintered stubbles

Loss of over-winter stubbles in farmland has led to a reduction in abundance of winter seed food, exacerbated by the fact that stubbles that do remain are generally less seed-rich due to increased harvesting efficiency and better weed control (Wilson et al., 1996*)*. Preference for winter stubbles as foraging habitat for seed-eating birds in winter is well established, and has been demonstrated for a suite of species including greenfinch, reed bunting, linnet, yellowhammer, goldfinch, skylark and grey partridge (Donald and Evans, 1994, Evans and Smith, 1994, Wilson et al., 1996, Wakeham-Dawson and Aebischer, 1998, Buckingham et al., 1999, Bradbury and Stoate, 2000, Moorcroft et al., 2002, Butler et al., 2005).

Gillings et al. (2005) used BBS and Winter Farmland Bird Survey (WFBS) data and found that the area of stubble in winter attracts increased numbers of several birds of conservation concern, e.g. squares with high densities of skylark in summer had relatively higher densities in winter if stubbles to some degree replaced crops. As well as attracting birds in winter, stubble availability in winter was found to have an effect on the 10-year breeding population trend, from 1994 to 2003. A positive linear relationship was found for eight species (yellowhammer, skylark, lapwing, stock dove, mistle thrush, starling, goldfinch and bullfinch). This relationship was quantified in detail for skylark and yellowhammer. Skylark populations in 1 km squares with less than 10 ha of stubble declined by 20 % between 1994 and 2003, compared to 34 % in the complete absence of stubbles. Populations on squares containing more than 10 ha of stubble only declined by 4 %, and the 10-year linear trend was stable or increasing only when stubble availability exceeded 20 ha per square. The results for yellowhammer were similar, with approximate stability at 15 ha of stubble or more. Nationally, only 50 % of squares contained stubbles, and in squares containing stubble the median area was 12 ha, with the overall average currently being 2 ha per 1 km (Gillings and Fuller, 2001).

Despite the evidence that stubbles constitute a preferred foraging habitat for farmland birds in winter, they still can support relatively low numbers of birds (Gillings and Fuller, 2001, Vickery et al., 2005). Gillings and Fuller (2001) estimated that only 46 % of stubbles were weedy and hence provided a valuable foraging habitat for farmland birds. Density of seeds within the stubbles is greatly influenced by pesticide use in the preceding crop. Stubbles which are preceded by a low-input spring cereal support higher densities of cirl buntings in winter, as well as yellowhammers and reed buntings (Bradbury et al., in press). Thus, the HLS option for stubble preceded by low-input spring cereal is likely to prove particularly beneficial. Thus, measures to increase weed abundance in the stubbles, such as the

HLS option for a preceding low-input spring crop, and Tir Gofal option for winter stubbles following an unsprayed crop, could greatly reduce the areas of stubble required to change population trends for these species.

The fixing of the set-aside rate to 0 % in 2007 has greatly reduced the area of over-wintered stubbles, increasing the need to boost winter seed food for birds in the UK. There was a 14 % reduction in the area of uncropped land (set-aside and fallow) in England between 2006 and 2007, due to factors such as increased cropping prices and a move to biofuel production. A reduction in the area of uncropped land (set-aside and fallow) in England of just over 50 % was predicted for 2008 (Defra Agricultural Change and Environment Observatory, 2008). Although the area of uncropped land was already decreasing (by 14 % between 2006 and 2007) due to factors such as increased cropping prices, it is considered that this change is much greater than would have been expected without the zero set-aside rate (Defra Agricultural Change and Environment Observatory, 2008). Work to evaluate the loss of set-aside and the implications and possible mitigation measures for farmland birds is underway (Defra Agricultural Change and Environment Observatory, 2008).

3.2.8 Skylark plots

The decline of skylark populations has frequently been attributed to loss of spring sown crops, resulting in fewer breeding attempts being made, as winter sown crops become too dense for late breeding attempts to be made (Donald and Vickery, 2000). Skylark plots consist of undrilled areas in winter wheat fields, and have been shown to provide foraging resources for skylark during the breeding season, as well as other farmland bird species (Cook et al., 2007, Morris et al., 2007).

Densities of skylark territories in winter wheat fields with and without plots are similar early season, but as the breeding season progresses, densities on conventional winter wheat fields decrease but fields with skylark plots retain early breeding season densities, and are 40 % higher than those in conventional fields (Morris et al., 2007). Plots also increase skylark productivity, with higher clutch sizes, and a non-significant trend for higher nest survival, resulting in pairs in fields with skylark plots producing an additional 0.5 chicks per attempt (Morris et al., 2007). These benefits appear to be due to increased foraging habitat, rather than nesting habitat, as the plots are rarely used for nesting, and it is thought this is due to greater accessibility, rather than abundance, of food resources (Morris et al., 2007).

