SESSION 9C

BIODIVERSITY IN ARABLE ECOSYSTEMS

Chairman: Dr K F Walters
CSL, York, UK

Session Organiser: Dr N D Boatman
CSL, York, UK

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Biodiversity in different farming systems

A R Leake
The Allerton Trust, Loddington House, Loddington, Leicestershire, LE7 9XE, UK
Email: aleake@allertontrust.org.uk

ABSTRACT

The nature of arable agriculture is to create conditions conducive to crop growth and to suppress species which are deleterious to this. Conventional, integrated and organic farming systems have similar overall objectives which are achieved using differing means. The effects of various practices, within each system, upon birds, invertebrates, soil fauna and flora are considered.

INTRODUCTION

Farming and biodiversity are not good bed-fellows. Man's agricultural objectives are to concentrate the growing of a single species within a defined area and, in order to increase the production from this area to as far as possible exclude any other organisms that might cause detriment. In so doing many organisms which form part of the food chain within the farmed area, but may not directly affect the crop, are also diminished.

To date world food supplies have been increased by the cultivation and planting of natural and semi-natural habitats and by agricultural intensification. The latter includes nutrient inputs, chemical crop protection, selective plant breeding and development of efficient production and harvesting machinery. The FAO estimates that an annual increase of food production of about 2 to 3% will be needed and this will be met primarily by intensification. This has biological implications since there is generally a positive correlation between the yield of crop plants and their susceptibility to pathogens, pests and weeds. So as yields increase there has to be a commensurate effort to mitigate crop losses, i.e. further intensification. A study in Germany (Oerke *et al.* 1994) showed that a 50% yield increase required a doubling of fertiliser use but an increase in crop protection by a factor of between five and ten. It is this spiralling intensification which has given rise to concerns about the effects upon biodiversity.

Alongside this intensification, specialisation has occurred at a landscape and regional level. The advent of global food trade and efficient transport links means that the necessity to produce food close to the point of consumption no longer exists. Hence within the UK there has been a polarisation of arable land in the east and grassland in the west. As a consequence the east has lost pasture and livestock and the west some of its arable cropping. The associated habitats are also lost and this has been detrimental to a number of species such as lapwing and corn bunting for instance.

FARMING SYSTEMS

Integrated farming

The intensification of agriculture and the widespread use of inputs to control crop antagonists has led to a build up of resistance by some species. The first case of resistance to pesticides was detected in 1914 (van Emden & Peakall, 1996). By 1990 cases of resistance in insects and mites alone exceeded 500, with diseases, weeds, nematodes and rodents all exhibiting symptoms (Schulten, 1990). In some crops, pests were able to multiply in the absence of any effective control and cause complete crop failure (van Emden & Peakall, 1996). In each case the solution was found by a combination of biological control with judicious use of insecticide. Over time this became known as Integrated Pest Management or IPM (Apple & Smith, 1976). This has subsequently evolved to Integrated Crop Management (ICM) where other aspects, such as soil management and crop rotation are considered and ultimately to Integrated Farm Management (IFM) where the wider ecological infrastructure is included in the farm's overall management plan. IFM has been variously defined (El Titi *et al*, (1993), Harwood, (1990); European Commission, (2002)). In general all definitions include economic viability, reduction in pollution and replacement where possible of off-farm inputs. Significantly the concept includes specific reference to biodiversity, which sets it apart from conventional production driven systems.

Organic farming

The increasingly widespread use of synthetic inputs over the past half century has caused concern amongst many who believe that agricultural intensification is environmentally damaging and detrimental to society generally. In 1943 Lady Eve Balfour published a book called The Living Soil (Balfour, 1943), which sought to connect the health of the soil with the health of farming, the environment and society. While the public perception of what organic farming is can be summarised in the phrase 'organic farming is farming without chemicals' (Lampkin, 1990), this is overly simplistic. A better definition is 'organic farmers seek to create integrated, sustainable agricultural systems by relying first and foremost on ecological interactions and biological processes for crop, livestock and human nutrition and protection from pests and diseases' (Lampkin & Arden-Clarke, 1990). This represents a return to cultural, biological and mechanical means of controlling pests, diseases, weeds and providing nutrients to the crop in a way practiced before the introduction of synthetic inputs. As a result, yields can be between 20 and 50% lower than in conventional or integrated systems (Leake, 1999) and as a consequence, prices paid by consumers for organic food are generally higher. The reasons consumers are prepared to pay such premiums are associated with food safety, (including the absence of pesticide residues) and environmental protection, including biodiversity.

BIODIVERSITY IN DIFFERENT FARMING SYSTEMS

While there is obvious agronomic justification for creating species poor systems, current public perception of is generally critical of them. Certain indicators, such as birds, receive a great deal of attention even though population decline is associated with loss of food sources and habitat over which little direct concern is expressed. This paper will consider the evidence published as to the effects of conventional, integrated and organic farming systems upon birds, flora, insects and soil fauna in arable ecosystems.

Farming systems and birds

The British Trust for Ornithology's Common Bird Census (CBC) has recorded substantial declines in many farmland bird species since the 1970's (Fuller *et al*, 1995). Compared to a general fall of 7% in the CBC index overall, farmland birds have shown a decline of 40% (Gregory *et al*, 2000). The steepest declines are associated with a period of agricultural intensification and the interactions occurring are complex. The monitoring of 11 resident bird species in the Boxworth experiment (Grieg-Smith *et al*, 1992) where a 'full-insurance' pesticide regime was compared with a supervised and integrated approach indicated no response by the species concerned to management regimes. Five of the species studied (tree sparrow, blue tit, great tit, starling and wren) showed clear changes during the course of the project but these were apparently not related to the effects of pesticides (Fletcher *et al*, 1992). In occasional incidences the use of pesticides has been shown to increase levels of prey available. Tree sparrows increased their consumption of aphids from around 20% of diet to around 60% 3 days after spraying with a corresponding reduction in the consumption of ground beetles (Hart *et al*, 1992). Biochemical studies of chicks showed increased exposure to the active ingredient as a result of this dietry change but in terms of overall survival the effect was not apparently serious. However, elsewhere the use of broad spectrum insecticides during June halved gamebird chick survival (Potts & Aebischer, unpublished).

Other factors associated with high output agriculture have been shown to affect farmland biodiversity, and in turn to affect bird numbers. These include the depression of botanical diversity in grasslands, the simplification of cropping patterns, the narrowing of the base of wildlife food chains and depression of populations either directly or by reducing their food supply (Campbell *et al*, 1997; Ewald *et al*, 1999; Kirkham *et al*, 1992). Structural changes to the crop canopy may also exert an effect, for example the sowing winter cereals which achieve high tiller survival when treated with nitrogen producing dense crops unattractive to skylarks (Donald *et al*, 1999).

Integrated farming and birds

There is a general lack of evidence as to the effects of IFM techniques on a range ecological indicators.

A recent study which examined 10 research and 32 commercial systems showed very few measuring bird populations. However all systems reported often substantial reductions in the use of inputs and consequential improvements in biodiversity generally (European Commission, 2002). In the UK The Integrated Arable Crop Production Alliance which co-ordinates the research findings from 9 IFM sites recorded a 40% reduction in herbicide and insecticide active ingredient (Anon, 1998). A key component of IFS is a preference for less intensive soil cultivation which tends to leave crop residues, volunteers and weed seeds shed in the previous crop close to the surface. In a split field study in 1996, four species showed strong preferences for direct drilling over ploughing (Saunders, 1999, table 1). Holland *et al*, (1994) showed increases in 13 sets of indicators including beneficial arthropods, birds and mammals, earthworms, soil microbes, no decreases, but noted that in most IFM studies these aspects were not examined.

Table 1. Number of birds visiting between November 1995 and February 1996

	Skylark	Tree sparrow	Chaffinch	Yellowhammer
Ploughed	7	0	0	0
Direct drilled stubble	157	117	35	159

Information regarding the management of the non-cropped habitat, another key component of IFM, is more compelling. A decade of research by The Game Conservancy Trust at Loddington, Leicestershire has shown substantial benefits to a range of farmland birds (Stoate & Leake, 2002). Crop diversity across the landscape has been increased by distributing different crops around the farm rather than the 'block cropping' that is carried out in conventional systems. Field sizes have been reduced by dividing large fields with 20m strips of set-aside which are subsequently sown with wildlife mixtures. Beetle banks, conservation headlands, 2m field margins and hedgerow management plans are also important in the provision of food and shelter. Between 1992 and 1998 non-game birds increased by 42% at Loddington, and for some species, for example, song thrush an increase of 243% was recorded (Boatman *et al*, 2000).

Organic farming and birds

There have been considerably more data gathered on the effects of organic farming upon birds, than the effects of IFM. The results have been highly variable. In the most intensive study 22 farms of organic status including arable, mixed and pastoral enterprises were compared with nearby conventional farms. While bird abundance exceeded that of conventional farms in 50 of 68 individual cases, significance was only established for two (WWF, 2000). Furthermore, much of the variation was attributed to field boundary effects (25% more birds) and to hedgerow height and width, aspects of the landscape which are not necessarily related to the farming system being practised. The benefits of organic farming to birds, in terms of biodiversity and numbers, were only present in one year in three (House of Lords, 1999). However a common trend from this, and a similar Danish study, is that most species were commoner on organic farms in winter, this being particularly true for seed eating birds (Greenwood, 2000). One species which appears to fare particularly well under organic management is the skylark. In the BTO study skylark densities were twice those on conventional farms and breeding success was three times higher. It is difficult to identify the precise reasons but maybe due to a mixture of autumn and spring sown crops provided greater opportunities for second and third nesting attempts by moving between fields. Organic crops are also sparser and may contain a greater abundance of invertebrates (Wilson *et al*, 1997).

Farming systems and invertebrates

The interactions between invertebrate communities vary widely according to crop type and season. The communities are made up of mites, beetles, spiders, flies, springtails and other groups. A long term study by The Game Conservancy Trust in Sussex showed that the overall abundance of invertebrates (excluding mites) showed little change between 1972 and 1989 under conventional farm management (Aebischer, 1991). However 32% of this total was made up of springtails, whose annual density increased significantly during this period, and when these were deducted the invertebrate total showed a significant annual decline of 4.2%, equivalent to halving the abundance over the 20 year period. Such a persistent decline reflects a multiplicity of factors. Effects of a single treatment were well documented in the Boxworth

Experiment where populations of money spiders declined by between 54 and 86% under high rate insecticide usage compared to the modified approach. Ladybirds declined by 91% but ground beetles and soldier beetles were, on average 4% and 24% higher. Results from the SCARAB and TALISMAN projects showed that even following broad spectrum insecticide applications where catches of certain species declined to zero, effects were short-lived with recovery occurring within that season. Some springtail species did not recover however, and recovery had still not occurred two years after all use of insecticides had ceased (Young *et al*, 2001).

A comparison between six split fields practicing conventional and IFM techniques, measuring non-target arthropods, concluded that numbers and diversity varied most between sites, years and crops and least between the two farming systems (Holland *et al*, 1998). This is surprising, given that enhancement of beneficial agents is a central objective of IFM (Stern *et al*, 1959). However an examination of soil management effects on cereal pests at the LIFE Integrated farming trials found large and consistent differences related to tillage methods, with the non-plough tillage methods associated with IFM showing less aphid infection, higher numbers of spiders, but inconclusive regarding beetles. Also, specific measures such as conservation headlands and beetle banks have been shown to increase invertebrate numbers (Sotherton, 1991).

Comparisons of invertebrate numbers found on organic and conventional farms have been generally inconclusive. Research on spider assemblages in organic wheat concluded that organic systems can *potentially* sustain larger and more diverse spider communities (Feber *et al*, 1998). Brooks *et al* (1990) found no difference in total invertebrate numbers, but higher numbers of individual groups such as carabid beetles and dipteran larvae. Research by Moreby *et al* (1994) supports this, with Diptera, Hemiptera and cantharid Coleoptera at higher densities in conventional fields and weevils, spiders, springtails, plant hoppers and sawfly larvae higher in organic crops.

Farming systems and soil fauna

The soil environment provides habitat for a huge range of microbial organisms and is directly affected by the physical intervention of cultivation practice. Changes in the soil physical structure affect the edaphic organisms and this in turn influences chemical processes and crop nutrition. Applications of pesticides in conventional farming systems have shown short-term negative effects on soil microbial activity, with soil type and condition exhibiting greater influence (Young *et al*, 2002). Pesticides also caused some short-lived effects but these were small compared with natural variation found in the earthworm populations. Studies of earthworm populations in Integrated Farming Systems show a strong correlation with reduced tillage (Jordan *et al*, 2000; El Titi, 1995). In organic systems earthworm populations increase during the ley period but decline following ploughing and seedbed cultivations for arable crop establishment. Comparisons of microbial biomass and available P and K in earthworm casts showed levels consistently greater in the conventional compared to the organic system (Neale, 1997). Likewise crops treated with inorganic nitrogen showed great increases in earthworm numbers compared to unfertilised plots, presumably due to an increase in plant biomass. White clover grown to build fertility in organic systems has been found to inhibit earthworm activity (Lampkin, 1990), and where earthworm populations are depleted by major cultivation disturbance uncropped areas such as field margins and beetle banks have been shown to be important as buffering and breeding reservoirs across a wide range of farming systems (Brown, 1999).

Farming systems and flora

Floral species diversity in natural ecosystems is strongly and negatively correlated with fertility, except at the very lowest levels (Hall, 1995). This, coupled with increased use of herbicides in conventional systems over the past 30 years, means that organic farms tend to support greater and more diverse plant communities. A study of plant biodiversity on lowland organic farms recorded five times as much biomass of wild plants in arable fields and 57% more species (WWF, 2000). Moreby *et al*, (1994) observed that three times as many species were present when herbicides were not used, but Greenwood (2000) suggests that in grassland systems the benefits are less than in arable systems, differences are often highly species-specific, and even within species they are not necessarily constant.

A study of ridge and furrow grassland in East Leicestershire which had never received inorganic fertiliser showed the presence of 24 species compared to 7 in the neighbouring conventional farm (Doherty, 1996), giving an indication of the overwhelming effects that nutrients exert upon plant biodiversity. However since much organic land is converted from conventionally farmed land, nutrient levels (in particular P & K) are likely to be much higher than are found in unfertilised land.

DISCUSSION

It is apparent that it is not the farming system *per se* which exerts the greatest influence upon biodiversity but aspects associated with the system that cause the effect. Hence it is the tendency for organic farms to operate diverse crop rotations, incorporate livestock, have both autumn and spring sown crops and demonstrate incomplete weed control that will be beneficial. An IFM system practicing non-inversion tillage is likely to benefit soil fauna and birds. On conventional farms sensitive management of features such as field margins, hedgerows and set-aside can mitage against single species dominance of the cropped area. An increase in the farmed area converted to organic methods or IFM coupled with the implementation of stewardship schemes on conventional farms is likely to increase biodiversity at a national level. The lack of information on the influence of IFM on biodiversity is of concern. Furthermore, we need more information on the mechanisms within each of the systems rather than more system studies and comparison.