Highest densities of birds are found in fields with both skylark plots and field margins, with yellow wagtails having territory densities that are five times higher in fields with both of these prescriptions that fields lacking one or both of them (Cook et al., 2007). Territory densities of species with Biodiversity Action Plans was 2.8 times higher on fields with both options than on conventional winter wheat fields (Cook et al., 2007). This is likely to be due to the combination of abundant and accessible food resources provided by margins and skylark plots respectively.

Feedback from some farmers drilling at high speeds suggests that creating skylark plots by cessation of drilling is difficult, and that spraying off plots would be a more popular option (R Winspear, pers. comm.). Vegetation structure of plots sprayed out during the autumn does not differ from those created by drilling, although delaying spraying out until the following spring is not good for foraging birds, as it results in a mat of dead vegetation for much of the breeding season (Tony Morris pers comm). Thus, a review of ES options in 2007 will recommend allowing establishment of skylark plots by spraying out.

3.2.9 Undersown spring cereals

Undersowing is a traditional rotational cropping practice where a spring crop is sown along with a grass/legume mix, and following harvesting, a grass ley is allowed to develop. Whilst pesticide inputs are not restricted on this option, undersowing of spring cereals reduces the need for agrochemical inputs.

The GWCT's study of grey partridge found that populations on four intensive farms in the study area decreased on average by 72 % between 1970 and 1994, whilst those on the only farm to retain a traditional ley farming system remained stable (Aebischer and Potts, 1998). Undersowing may be beneficial due to provision of cereal stubbles, or as undersown spring cereals and also grass leys are rich foraging areas, particularly for chick-food invertebrates e.g. sawfly larvae (Aebischer, 1990). Higher densities of skylark (Wakeham-Dawson et al., 1998) and corn bunting (Aebischer and Ward, 1997) have been found in areas with undersown cereals. However, the Arable Stewardship Pilot Scheme found that abundance of Hemiptera and Carabidae did not differ greatly between undersown and conventional spring cereals (ADAS, 2001). Vickery et al. (2007) showed that small insectivorous farmland bird species used undersown spring cereal margins more than conventional grass margins in pastoral-dominated areas of England.

However, undersown cereals do not constitute good foraging sites for solely granivorous species, as weed seed abundances (Critchley et al., 2004), and accessibility (Evans et al., 2004) may be low, and thus there is little evidence for preference of undersown over-winter stubbles (Moorcroft et al., 2002).

3.3 Scale of management required

Although many studies have demonstrated the benefits of a range of agri-environment options to farmland birds, quantification of how much of particular options is desirable, for example how much would be required to stabilise the population of a particular species, is relatively rare.

As discussed in section 3.1.1, reduced chick survival due to decreased invertebrate food, as a result of insecticide and herbicide applications, has been implicated in the decline of the grey partridge, and this can be compensated for by provision of conservation headlands, where pesticide inputs are restricted in the outer section of cereal fields. It has been estimated that approximately 5 ha per 100 ha of insect-rich habitat within the cropped area, such as conservation headlands, is sufficient to achieve a sustainable grey partridge population (Aebischer et al., 2003, Table 3).

Gillings et al. (2005) used BBS and WFBS data and found that increased areas of stubble in winter attracted increased numbers of several birds of conservation concern in winter, but also affected breeding population trends (from 1994 to 2003). A positive linear relationship was found for eight species (yellowhammer, skylark, lapwing, stock dove, mistle thrush, starling, goldfinch and bullfinch). This relationship was quantified in detail for skylark and yellowhammer. Skylark populations in 1 km squares with less than 10 ha of stubble declined by 20 % between 1994 and 2003, compared to 34 % in the complete absence of stubbles. Populations on squares containing more than 10 ha of stubble only declined by 4 %, and the 10-year linear trend was stable or increasing only when stubble availability exceeded 20 ha per square (Table 3). The results for yellowhammer were similar, with approximate stability at 15 ha of stubble or more (Table 3). However, only about half of stubbles are weedy and so provide a good food source for farmland birds. Options to increase weed abundance in stubbles, such as it being preceded by a low-input spring cereal, as discussed in section 3.2.7, could greatly reduce the area of stubbles required. The fixing of the set-aside rate to 0 % in 2007 has greatly reduced the area of over-wintered stubbles, and so there is an urgent need to boost winter seed food for birds in the UK.

Thus, assessments of the area of different agri-environment options required in the landscape to achieve population stability (or increase) for farmland bird species are rare. However, in a recent review (Vickery et al. 2008), assessments have been made, based on the current area of agri-environment options in the landscape, of to what degree these options would be required to affect key demographic rates for different species of birds in order to cause population growth, and whether this change is considered achievable.