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Biodiversity in British agro-ecosystems: the changing regional landscape context

S Petit, D C Howard, S M Smart, L G Firbank
NERC Centre for Ecology and Hydrology, CEH Merlewood, Grange-over-sands, Cumbria, LA11 6JU, UK
Email: spet@ceh.ac.uk

ABSTRACT

Landscape context is important for understanding patterns and dynamics of biodiversity and therefore valuable in the design of effective management strategies. In this paper, we compare the landscape context of the three main lowland agricultural regions of Great Britain. To describe the regional context and its recent change, we selected three sets of indicators relevant to biodiversity and measuring i) the spatial pattern of the mosaic and of individual habitats, ii) the temporal stability of habitats and iii) the ecological condition of habitats. The current state of these indicators and their change over the last 15-20 years were derived from the successive Countryside Surveys (1978, 1984, 1990 and 1998). Our results show that while the three zones share common trends (e.g. trends toward fewer but larger parcels of arable land and towards the eutrophication of the vegetation characteristic of infertile grassland), they do differ in many ways, which calls for a regionalisation of large scale conservation schemes. We discuss the implications of trends in each region for broad-scale conservation of biodiversity

INTRODUCTION

There is an increasing awareness that landscape context is important for understanding biodiversity in agricultural ecosystems (Burel *et al.*, 1998 ; Mazerolle & Villard, 1999; de Blois *et al.*, 2002, Steffan-Dewenter *et al.*, 2002). Landscape structure and dynamics may constrain local species occurrence sometimes to a larger extent than the actual management that operates at the field scale (Le Coeur *et al.*, 1997, Weibull *et al.*, 2001). When designing specific management prescriptions to enhance biodiversity in agricultural landscapes, a good knowledge of the landscape context where a field sits can help in three ways. Firstly, management can be targeted better, by estimating the potential of a field for responding to field-scale management measures; for example, it is possible to estimate what can be expected from a restoration programme in a specific field given the regional landscape level of seed sources and habitat connectivity (Muller *et al.*, 1998). Secondly, large-scale pressures impacting at the field level can be identified, and restoration measures may be designed to mitigate these impacts, e.g. fragmentation of a specific habitat type. Finally, it is possible to evaluate conservation measures in a wider context, where non-designated sites are used as a reference dataset (Carey *et al.*, 2002).

In this paper, we present the large-scale landscape context and its recent dynamics in three lowland agricultural regions of Britain, using indicators describing spatial pattern, temporal stability and vegetation communities. We discuss the implications of differences between regions for conservation.

METHODS

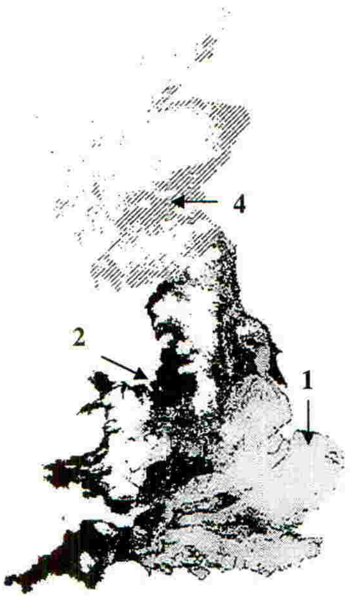

Figure 1. Location of the 3 British lowland environmental zones.

The data were collected during the successive Countryside Surveys of Great Britain (CS), a monitoring programme designed to evaluate national stock and change of land cover, landscape features, vegetation, soil and freshwater. Stock and change are reported for 23 Broad Habitats as defined in the UK Biodiversity Action Plan (Jackson, 2000). Detailed methodology and results can be found at www.cs2000.org.uk and Firbank (in press). The ground survey consists of 1 km square samples randomly selected, stratified according the 40 Land Classes of the ITE Land Classification (Bunce *et al.*,1996). The same squares are visited repeatedly; vegetation was recorded in 1978 (256 squares), 1990 (508 squares) and 1998 (569 squares), landscape features and habitats were recorded in 1984 (384), 1990 (508 squares) and 1998 (569 squares). Results can be presented at the Land Class level (40) or aggregated into the six Environmental Zones that divide GB into regions that have similar climates and topographies

In this paper, we are comparing the three zones dominated by lowland agriculture (figure1): the Easterly Lowlands of England and Wales (EZ1), the Westerly Lowlands of England and Wales (EZ2) and the Scottish Lowlands (EZ4).

To describe the zones and how they have changed over the last decades, we estimated from CS data three sets of indicators that are recognised to be relevant to biodiversity in agricultural landscapes. The first set of indicators describes the spatial pattern of the landscape mosaic found in the 1 km squares as well as the pattern of individual habitat types. These measures are calculated for 1998 and change between 1984 and 1998 was estimated using squares surveyed in both periods. Heterogeneity (or diversity) and equitability (or evenness) refer to diversity indices of the mosaic. The second indicator measures the temporal stability of semi-natural habitats, mainly the origin of 1998 semi-natural habitats and the fate of 1984 semi-natural habitats. The third set of indicators are scores of species assemblages recorded in vegetation plots that reflect the ecological condition of habitats (Firbank *et al.*, 2000). We will present here stock and change in Ellenberg fertility scores (Hill *et al.*, 1999), overall species richness and species richness of food plants for bird and butterflies (Smart *et al.*, 2000).

RESULTS

Spatial pattern

Composition of the landscape

EZ1 is strongly dominated by the Arable Broad Habitat, which has remained stable in area between 1984 and 1998. Improved grassland has decreased while Woodland has increased (Table 1). EZ2 and EZ4 are both mixed agricultural landscapes dominated by Improved grassland, and exhibit similar trends between 1984 and 1998.

Table 1: The landscape composition in 1998 and its change between 1984 and 1998 in the three Environmental Zones. All figures are percentage of the total area. Bold = significant change ($p < 0.05$).

	Stock 1998			Change 1984-98		
Broad Habitat	*EZ1*	*EZ2*	*EZ4*	*EZ1*	*EZ2*	*EZ4*
Arable	53	22	22	-1	-3.7	-6
Improved grassland	20	40	29	**-13**	**-9**	-10
Woodland	10	10	13	**10**	3	27
Neutral grasslands	2	3	5	23	56	**87**
Other grasslands	+	1	3	-7	-23	**-33**
Other semi-natural habitats	1	4	14	225	11	-2

Spatial characteristic of the mosaic

EZ1 is more homogeneous than EZ2 and EZ4 with fewer habitat types and a dominant habitat type (Arable) taking up more area (Table 1). It became more diversified between 1984 and 1998 both as a result of an increase in the number of Broad Habitat types (Table 2) and a (non-statistically significant) decrease in the average area of the Arable habitat.

The heterogeneity and equitability of the landscape mosaics increased significantly in EZ2 and EZ4 but the number of habitat types has remained stable (Table 2). This indicates that habitat types are more equally distributed within 1 km squares, as shown by the significant decrease of area taken by the dominant habitat in EZ2, Improved grassland (Table 1).

Table 2: The spatial pattern of the mosaic of habitats within sample 1 km squares in 1998 and its change between 1984 and 1998. Bold = significant change

	Stock 1998			Change 1984-98		
	EZ1	*EZ2*	*EZ4*	*EZ1*	*EZ2*	*EZ4*
Heterogeneity (Shannon)	1.3	1.8	1.9	0.04	**0.08**	**0.1**
Equitability	0.29	0.37	0.41	0.01	**0.02**	**0.02**
Number of broad habitats	7	8	9	**0.33**	0.25	0.1
Average area per habitat (ha)	13	12	11	**-1.05**	**-0.49**	-0.25
Area of dominant habitat (ha)	62	52	51	-0.29	**-3.27**	-2.5

Spatial organisation of individual broad habitats

The average size of parcels of Arable habitat in 1998 was the highest in EZ1, more than twice that in EZ2 (Table 3). There were also fewer parcels per square in EZ2 and EZ4. Although the total area of Arable was stable in the three zones between 1984 and 1998 (Table 1), there was a decrease in the average number of parcels per square and an increase of the average size of parcels per square (significant only in EZ1), indicating homogenisation. Analysis of the average parcel size of Arable per square in 1984, 1990 and 1998 by class of parcel size showed that in EZ1, the increase resulted from a loss of squares with average parcel size below 2 ha and a gain of squares with average parcel size around 4 ha. In EZ2, the frequency distributions of parcel size were similar for the sample periods, with a mode at 2-3 ha. In EZ4, a clear shift was observed in the distribution of average parcel size between 1984, where most squares had an average size of parcels of 1 ha, to1998 where the average size had increased to 2.5 ha.

There was a loss of the total area of Improved grassland throughout the zones between 1984 and 1998 (Table 1). In EZ1, this trend meant a parallel decrease in the number of parcels and in the average size of parcels per square (implying habitat fragmentation). In EZ2 and EZ4, the decrease in parcel number occurred in parallel with an increase in their average size (implying homogenisation). In EZ4, there was a notable increase in the frequency of squares with large average parcel size (>1.5ha).

Neutral grassland increased in total area in all 3 zones, although usually not to a statistically significantly extent (Table1). In term of spatial pattern, parcels tended to be larger in EZ1 than in EZ2 and EZ4. The trend between 1984 and 1998 was an increase of both the number of parcels and their average size per square (de-fragmentation), a trend highly significant in EZ2 and EZ4 (Table 3).

Table 3: Spatial pattern of the Arable, Improved grassland and Neutral grassland Broad Habitats in 1984 and in 1998 estimated on repeat squares only. Bold indicates a significant change between 1984 and 1998.

	Stock 1998			Change 1984-98		
	EZ1	*EZ2*	*EZ4*	*EZ1*	*EZ2*	*EZ4*
Arable						
Number of parcels per km^2	20.5	12.4	11.7	**-5.1**	**-2.3**	**-7.1**
Average size of parcels (ha)	3.1	1.3	1.7	**0.7**	0.13	0.27
Improved grassland						
Number of parcels per km^2	20.0	47.4	30.8	**-4.4**	**-10.9**	**-10.4**
Average size of parcels (ha)	0.89	0.93	0.95	-0.04	**0.1**	**0.18**
Neutral grassland						
Number of parcels per km^2	8.3	9.8	18.3	0.38	**2.9**	**5.3**
Average size of parcels (ha)	0.40	0.31	0.27	0.15	**0.1**	**0.1**

Temporal stability of the landscape elements

Analyses of flows between broad habitat types gives a more complete picture of changes that are occurring than the resulting net change. Substantial areas recorded in 1998 as 'semi-natural' habitat had been used as intensive agriculture in 1984 (Figure 2). This situation was especially true in EZ1 where about 40% of existing areas of semi-natural habitat originated either from Arable or Improved grassland in 1984. The fate of 1984 semi-natural habitats confirmed this high instability in EZ1, as only 65% of the 1984 area of semi-natural habitat was carried over to 1998. Temporal stability was the highest in EZ4, where 80% of the area of semi-natural habitat had been so since 1984 and 80% of the 1984 stock of semi-natural habitat had remained so. Not surprisingly, the turnover mostly occurred between semi-natural grasslands and intensive agriculture. The stability of semi-natural grasslands between 1990 and 1998 was especially low in EZ1 and EZ2, where about 50% of neutral and acid grasslands were converted to intensive agriculture. By contrast, 95% of acid grassland found in 1990 in EZ4 was of the same habitat type in 1998.

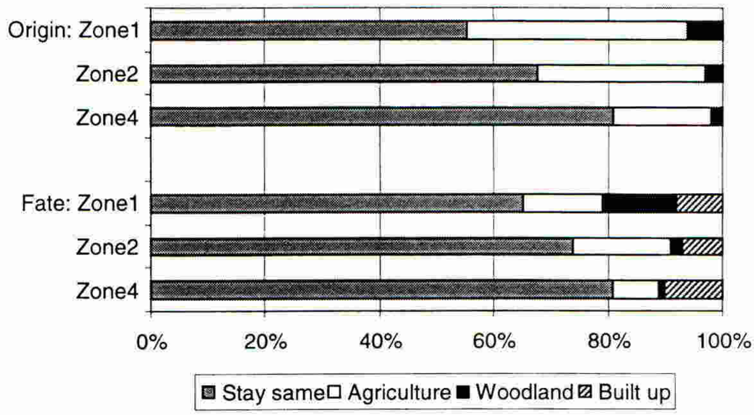

Figure 2: Origin in 1984 of 1998 semi-natural habitats and fate of 1984 semi-natural habitats in 1998. Semi-natural (Neutral, Acid and Calcareous grassland, Bracken, Dwarf shrub heath, Bog, Fen marsh swamp). Intensive agriculture (Arable, Improved grassland).

The condition of habitats

Analysis of the vegetation indicators in the 3 zones showed differences in trends between 1978 and 1998. EZ1 was characterised by a decline in the quality of the vegetation types found in agro-ecosystems, whether within crops, field boundaries or grasslands (Table 4), with a significant decline of species richness, and/or food resources for farmland birds and butterflies. In this zone, the vegetation of both fertile and infertile grassland has become more characteristic of more fertile habitats, suggesting a degree of eutrophication between 1978 and 1998. By contrast, the same vegetation types found in EZ2 and EZ4 exhibited little change between 1978 and 1998. Plant communities found in crops and field boundaries did not change significantly over that period but there was a decline in species richness and food resources for animals for fertile grassland in EZ4. A trend common to all 3 zones was the decline in the quality of vegetation typical of infertile grasslands where there was both a significant increase in Ellenberg Fertility scores and a decrease in species richness overall and for animal food species. Figure 3 shows the frequency distribution of plots for classes of species richness in the 3 zones for 1978, 1990 and 1998. There was an increase in the frequency of plots with fewer plant species and in EZ2 and EZ4, there was a clear shift of the mode of the distribution towards fewer species between the 3 periods.

Table 4: Significant changes between 1978 and 1998 in the 4 condition indicators for the main aggregate classes of the Countryside Vegetation System (repeat plots). N = Ellenberg fertility score, SR = Species richness, BIFP = Bird food plant, BUFP = Butterfly food plant.