Vickery et al. (2008) identified the key demographic parameters driving the decline of 19 species of farmland bird and, for each of the 19 species, constructed models to quantify the change in the key parameter required to achieve population growth of 1 % over the course of a year. ELS options that were known to (probable scenario), or considered likely to (potential scenario), affect these demographic rates were identified, and their area in the landscape quantified. Based on the current total area in England of all potentially beneficial options, the change necessary in the demographic parameter in order to achieve population growth rate was calculated. Whether the required demographic parameter was achievable was then evaluated by comparing it to the current value of the demographic rate, and to the maximum-recorded value (where possible for stable or increasing UK populations). If the required demographic parameter was higher than the maximum-recorded value, it was considered unachievable. If it was higher than the current rate, but by a relatively large amount (> 25 %), it was considered 'possible'; if it was less than this it was considered 'probable'. The results of these analyses are presented in Table 4.

Using skylark as an example:

- the key demographic rate was identified as number of nesting attempts
- 10 % of farmland has ELS options that have a probable effect on the number of nesting attempts
- 34 % of farmland has options that have a potential effect on the number of nesting attempts.

The above information was combined with models to identify how much the demographic rate would have to increase to achieve population growth. For skylark, the required increase in nesting attempts associated with the ELS options was to 3.4 for the probable scenario, and to 2.4 for the potential scenario. As 3.4 is higher than the maximum recorded number of nesting attempts for a population, this was considered unachievable, whereas it was considered probable that an increase to 2.4 was achievable.

When looking at probable effects of options, ELS options were considered likely to lead to population growth for only three species, based on their current area. When the analyses were extended to potential effects, the target rate was considered achievable for many more species.

These models were used to assess whether the required change in key demographic rates to bring about population growth were achievable, based on the current level of uptake of ELS options. However, similar models could be used to assess what area of specific options was required in order to increase the populations of different bird species, where information existed on the changes in demographic rates achieved by particular options. This highlights the need for further quantification of the effects of agri-environment scheme options on different demographic rates to aid analyses such as these.

4. Discussion

The grey partridge remains the only species for which population-level effects of pesticide use have been demonstrated. Loss of insect food, due to both insecticide and herbicide use, has reduced chick survival rates sufficiently to cause population declines (Potts and Aebischer, 1995). For corn buntings, abundance of chick food invertebrates has been found to be affected by the number of insecticide, fungicide and herbicide applications, and chick food abundance affects parental foraging behaviour, as well as chick mass and nest survival (Brickle et al., 2000). However, models to assess whether pesticides have population level effects for corn bunting have not been conducted. Use of breeding season insecticides can affect yellowhammer foraging behaviour early in the season, chick condition (Morris et al., 2005) and brood reduction (Boatman et al., 2004). However, modelling suggests that the impact of these effects on overall population growth rate is likely to be relatively minimal, with variation in over-winter or first year survival being the most likely driver of declines (J Crocker, unpublished data). This highlights the importance of provision of year-round food for farmland bird populations. However, it should be noted that the models only took into account effects of breeding season insecticide applications, and of invertebrate food abundance on chick growth and survival in the nest. Breeding season insecticide applications could also be affecting survival of fledglings after they have left the nest, adult survival, or future reproductive output of first-year or adult birds.

Although most research on the effects of pesticides on farmland birds has focused on short-term effects of insecticide use during the breeding season, a recent study demonstrated effects of herbicide applications on winter food supply and farmland bird densities (Bradbury et al., in press). Over-winter stubbles that were preceded by a low input spring cereal contained more broad-leaved weed seeds and supported higher numbers of cirl buntings, yellowhammers and reed buntings. These stubbles were available as part of a Special Project for cirl buntings under the no longer available Countryside Stewardship Scheme, but similar options are now available under agri-environment schemes in England and Wales. Although these increased winter densities on weedier stubbles have not been related to breeding population trends, it is considered likely they could affect them. Gillings et al. (2005) demonstrated a link between availability of over-winter stubbles on breeding population trends for a number of species, and identified the amount of stubble required in the landscape to achieve population stability for skylarks and yellowhammers, and it is likely these amounts would be reduced if looking at weed rich stubbles such as these.