Aggregate class	**EZ 1**	**EZ2**	**EZ 4**
Crops and weeds (cultivated grounds)	Decline SR, BUFP	Increase BUFP	-
Tall grass and herb (field edge)	Decline BIFP, BUFP		
Fertile grassland (improved and intensively managed)	Increased N Decline BIFP, BUFP	-	Decline SR, BIFP, BUFP
Infertile grassland (unimproved)	Increased N Decline SR, BIFP, BUFP	Increased N Decline SR, BIFP, BUFP	Increased N Decline SR, BIFP, BUFP

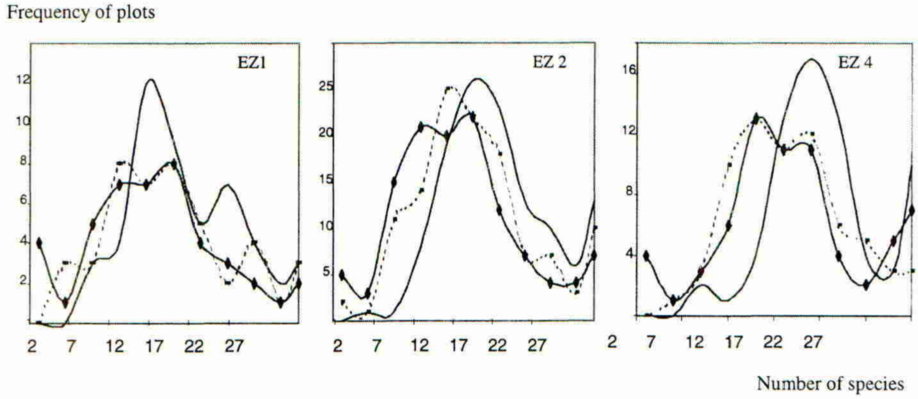

Figure 3: Frequency distribution of vegetation plots in classes of species number presented per zone for 1978 (black line), 1990 (dotted line) and 1998 (diamonds).

DISCUSSION

This paper shows that the 3 main lowland agricultural zones of Britain, although sharing common trends, are also experiencing different processes (Table 5).

In EZ1, parcels of arable land were much larger than in the other zones and their average size had been increasing recently while improved grasslands appeared to get increasingly fragmented. Vegetation quality had been declining throughout all the main habitats present in the agricultural landscape, although it should be noted that changes seemed to have been reversed in cultivated parcels between 1990 and 1998 (Haines-Young et al, 2000). There seems to have been a general shift towards decreasing suitability for maintaining biodiversity in agricultural habitats over the last 15-20 years. Two issues that should specifically be addressed in EZ1 are the fragmentation of the Improved grassland habitat and the general decline of vegetation quality, notably in grassland where it is occurring in parallel to habitat eutrophication. The two mixed landscapes, EZ2 and EZ4, shared many trends. Although there is an increased heterogeneity of the landscape mosaic as a whole (higher representation of less dominant habitat types), the homogenisation in the spatial pattern of parcels used for intensive agriculture (fewer and larger parcels) appears to be an issue in both zones. If the total area of Improved grassland keeps decreasing, homogenisation could lead to increased isolation between habitat patches, at least locally. However, despite homogenisation and the drastic loss of field boundaries between 1984 and 1990 in EZ2 (Firbank, in press), we detected no signal of a significant decline in the quality of vegetation found in crops, field edges or improved grassland in this zone. The situation is less clear in EZ4 where the quality of grasslands appears to be at risk, despite a relatively low turnover.

Table 5: Summary of trends 1984-1998 (1978-1998 for vegetation) for the selected indicators

Indicator	**EZ1**	**EZ2**	**EZ4**
Arable - spatial pattern	Total area stable + Homogenisation		
Neutral grassland - spatial pattern	Increase in total area + Diversification		
Mosaic - spatial pattern	Diversification (more habitats)	Increase in heterogeneity (more parcels of the same habitats)	
Improved grassland – spatial pattern	Decrease in area + Fragmentation	Decrease in area + Homogenisation	
Stability of semi-natural habitats	Low	High	
Vegetation quality	Decline in all vegetation types	Decline in infertile grassland	Decline in fertile and infertile grassland

These results show that there are large-scale processes that continue to influence landscape structure and vegetation, some of which are acting at a regional scale rather towards a more generic 'lowland agricultural landscape'. Any attempts to conserve biodiversity across agricultural landscapes should take these trends into account. In particular, "broad and shallow" agri-environment schemes, aimed at the management of large numbers of whole farms at a low cost per unit area (Policy Commission, 2002), should be aimed at controlling vegetation eutrophication throughout the farmed landscape (and especially on infertile grassland, whether in whole fields or along field boundaries), to control the fragmentation of improved grasslands in EZ1, and to reduce the homogenisation of the spatial pattern of arable and improved grasslands in the mixed landscapes. Should the trends we have observed continue unchecked, the landscapes will become more coarse-grained, reducing their value to many species that require fine-scale habitat mosaics, and their aesthetic character.

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Indirect effects of pesticides on breeding yellowhammers *Emberiza citrinella*

A J Morris, R B Bradbury
The Royal Society for the Protection of Birds, The Lodge, Sandy, Beds, SG19 2DL, UK
Email: tony.morris@rspb.org.uk

J D Wilson
RSPB, Dunedin House, 25 Ravelstone Terrace, Edinburgh, EH4 3TP, UK

ABSTRACT

In the UK, there is little evidence of direct lethal or sub-lethal effects of agricultural pesticides on birds. However, an indirect effect via the removal of food by pesticides may be a factor in population declines of several species associated with arable farmland, including the yellowhammer *Emberiza citrinella*. Using data collected on lowland farmland between 1995 and 2001, we show that pesticide use in the breeding season influences foraging behaviour and breeding performance. Before the end of June, when chicks are reliant on invertebrate food, birds rarely foraged in arable crops recently sprayed with insecticide. Chick body condition was negatively related to insecticide use in fields adjacent to the nest and there was a trend for chick mortality to be greater in nests situated next to fields sprayed with herbicide in summer.

INTRODUCTION

The last three decades have seen changes in many aspects of agricultural practice in the UK, reflecting policy and technology-driven increases in agricultural productivity. An increase in the use of pesticide has been prominent among the changes, with the number of product applications of fungicide, herbicide and insecticide all having risen dramatically during the past 30 years. A number of farmland bird species have declined in population or range at the same time as farming has intensified.

It is well known that many pesticides widely used in the 1960's and 70's, such as the organochlorine DDT, caused declines in birds, notably raptors such as sparrowhawk (*Accipiter nisus*), through poisoning or reduced fecundity of adults (Newton, 1995). When applied according to guidelines, there is little evidence that newer types of pesticides are currently having significant population effects via direct toxicity.

However, there has been increasing evidence that pesticides may be having an indirect effect, via the depletion of food supplies. Such an effect has been experimentally proven for the grey partridge (*Perdix perdix*) (Rands, 1985) and suggested by autecological studies of other species both in this country and North America. Here, we examine how behaviour and reproductive performance of yellowhammers (*Emberiza citrinella*) breeding on arable farmland varied with different pesticide regimes.

METHODS

Bird recording and invertebrate sampling

Yellowhammers were studied during the breeding seasons (May-August) of 1995-1997 and 1999-2001 on mixed grass/arable farms in Oxfordshire (n = 7 sites) and Lincolnshire (n = 1 site). Field observations of foraging behaviour were obtained for nests containing chicks aged ≥4 days old. All foraging visits to fields adjacent to the nest were recorded during one to three watches, each lasting 60-90 minutes. Breeding performance (chick body-condition, chick growth rates and chick survival) were measured using standard techniques outlined by Bradbury *et al.* (in press). To assess the impact of pesticide spraying on nestling-food resources, we vacuum sampled invertebrate populations in cereal fields adjacent to a subset of yellowhammer nests. Sampling was confined to cereals, as they comprised 70% of arable crops adjacent to nests and in order to minimise the degree to which differences in vegetation structure between crops influenced the efficacy of the catch. Samples were taken at <10m, 20m, 40m, 80m and 160m into the crop from the field boundary containing the nest.

Pesticide data

Conventional pesticide treatments were used on all study sites and were applied to 95% of arable crops. From the data supplied by farmers, two measures of pesticide use were derived for inclusion in our models.

1. The total number of product applications of fungicides, herbicides and insecticides (counting each repeat application of the same product and each product applied as a mixture in the same spray tank as different) for each arable field.
2. The time period of the last pesticide application prior to egg-hatch for each field and pesticide type. Time periods were initially expressed as 'no' (untreated during that crop-year), 'winter' (late August-February) or 'summer' (March-early August) applications.

On our study sites, the number of applications were fewer than the national average for insecticide sprays on winter barley crops, herbicide sprays on spring barley crops and fungicide sprays on rape, winter barley and wheat crops. In no case was the number of applications on the study sites significantly greater than the mean national average.

Analyses

Analyses were performed using the General Linear Mixed Modelling (GLMM) procedure in Genstat 5, which allowed for random factors to be included in the models to account for variation between sites (Welham, 1993). All analyses used a step-up procedure in which the Minimum Adequate Model (MAM) was attained when no variables could be added to the model without causing a significant change in deviance at the $P = 0.05$ level.

Tests of the effects of pesticides were made on invertebrate food supplies (n = 6662 individual invertebrates from 28 cereal fields adjacent to yellowhammer nests), yellowhammer foraging (n = 127 nests) and chick body condition (n = 122 broods), growth rates (n = 54 broods) and survival (n = 130 broods). For the tests of effects of pesticides on invertebrates and foraging, data were modelled using Poisson errors. In the foraging analysis, the number of visits from a nest to each available field was specified as the response variable and log field area as an offset, such that the number of foraging visits per unit area of each field equated to the 'foraging density'. Variations in chick body condition and growth rate were modelled using normal errors. Binomial models were used to find which variables explained a significant proportion of nestling mortality arising from starvation.

Yellowhammer nestling diet on British arable farmland includes cultivated cereal grain, once it becomes semi-ripe. As pesticide applications do not adversely affect the supply of grain, it is expected that pesticides will be less influential in determining foraging patterns of birds exploiting this food-resource. Therefore, foraging data were divided into two groups (termed 'grain available' and 'grain unavailable'). To make this division, we reviewed data on 1362 identifiable food items recorded during nest observations. The earliest dates recorded for chicks being fed barley grain were in the third week of June, with wheat (which ripens later) being frequently fed from the second week in July. Thus, the thresholds used for the availability of edible grain were June 21st in the case of barley and July 7th in the case of wheat. On this basis, 66 nests were classified as 'grain unavailable' and 61 nests as 'grain available'.

RESULTS

Effects of pesticides on invertebrate chick food

Invertebrate variables were consistently significantly lower in fields treated with insecticide during the summer than in fields with no or winter-only insecticide applications. There was weaker and less consistent evidence for effects of fungicides and herbicides (Table 1).

Table 1. Effect of pesticides in yellowhammer chick-food Minimum Adequate Models: + = positive effect; - = negative effect; $\cap$ = curvilinear effect (slight initial positive effect with strong negative tail); *ns* = non significant. Differences between factor levels for insecticide use are ranked from greatest to least abundance. '>' signifies the order of ranking and '>>>' represents significant differences between adjacent ranks at $P < 0.01$.

Variable	variable type	all invertebrates in diet	invertebrates important in diet	invertebrates ≥5mm	invertebrate biomass
date	continuous	+ (P = 0.044)	*ns*	*ns*	+ (P = 0.016)
# fungicides	continuous	- (P = 0.006)	*ns*	*ns*	*ns*
# herbicides	continuous	*ns*	*ns*	$\cap$ (P = 0.012)	*ns*
insecticide timing	factor: 3 levels 1 = no spray in crop year; 2 = winter only spray; 3 = summer spray	$P < 0.001$ 2>>>1>>>3	$P < 0.001$ 2>1>>>3	$P < 0.001$ 2>1>>>3	$P < 0.001$ 2>1>>>3

Effects of pesticides on foraging behaviour

For 'grain unavailable' nests, fields that received 'summer' insecticide inputs were used significantly less than fields that received 'no summer' insecticide. Foraging density in fields that received no summer applications of insecticide was nearly four times higher than in fields with summer applications (Figure 1). An examination of the raw data showed that it was rare for adults to forage more than once in fields with summer insecticide inputs (9.5% of such fields, all of which were adjacent to a single nest), compared to fields without summer insecticide inputs (32.4% of such fields). For 'grain available' nests, 'summer' insecticides did not significantly affect foraging patterns. The timing of fungicide or herbicide applications or the number of applications of any of the pesticide predictors did not significantly affect foraging patterns in either period.

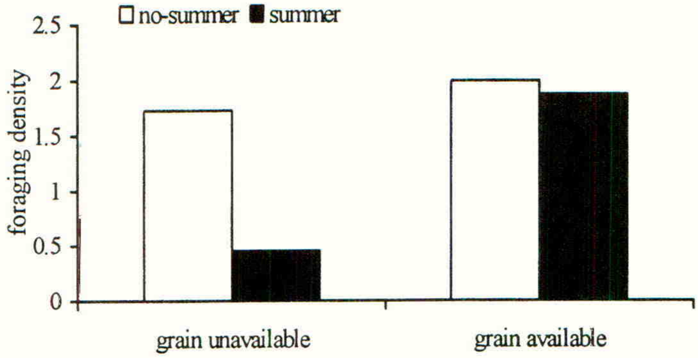

Figure 1. Predicted values (from model) of Yellowhammer foraging density per unit area of fields for 'grain unavailable' and 'grain available' nests with 'summer' and 'no-summer' insecticide applications.

Effects of pesticides on chick performance and survival

Chick body condition was poorer in nests exposed to more than one insecticide application and was very poor in nests with more than two applications (Figure 2).

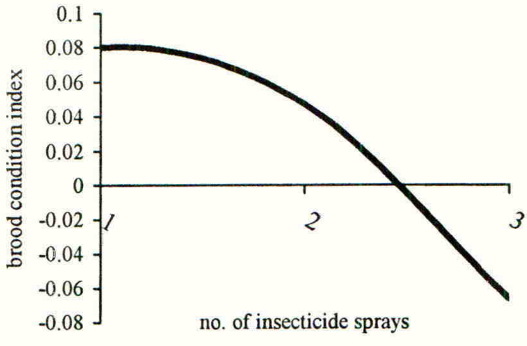

Figure 2. Predicted values (from model) of Yellowhammer chick-condition, with increasing numbers of insecticide applications.

No evidence was found of any effect of pesticides on chick growth rates. There was a trend (non-significant in the GLMM) for greater chick mortality in broods next to fields sprayed with herbicide during the summer.

DISCUSSION

During the period when yellowhammer nestlings are entirely dependent on invertebrate food, our results suggest an indirect effect of insecticides on yellowhammer foraging behaviour, brought about by the depletion of invertebrate chick-food resources. Multiple insecticide applications also resulted in poorer chick condition.

The only recorded multiple foraging visits to summer insecticide treated fields occurred 1-2 days after the application and could suggest this pair were feeding on moribund invertebrates. In cases where the fields treated with summer insecticide were seldom or never foraged in, much greater periods had elapsed since spraying and this potential source of food would no longer be available. Alternatively, differences in the types of insecticides applied could account for the differential in foraging. Pirimicarb, an aphid-specific carbamate, had been used on the fields with multiple foraging, while broad-spectrum pyrethroid or organochlorine products were used on other fields.