Other species that are likely to have been affected by loss of weed seed on farms include turtle doves and linnets. Both species feed exclusively on seeds all year, and have changed the type of seed fed to chicks dramatically since the 1960s, prior to agricultural intensification. Weed seeds accounted for the majority of chick diet in the 1960s for both species, but have since been largely replaced by wheat and rape seed for turtle dove (Browne and Aebischer, 2003), and part-ripe oil seed rape seed for linnet (Moorcroft et al., 2006). Turtle dove populations have declined since the 1970s and continue to do so rapidly (Eaton et al., 2007), whereas for linnet populations are now recovering, following severe declines during the 1970s and 1980s (Siriwardena et al., 1998). Thus for linnet, it would appear that availability of oil seed rape in the landscape has been sufficient to compensate for loss of weed seeds, although the population remains vulnerable to changes in cropping patterns. For turtle dove, however, average number of nesting attempts made per pair has roughly halved since the 1960s (Browne and Aebischer, 2002), and studies of the effects of herbicides on turtle dove would be of value. Provision of weed rich habitats would be likely to benefit the species.

Another species for which investigation of pesticide effects has not been conducted, but is considered valuable, is the yellow wagtail. Yellow wagtails are multi-brooded and late in summer show preference for nesting in potatoes (Gilroy, 2006), which receive relatively high numbers of pesticide applications (Pesticide Forum, 2006).

Evidence for effects of pesticides on skylarks remains ambiguous; with densities being higher on organic farms (Wilson et al., 1997), but these being associated with higher nest failure rates as nests can get destroyed during mechanical weeding operations (Kragten et al., in press). A small-scale study in Denmark also found effects of pesticides on insect chick food, number of nesting attempts made, nest survival, and chick diet, but many effects occurred only in poor weather (Odderskaer et al., 1997). For barn swallows, Evans (2001) found no effect of insecticide applications on availability of chick food insects, or where parents fed. This result may reflect a small sample size or genuine lack of an effect and further research would be valuable.

Most research into the indirect effects of pesticides on farmland birds relates to short-term effects of pesticides, for example studies of the effects of breeding season insecticide applications on productivity, or of herbicide applications on the preceding cereal crop on use of an over-winter stubble. However, it should also be remembered that the widespread introduction of pesticide use is considered to have caused large-scale losses of seed and invertebrate food over time that will have affected many species of farmland bird. Such long-term effects are difficult to demonstrate, with the best evidence for them coming from the GWCT's study of the grey partridge on the Sussex Downs.

Measures to reduce, or compensate for, the effects of pesticides can be via careful consideration of what, where and when to spray (see recommendations), or by provision of alternative food rich habitats on the farm, for example through agri-environment options. Options considered likely to be particularly beneficial in this sense have been discussed in this review. These are conservation headlands, margins and buffer strips, wild bird seed mixtures, beetle banks, fallow plots, over-winter stubbles, skylark plots and undersown spring cereals. Not all of these options will be available in each UK country, as agri-environment schemes vary, the relevant options for England, Northern Ireland, Scotland and Wales are presented in Table 1.

The key to success of these agri-environment schemes in increasing farmland bird populations is implementation on a sufficient scale. Introduction of the Entry Level Scheme in England goes some way to addressing this, as it is non-competitive and measures are available to all farmers. However, which options are chosen is up to the applicant, and several studies have shown that uptake of options is skewed, with a few options being chosen by a large proportion of applications, but over half of options being chosen by less than 3 % (Boatman et al., 2007). Amongst the options for arable farmers, field edge and boundary options are chosen more frequently than options for the cropped area, despite evidence that changes in the cropped area have been the most important drivers of farmland bird declines. Opinion as to whether provision of uncropped habitat can be sufficient to address farmland bird declines is divided (Butler et al., 2007, Holland et al., 2007).

Amongst the English agri-environment options highlighted in this review, Boatman et al (2007) found that uptake of some of these was less than 10 % on arable farms, with this applying to pollen and nectar flower mixture, beetle banks, skylark plots, conservation headlands and uncropped, cultivated margins. The most popular arable options were field corner management, wild bird seed mixture and overwintered stubbles, the latter two options proving exceptions to the rule that boundary and field edge options are generally most popular.

Similar results are found here. For England, buffer strips and margins, wild bird seed mixtures, and over-winter stubbles are the most popular of the options, all with several thousand agreements. Skylark plots, conservation headlands, undersown spring cereals and beetle banks are all less popular, with uptake in the hundreds. For Northern Ireland, numbers of agreements are lower, unsurprisingly given that only 3 % of agriculture in Northern Ireland is arable (DARD, 2007). Within the uptake figures, similar patterns are found, with margins, wild bird seed mixtures and over-winter stubbles proving most popular. Uptake of conservation headlands is low (63 agreements), as is that for undersown spring cereals (109 agreements) and particularly for fallow plots (3 agreements!). No options are available for beetle banks or skylark plots in Northern Ireland. In Wales, numbers of agreements are even lower, and although this reflects the lower arable area, even taking this into account there is evidence of relatively low uptake of arable options; only about 1 % of applications are on arable farms, despite it representing about 11 % of agricultural land in Wales (Howie, 2006). Unusually, in Wales, undersown spring cereals are the most popular of the optional prescriptions looked at (455 agreements), followed by winter stubbles (264), wild bird seed mixtures and conservation headlands (205 agreements each), with uptake of grass margins and uncropped margins being relatively low (20 and 75 agreements, respectively). Options for beetle banks, fallow plots and skylark plots are not available in Wales. In Scotland, unharvested crops (the equivalent of wild bird seed mixtures) are the most popular of the options considered likely to mitigate for pesticide usage, with 2 956 agreements including these, followed by creation of grass margins and beetle banks (2 449 agreements, information is not available for these options separately), and management of extended hedges is chosen in 929 agreements. Scotland is unusual in that conservation headlands are relatively popular (1 042 agreements) and spring cropping (associated with over-winter stubbles) the least popular option (< 100 agreements). Options for fallow plots and skylark plots are not available in Scotland. These results are similar to those found previously, in that uptake of in-field options such as conservation headlands, beetle banks and skylark plots are generally relatively low, compared to options for boundary features such as margins, and wild bird seed mixtures. Uptake of over-winter stubbles is also relatively high generally.

Thus, measures to try to alter uptake of prescriptions highlighted here as being beneficial, but with low uptake, would be valuable. Options for trying to alter the uptake of different agri-environment options include better guidance, increased points allocation, or a requirement to select from a particular range of options depending on farm type and location (Stevenson, 2007). The latter option would add further complexity to applying for agri-environment schemes and may prove unpopular. A survey conducted by Boatman et al. (2007) found that the main reason for picking an option was that the feature or management was already in place. However, for arable options, high point allocations were also frequently cited as a reason for selecting particular options (Boatman et al., 2007), and thus may prove the best method of influencing uptake.

Ideally, there would be quantification of the amount of different agri-environment prescriptions needed to benefit different farmland bird species. This is relatively rare, although a few examples have been presented in this review. However, in the absence of such quantification, measures should be taken to increase the range of options implemented, and a review of this by Defra is currently underway.

Further research

- Population modelling for species for which immediate effects of pesticides have been found on productivity, behaviour or densities (e.g. corn bunting), should be undertaken to ascertain whether these have population level effects.
- Studies to investigate effects of pesticides on species highlighted as being likely to be vulnerable to such effects, such as the turtle dove and yellow wagtail, would be valuable.
- Further quantification of the amounts of different agri-environment measures required to benefit farmland bird populations should be conducted.
- Investigations of the effects of fungicides on invertebrate populations could be conducted by manipulating fungicide use, to overcome problems of separating effects from those of insecticide applications, which are highly correlated with fungicide applications.

Recommendations

The Government's statutory code of practice (Defra, 2006, Scottish Executive, 2007) can be used to minimise unwanted effects of pesticides outside the cropped area. In addition to this, a number of ways of further reducing effects of pesticides on farmland birds are listed below (many are examples of Integrated Crop Management, or Voluntary Initiative best practice):

- Crops should be monitored regularly, and treatment thresholds used where available (Voluntary Initiative, 2003).
- Selective herbicide use can lead to increased seed and invertebrate abundances.
- Use of insecticides during the breeding season appears to have the most effect on farmland birds, and applications between mid March and harvest should be minimised.
- Spraying when broad-leaved crops and weeds are in flower should be avoided where possible.
- Integrated crop management techniques (Leaf, 2008) can be used to reduce the need for summer insecticides and broad spectrum herbicides, for example considering crop varieties that are resistant to orange blossom midge.
- Pesticides which will control the problem with least impact on non-target species can be chosen using the Environmental Information Sheets for the products. For example, the insecticide pirimicarb has relatively little effect

on non-target invertebrates compared to pyrethroid or organophosphate insecticides (Ewald and Aebischer, 1999).

- Timing, dose and number of applications should be judged on the lowest amount required to control the problem effectively. This must comply with label recommendations and cross-compliance, and should also bear in mind the risk of pesticide resistance with low doses of some pesticides, and added environmental costs associated with repeat applications if the first is not effective.
- Steps to minimise spray drift can be made when choosing of application equipment (e.g. choosing low drift nozzles), and setting boom height, taking note of weather conditions and allowing adequate buffer zones to protect vulnerable habitats, such as watercourses, hedgerows and unimproved habitats.
- Use of seed treatments where possible can reduce foliar insecticide use, but all treated seed should be buried and care taken to clean up spillages, as these can cause harm to birds and mammals when eaten.