Disruption of foraging patterns may have contributed to poor chick condition in broods exposed to multiple insecticide sprays. Provisioning rates of nestlings are likely to drop if adults undertake longer foraging flights to gather sufficient food, and, in the worst-case scenario, starvation may result if alternative food-rich foraging habitats are not available within reach of the nest. This study did not find any effect of insecticide use on nestling starvation. However, this does not necessarily mean that insecticides have no impact at the population level, as chicks leaving the nest with poor body condition are less likely to survive to reproductive age. There was a trend for increased starvation in broods exposed to 'summer' herbicide use. As yellowhammer chicks are rarely fed wild plant material in the UK, any such effect is likely to arise by herbicides indirectly reducing invertebrate numbers via the removal of their host plants.

As indicated, pesticide-induced depletion of invertebrate food is the likely determinant of the observed foraging patterns and poor chick performance. However, due to the fact that aphid numbers on Oxfordshire farms were very low during the duration of the study (and access constraints imposed by the outbreak of Foot and Mouth virus), the number of cereal fields sprayed in summer with insecticide that were also sampled for invertebrates was small (n = 6), with five of these fields being on one farm. Given these limitations, the results do indicate a negative effect of 'summer' insecticide use on the abundance of invertebrate taxa that are 'present' or 'important' in yellowhammer diet, many of which are not usually considered agricultural 'pests' (e.g. Arachnida, Coleoptera: Staphylinidae, and Hymenoptera).

Implications for conservation

These results suggest that insecticides are the type of pesticide most likely to affect yellowhammers and that the timing of application may be more important than the cumulative effects of repeated spraying. It should be reiterated that the use of certain classes of pesticides on the study sites was significantly less than the national average and therefore the results may actually under-represent the potential impact of pesticides on invertebrate food and birds. The negative relationships between insecticide treatments and invertebrate chick-food abundance, foraging behaviour and chick performance closely parallel results obtained by studies of similar species, e.g. chestnut-collared longspur (*Calcarius ornatus*) (Martin, *et al.*, 2000) and corn bunting (*Miliaria calandra*) (Brickle, *et al.*, 2000).

Careful consideration should be made before applying insecticides to crops during the breeding season. Insecticide treatments during the winter months appear to be far less disruptive to parental foraging patterns in yellowhammers, and probably to other farmland bird species with similar ecologies. With a decline in cereal grain prices from the mid 1990's and the spread of cost-effective integrated farming techniques, the use of summer sprays has reduced in recent years, with prophylactic spraying largely replaced by treatment only when pests exceed thresholds values. Our evidence suggests that a more widespread adoption of targeted farming systems would be likely to benefit breeding birds.

With mortality among non-target invertebrate food taxa often-approaching 100% seven days after spraying with broad-spectrum products, and with substantial effects persisting for up to two months after the application (Sotherton, 1990), less damaging target-specific insecticide products should be identified and their use encouraged.

Alternative unsprayed foraging habitats, such as grass-margins, can also be provided as a mitigating measure under the Countryside Stewardship Scheme. However, accessibility (determined by vegetation architecture), as well as abundance, of food-resources is likely to be a major factor influencing foraging site selection (Morris, *et al.*, 2002), and further investigation on what constitutes optimal foraging conditions is still needed.

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Increasing the Government's Farmland Bird Index through conservation management at the farm scale: a ten-year demonstration

C Stoate
The Allerton Research and Educational Trust, Loddington, Leicestershire LE7 9XE, UK
Email: chris.stoate@ukonline.co.uk

ABSTRACT

Conservation management implemented within agri-environment schemes to reverse the decline in a Farmland Bird Index (FBI) of twenty species has already been adopted on a Leicestershire farm where twelve of these species have been monitored over a ten-year period. Overall abundance of the FBI species present increased rapidly in response to conservation management, but there was considerable variation between species. The Government's Biodiversity Action Plans (BAP) set conservation targets for species of greatest conservation concern, including species not included in the FBI. This paper suggests that these provide an important additional indicator of successful systemic application of whole farm management.

INTRODUCTION

In response to widespread and sustained declines in populations of birds and other wildlife associated with farmland habitats, the UK Government instigated a number of conservation measures within agri-environment schemes under EU Regulation 1257/99. Increasingly, farmers are encouraged to take a 'whole farm approach' to sustainable management of farmland. As one of its fifteen 'headline indicators' of sustainability (Quality of Life), the Government has selected 105 bird species as being indicative of sustainable development (DETR, 1999). Of these, 20 comprise the Department of Environment, Food and Rural Affairs' (DEFRA) Farmland Bird Index (FBI). The Government's Public Service Agreement (PSA) aims to 'care for our living heritage and preserve natural diversity by reversing the long-term decline in the number of farmland birds by 2020, as measured annually against underlying trends'.

The Common Birds Census (CBC), organized by the British Trust for Ornithology (BTO) provides historical data on changes in bird abundance while data derived from the BTO's Breeding Bird Survey (BBS) continue to provide an annual index of abundance for these species. The Farmland Bird Index has fallen by around 40% since the 1970s'. However, species have varied in their rates of population decline and are likely also to vary in the rate at which they respond to conservation management. In fact, the index includes some species that are already increasing. Three of the species (Woodpigeon, Rook and Jackdaw) are commonly perceived by farmers as pests. This may influence farmers' willingness to encourage these species. Farmers regard other species, such as grey partridge and skylark, more positively. Eight of the 20 species are identified as Biodiversity Action Plan (BAP) species because of conservation concern arising from their recent severe population declines. Targets are set for these species individually. Thirteen of the twenty-six UK BAP species are strongly associated with farmland habitats (Swash *et al.*, 2001).

An important mechanism for restoring farmland wildlife in England is the Government's Countryside Stewardship (CS) scheme. This includes a range of management options designed to meet the ecological requirements of many farmland birds throughout the year. Such options include the maintenance of winter stubbles and fallows, low-input spring-sown crops, and the creation of seed-bearing crop mixtures as food in winter. Other options comprise perennial grass habitats in field boundaries and in the form of 'beetle banks' (Thomas *et al.*, 1991), 'conservation headlands' (Sotherton, 1991) to provide invertebrate food in summer, and the management of hedges and other farmland habitats. Different bird species are likely to respond in different ways and at different rates to such management.

The 'whole farm approach' incorporates associated habitats such as farm woods, ponds and streams. Such an approach has been developed and implemented at the Allerton Project research and demonstration farm (Loddington) in Leicestershire since 1993. Bird numbers have been monitored since 1992. This provides an opportunity to preview the potential response of Farmland Bird Index species to conservation management, including that now being encouraged with Government support across the country.

METHODS

Loddington is a mixed arable and livestock farm of 333ha. Conservation management was implemented in the form of a wild game management system and therefore included predator control during the nesting season, and supplementary feeding during the winter, as well as habitat management options that are now available under Countryside Stewardship. Nest boxes were provided specifically for Tree Sparrows in 1999. The conservation management system was implemented in 1993 and is described more fully by Stoate and Szczur (2001a) and Stoate (2002). Bird monitoring at Loddington started in 1992 (baseline year) and took two forms: annual transects for all species (11.5 km in length, repeated four times between April and July) and territory mapping for all breeding species except woodpigeon and jackdaw in 1992, 1998 and 2001 (Stoate and Szczur, 2001a, Stoate, 2002).

Some of the 20 FBI species were not present a Loddington (Corn Bunting, Lapwing and Barn Owl), were present at densities that were too low to record using transects (Grey Partridge, Turtle Dove, Kestrel, Starling and Tree Sparrow), or were not reliably surveyed using this method because of their flocking behaviour (Rook). Results for the remaining 11 species are described in this paper. Each species is allocated to one of the categories, BAP species, pest species, or other FBI species. The additional territory mapping data include counts for Tree Sparrow.