- Agri-environment measures that are likely to further compensate for the effects of pesticides on farmland birds, by providing alternative food-rich habitat on the farm, are discussed in the report and listed in Table 1.
- Measures to increase the uptake of agri-environment options likely to be beneficial, but with low uptake, such as increasing their points allocation, should be considered.

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Table 1. Uptake of agri-environment options likely to compensate for the effects of pesticides

a) England

Measure	Code	Description	Main resource provided	Number of agreements*
Conservation headlands				
Entry level	EF9	Conservation headlands in cereal fields.	Summer invertebrate food	93
	EF10	Conservation headlands in cereal fields with no fertilisers or manure.	"	163
Higher level	HF14	Unharvested, fertiliser-free conservation headlands (rotational).	"	116
Margins and buffer strips				
Entry level	EE1	2 m buffer strips on cultivated land.	"	1 695
	EE2	4 m buffer strips on cultivated land.	"	3 393
	EE3	6 m buffer strips on cultivated land.	"	5 471
	EF4	Pollen and nectar flower mixture.	Summer plant and invertebrate food	1 734
	EF11	6 m uncropped, cultivated margins on arable land.	Summer seed food	368
Higher level	HE10	Floristically enhanced grass margin.	Year-round plant and invertebrate food	296
Wild bird seed mixtures				
Entry level	EF2	Wild bird seed mixture.	Year-round seed food	3 044
Higher level	HF12	Enhanced wild bird seed mix plots (rotational or non-rotational).	"	546

Beetle banks				
Entry level	EF7	Beetle banks.	Year-round invertebrate food	408
Fallow plots				
Higher level	HF13	Fallow plots for ground-nesting birds (rotational or non- rotational).	Summer invertebrate food	354
Higher level	HF20	Cultivated fallow plots or margins for arable flora (rotational or non- rotational).	Summer plant and invertebrate food	119
Over-winter stubbles				
Entry level	EF6	Over-wintered stubbles.	Winter seed food	3 967
Higher level	HF15	Reduced herbicide, cereal crop management preceding over-wintered stubble and a spring crop (rotational).	"	89
Skylark plots				
Entry level	EF8	Skylark plots	Summer invertebrate food	501
Undersown spring cereals				
Entry level	EG1	Undersown spring cereals.	Summer invertebrate food	328

Source: Defra, figures as at 23/01/08.
Information on Entry Level Stewardship prescriptions downloadable from:
http://www.defra.gov.uk/erdp/schemes/els/handbook/default.htm
Information on Higher Level Stewardship prescriptions downloadable from:
http://www.defra.gov.uk/erdp/schemes/hls/handbook/default.htm

b) Northern Ireland

Measure	Code	Description	Main resource provided	Number of agreements
Conservation headlands				
	4(ii)	Conservation cereal N.B. This is available as a whole field or a field margin option.	Summer invertebrate food	63
Margins and buffer strips				
	2(i)	Ungrazed grass margins	"	1 035
	4(v)	Rough grass margins	"	110
Wild bird seed mixtures				
	4(ii)	Wild bird cover	Year-round seed food	974
Fallow plots				
	4(vi)	Lapwing fallow plots	Summer invertebrate food	3
Over-winter stubbles				
	4(i)	Retention of winter stubble	Winter seed food	656
Undersown spring cereals				
	4(iv)	Undersown cereals	Summer invertebrate food	109

Source: DARD, figures as at 31/12/2007.
Information on Countryside Management Scheme and Environmentally Sensitive Areas scheme downloadable from:
http://www.ruralni.gov.uk/index/environment/countryside_management_main/pubs/agri-environment-scheme-leaflet-and-booklets.htm

c) Scotland

Information on both proposed measures from the Scottish Rural Development Programme (SRDP), which was not yet open when this document was written, and measures from the Rural Stewardship Scheme, which is now closed to new applicants, are both presented, so that information on both current options as well as levels of uptake are provided.

Scottish Rural Development Programme

Codes presented are from 'SRDP 2007-2013 Annex 3: Description of measures'.

Measure	Code	Description	Main resource provided
Conservation headlands			
	Arable Fields (a)	Biodiversity Cropping on In-Bye (Tier 2 and Tier 3) N.B. This option is for plots of spring cereals, fodder root crops or fodder rape, with restricted pesticide inputs.	Summer invertebrate food
Margins and buffer strips			
	Field margins and boundaries (d)	Management of Grass Margins and Beetlebanks in Arable Fields (Tier 2 and Tier 3)	Grass margins: summer invertebrate food
Wild bird seed mixtures			
	Wildlife on farmland and other types of land (a)	Wild Bird Seed Mix/Unharvested Crop (Tier 2 and Tier 3)	Year-round seed food
Beetle banks			
	Field margins and boundaries (d)	Management of Grass Margins and Beetlebanks in Arable Fields (Tier 2 and Tier 3)	Beetlebanks: Year-round invertebrate food
Over-winter stubbles			
	Arable fields (d)	Retention of Winter Stubbles (Tier 2)	Winter seed food