RESULTS

Transects

The composite FBI for Loddington increased during the period of conservation management by 6.58% per year (Figure 1), but differed between the three groups. Mean annual increases were greater for pest species and 'other' FBI species than for BAP species (Table 1), and pest species were the most abundant (Table 1, Figure 2).

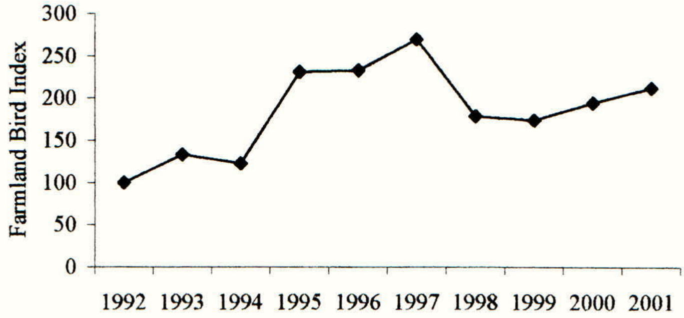

Figure 1. Changes in the Farmland Bird Index calculated from data for the 11 species recorded at Loddington.

Table 1. Changes in bird abundance derived from transect data.

Category	Species	% transect sample in 2001	Mean annual % change
Biodiversity Action Plan species	Skylark	3.7	
	Linnet	9.4	
	Reed Bunting	<0.1	-0.97
Other species	Stock Dove	<0.1	
	Whitethroat	6.5	
	Yellow Wagtail	<0.1	
	Greenfinch	8.1	
	Goldfinch	2.6	
	Yellowhammer	6.3	9.71
Pest species	Jackdaw	3.3	
	Woodpigeon	58.5	9.28

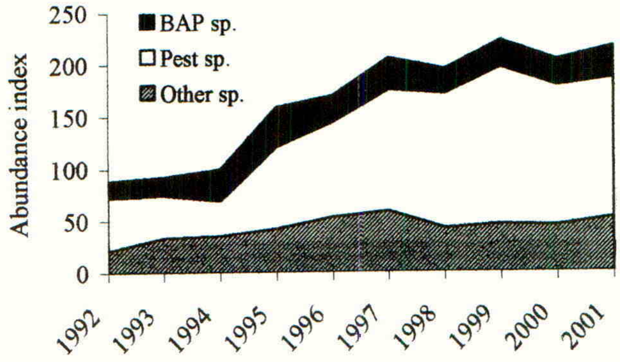

Figure 2. Changes in abundance of Biodiversity Action Plan species, pest species and remaining Farmland Bird Index species recorded at Loddington, following implementation of conservation management in 1993.

Territory mapping

BAP species increased in abundance over the study period. This increase was attributable to increases in Linnet and Tree Sparrow alone (Figure 3). Reed Bunting and Skylark numbers remained stable. 'Other' species also increased in abundance, most of this change being attributable to Whitethroat, Greenfinch and Goldfinch, with little change in numbers of the other species (Figure 4).

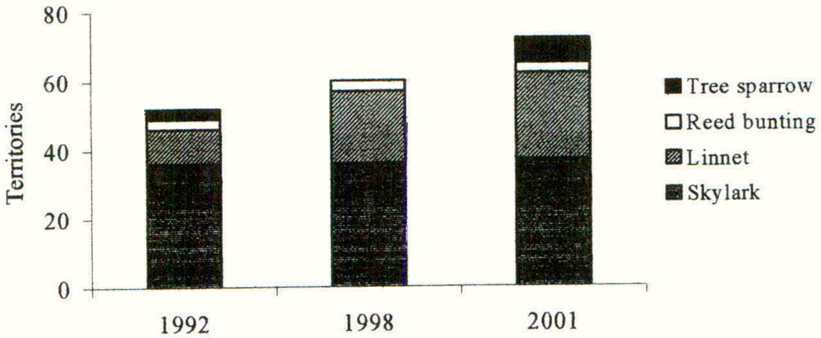

Figure 3. Changes in numbers of breeding territories of Biodiversity Action Plan species within the Farmland Bird Index at Loddington (territory mapping data).

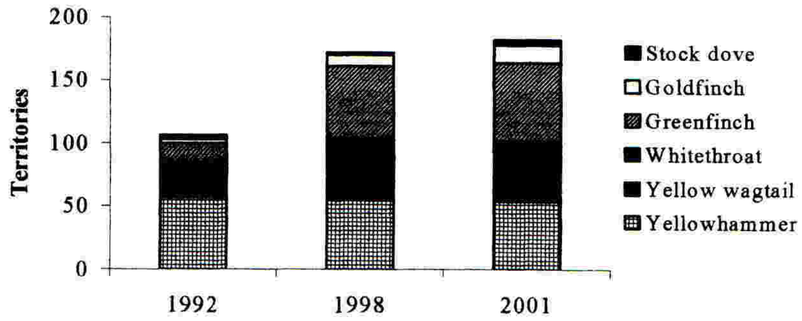

Figure 4. Changes in numbers of breeding territories of 'other' bird species within the Farmland Bird Index at Loddington (territory mapping data).

DISCUSSION

These results indicate that it is likely to be possible to restore breeding numbers of some FBI species at the farm scale through implementation of a whole farm management system. As well as management practices included within the Countryside Stewardship scheme, the conservation system at Loddington included predator control and supplementary feeding in winter and the contribution of either of these to increases in breeding populations is not known. The role of predator control is currently being tested, by ceasing it as from autumn 2001, and continuing the annual monitoring of breeding abundance.

CS agreement holders will include farms where predator control and winter feeding are already being practised for the purposes of game management, and conservation benefits may most closely resemble those at Loddington where this is the case. Some individual management options within CS are known to have a direct impact on breeding abundance of species not influenced by predator control or winter feeding (e.g. Whitethroat (Stoate & Szczur, 2001b)). In the case of Tree Sparrows, winter numbers increased in response to provision of winter food and cereal wildlife seed mixture crops, but breeding numbers did not increase until nest boxes were also provided.

Although these results indicate that overall abundance of Farmland Bird Index species could increase in response to conservation management such as that implemented under CS, there is likely to be considerable variation between species, according to their differing requirements.

At Loddington, BAP species within the Farmland Bird Index increased less than 'other' FBI species, suggesting that those species that are recognised by the Government to be most in need of conservation may be the slowest to respond. However, Linnet and Tree Sparrow responded to conservation management, and Skylark was already present at high density at the start of the project because of the presence of set-aside at Loddington (not present on neighbouring farms at the time, where densities were lower (Stoate & Szczur, 2001)). Only Reed Bunting was present in low numbers and failed to respond to the conservation system. BAP species not included in the Farmland Bird Index (Song Thrush, Spotted Flycatcher and Bullfinch) showed greater increases in abundance. Stoate (2002) showed that for the passerine community as a whole, BAP species and other nationally declining species increased more than nationally stable or increasing species over the same period.

For the Farmland Bird Index species though, these results confirm the importance of assessing each species trend separately, supporting the recommendation of Freeman *et al*. (2001). More targeted conservation management may be necessary to meet the requirements of some BAP species in those areas where they occur. Some caution needs to be exercised however. The FBI is an indicator of sustainability, and targeting individual species in order to increase the index could be regarded as unreasonable manipulation of this sustainability index. Every effort should be made to ensure that conservation management is consistent with wider rural development objectives for integrated and sustainable management and use of natural resources on farmland.

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