Information on the Scottish Rural Development Scheme downloadable from: http://www.scotland.gov.uk/Publications/2007/07/20145359/82

Rural Stewardship Scheme

Measure	Code	Description	Main resource provided	Number of agreements
Conservation headlands				
	6.2	Management of conservation headlands	Summer invertebrate food	1 042
Margins and buffer strips				
	6.1	Creation of grass margins or beetlebanks in arable fields	Grass margins: Summer invertebrate food	2 449
	6.3	Management of extended hedges	Summer invertebrate food	929
Wild bird seed mixtures				
	7.4	Unharvested crops	Winter seed food	2 956
Beetle banks				
	6.1	Creation of grass margins or beetlebanks in arable fields	Beetlebanks: Year-round invertebrate food	2 449
Over-winter stubbles				
	7.2	Spring cropping	Winter seed food	97
Undersown spring cereals				
	7.1	Introduction or retention of extensive cropping *N.B. Undersowing can be selected within this option (it is not a compulsory part)*	Undersowing option: Summer invertebrate food	Information not available on undersowing option

Source: Scottish Executive, figures as at 01/04/07.

d) Wales

Measure	Code	Description	Main resource provided	Number of agreements
Conservation headlands				
	TG24A	Unsprayed cereal, rape and linseed crops: Existing arable land. N.B. This is available as a headland or whole-field option.	Summer invertebrate food	205
Margins and buffer strips				
Tir Cynnal	Habitat creation option 3	Leaving uncropped margins on cereal land.	"	Data not available
	Habitat creation option 4	Creating grass margins on cereal land.	"	Data not available
Tir Gofal	TG28	Rough grass margins alongside cereal and root crops.	Summer invertebrate food	30
	TG29	Uncropped fallow margins alongside arable and root crops.	Summer invertebrate food	75
Wild bird seed mixtures				
	Habitat creation option 6	Establishment of wild bird cover crop.	Year-round seed food	Data not available
	TG30	Establishment of wildlife cover crops.	Year-round seed and invertebrate food	205
Over-winter stubbles				
Tir Gofal	TG25A	Retention of winter stubbles in cereal, rape and linseed crops: After a conventionally grown crop.	Winter seed food	82

	TG25B	Retention of winter stubbles in cereal, rape and linseed crops: After an unsprayed crop.	Winter seed food	182
Undersown spring cereals				
Tir Gofal	TG26	Spring sown cereals undersown with grasses and legumes.	Summer invertebrate food	455

Source: Countryside Council for Wales, 2006.

Information on Tir Cynnal downloadable from:
http://new.wales.gov.uk/topics/environmentcountryside/farmingandcountryside/farming/agri_env_schemes/tircynnal/?lang=en
Information on Tir Gofal downloadable from:
http://new.wales.gov.uk/topics/environmentcountryside/farmingandcountryside/farming/agri_env_schemes/tirgofal/?lang=en

Table 2. Effects of pesticides on farmland birds

Species	Effect	Strength of evidence
Grey partridge	Effect of herbicides and insecticides on insect chick food, chick survival, and population size.	Strong evidence.
Corn bunting	Effect of insecticides, herbicides and fungicides on insect chick food, which affects where parents feed, chick mass and chick survival.	Strong evidence, but effect on population not assessed.
Yellowhammer	Effect of insecticides in breeding season on insect chick food, where parents feed, and likelihood of chicks starving.	Strong evidence, but population models suggest unlikely to be main limiting factor at present.
	Effect of herbicide applications to spring cereal on bird numbers on the following over-winter stubbles.	Strong, but population level effects not investigated.
Skylark	Effects of pesticides on insect chick food, nest survival, number of nesting attempts and chick diet, but most effects only occurred in poor weather.	Ambiguous, needs further research.
Cirl bunting	Effect of herbicide applications to spring cereal on bird numbers on the following over-winter stubbles.	Strong, but population level effects not investigated.
Reed bunting	Effect of herbicide applications to spring cereal on bird numbers on the following over-winter stubbles.	Strong, but population level effects not investigated.
Turtle dove	**Not investigated**, but change in main diet from weed seeds to crop seeds suggest effects may exist.	Effects considered likely, firm evidence lacking.
Linnet	**Not investigated**, change in main diet from weed seeds to crop seeds suggest effects may exist, but have been compensated for by increased oilseed rape availability.	Population now recovering, but vulnerable to changes in cropping.
Yellow wagtail	**Not investigated**, preference for nesting in potatoes and peas late in season, which all receive a high number of pesticide applications, means effects may be important.	Not been investigated.
Barn swallow	No effect found on chick food invertebrates, or where parents feed.	Current evidence suggests no effect, but sample size low.

Table 3. Quantification of the amount of some food rich habitats required to achieve stable populations for farmland bird species

Species	Limiting factor	Option	Amount required to achieve stable population
Grey partridge	Chick survival	Insect rich habitat e.g. Conservation headlands	5 ha per 100 ha (5 %) of cropped area
Yellowhammer	Over-winter survival	Over-winter stubble availability	15 ha per 100 ha (15 %) in the landscape
Skylark		Over-winter stubble availability	20 ha per 100 ha (20 %) in the landscape

Table 4. The percentage of the population (= % farm area) serviced by ELS options that are likely to affect the key parameter and the level of the key parameter required to result in 1 % annual population growth. If the parameter required exceeds the maximum parameter (i.e. % change from max is positive) then the target is considered unrealistic to achieve. If parameter required exceeds 25 % of the current estimate, the target is defined as 'POSSIBLE'. FPA = fledglings produced per breeding attempt, S_{JV} = juvenile survival, S_{FY} = first-year survival, NA = number of breeding attempts per year. **Source: Vickery et al. (2008)**

(a) Probable effect on key parameter

Species	Key param.	% area affected	Parameter required	% change (baseline)	% change (max)	Achievable target
Kestrel	FPA	31.5	3.8	9	-2	YES
	S_{JV}	33.5	0.28	9	-30	YES
Grey Partridge	FPA	24.1	12.5	45	-14	POSSIBLE
Lapwing	FPA	0				NO
Turtle Dove	NA	1.5				NO
Skylark	NA	10.2	3.4	70	24	NO
Yellow Wagtail*	NA	3.8	5.5	340	175	NO
Starling	S_{FY}	23.4	0.62	70	1	NO
Tree Sparrow	S_{JV}	14.2	0.32	5	-20	YES
Linnet	FPA	1.0				NO
Yellowhammer	S_{FY}	14.2	0.73	38	16	NO
Reed Bunting	FPA	1.8				NO
	S_{FY}	14.2	0.48	17	-14	YES
Corn Bunting	NA	14.9	2.25	80	-25	POSSIBLE
	S_{FY}	14.4	0.82	86	52	NO

(b) Potential effect on key parameter

Species	Key param.	% area affected	Parameter required	% change (baseline)	% change (max)	Achievable target
Kestrel	FPA	34.7	3.78	8	-3	YES
	S_{JV}	35.6	0.27	5	-32	YES
Grey Partridge	FPA	34.8	11.4	33	-20	POSSIBLE
Lapwing	FPA	28.4	1.68	3	-39	YES
Turtle Dove	NA	30.1	2.59	61	-11	POSSIBLE
Skylark	NA	33.6	2.4	20	-12	YES
Yellow Wagtail*	NA	28.2	1.71	37	-14	POSSIBLE
Starling	S_{FY}	33.9	0.54	48	-12	POSSIBLE
Tree Sparrow	S_{JV}	29.4	0.31	2	-13	YES
Linnet	FPA	30.2	2.79	12	-7	YES
Yellowhammer	S_{FY}	32.5	0.61	15	-3	YES
Reed Bunting	FPA	27.0	2.2	11	-19	YES
	S_{FY}	22.7	0.46	12	-18	YES
Corn Bunting	NA	33.6	1.7	36	-43	POSSIBLE
	S_{FY}	29.5	0.62	41	15	NO

Appendix

Table 1. Scientific names for bird species included in text

Common name	Scientific name
Pink-footed goose	*Anser brachyrhynchus*
Greylag goose	*Anser anser*
Red kite	*Milvus milvus*
Sparrowhawk	*Accipiter nisus*
Kestrel	*Falco tinnunculus*
Grey partridge	*Perdix perdix*
Lapwing	*Vanellus vanellus*
Stock dove	*Columba oenas*
Turtle dove	*Streptopelia turtur*
Barn owl	*Tyto alba*
Skylark	*Alauda arvensis*
Barn swallow	*Hirundo rustica*
Yellow wagtail	*Motacilla flava*
Song thrush	*Turdus philomelos*
Chough	*Pyrrhocorax prrrhocorax*
Greenfinch	*Carduelis chloris*
Goldfinch	*Carduelis carduelis*
Linnet	*Carduelis cannabina*
Yellowhammer	*Emberiza citrinella*
Cirl bunting	*Emberiza cirlus*
Reed bunting	*Emberiza schoeniclus*
Corn bunting	*Miliaria calandra*