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Post-war changes in arable farming and biodiversity in Great Britain

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Summary

1. Agriculture represents the dominant land use throughout much of western Europe, and a significant part of European biodiversity is associated with this habitat. We attempted to quantify the changes in agriculture and biodiversity in Britain since the 1940s.
2. There have been widespread declines in the populations of many groups of organisms associated with farmland in Britain and north-west Europe. The declines have been particularly marked amongst habitat specialists; many of the taxa still common on farmland are habitat generalists.
3. Farming practices have become increasingly intensive in the post-war period, with a dramatic reduction in landscape diversity. Since 1945, there has been a 65% decline in the number of farms, a 77% decline in farm labour and an almost fourfold increase in yield. Farms have become more specialized; the greatly increased use of machinery has made operations quicker and more efficient, but has resulted in the removal of 50% of the hedgerow stock. Autumn sowing of crops has become predominant, with winter stubbles now far less prevalent. The number and extent of chemical applications has increased greatly, but the net amount applied, and their persistence, has decreased in recent years.
4. Intensification has had a wide range of impacts on biodiversity, but data for many taxa are too scarce to permit a detailed assessment of the factors involved. Reduction in habitat diversity was important in the 1950s and 1960s; reduction in habitat quality is probably more important now.
5. As a case study, the declines in populations of seed-eating birds populations were assessed in relation to changing agricultural management. Generally, the declines were likely to be caused by a reduced food supply in the non-breeding season, although other factors may be important for particular species.
6. Agriculture will face a number of challenges in the medium term. While research into the mechanisms underlying species and habitat associations, and their interaction with scale, will be critical in under-pinning management, consideration of farmer attitudes and socio-economic factors is likely to be as important. Biodiversity may benefit from integrated farming techniques but these need to incorporate environmental objectives explicitly, rather than as a fringe benefit.

Key-words: agricultural intensification, landscape diversity, pesticide use, population declines, seed-eating birds, soil seed banks

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Introduction

Throughout Europe, farmland represents the major land use. In Britain 18·3 million ha (75% of the land area) was agricultural land in 2000, with arable crops and grass under 5 years old representing about 35% [Department of Environment, Food and Rural Affairs (DEFRA) statistics, available at http://www.defra.gov.uk/esg]. In the past, conservation effort has often focused on semi-natural habitat or species that have localized distributions; however, the total conservation interest of farmland may exceed that of

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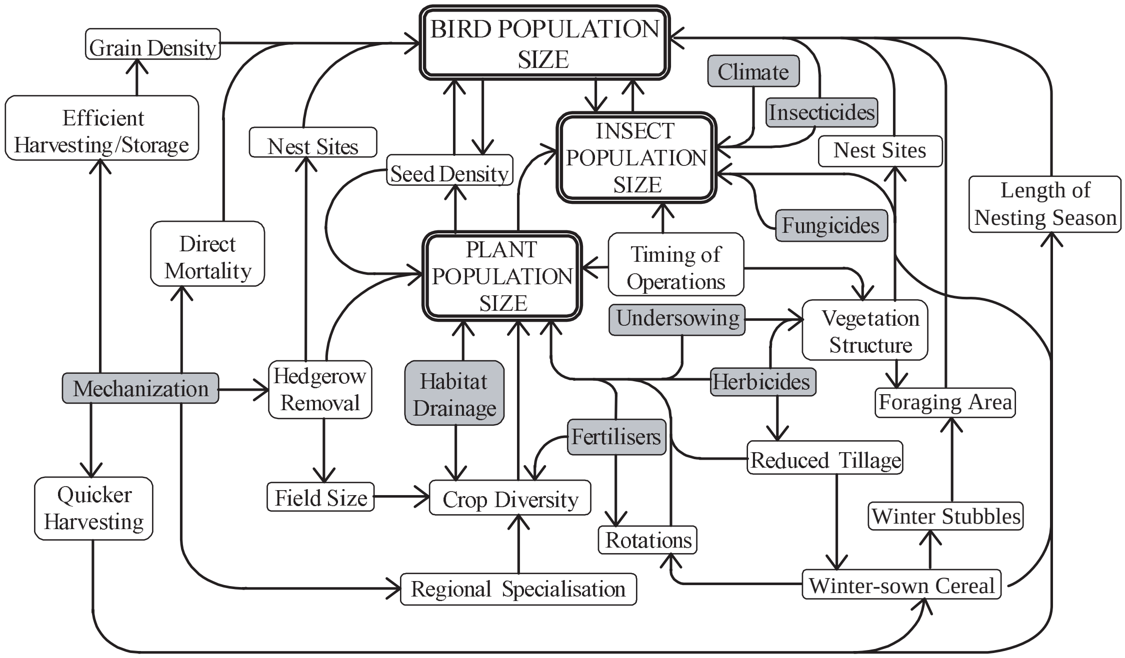

Fig. 1. Potential causes of population change in plant, insect and bird populations resulting from changes in arable management. The major drivers are highlighted by shading. Note this figure is not intended to be comprehensive.

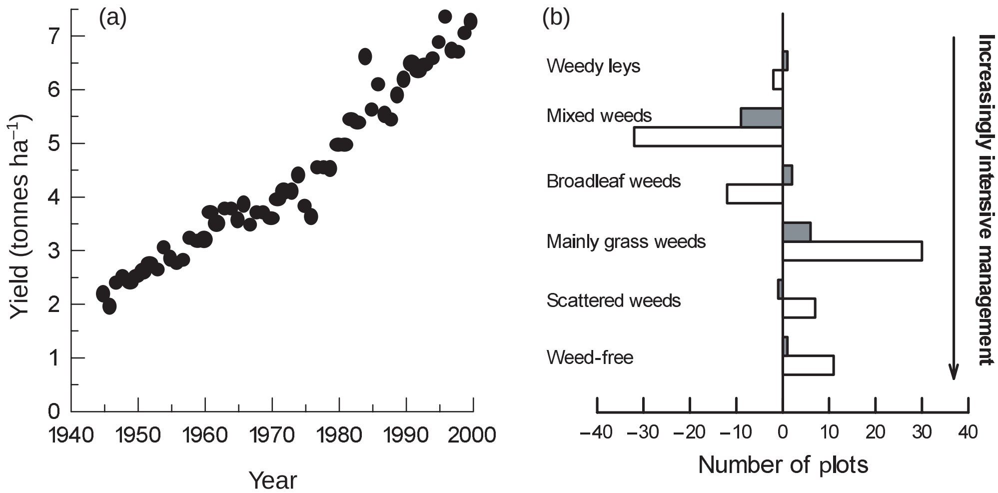

Fig. 2. Increasing efficiency in arable farming. (a) Average cereal yield in Britain (DEFRA statistics). (b) Change in vegetation communities on Countryside Survey arable plots between 1978 and 1990 in arable (open bars) and pastoral (filled bars) landscapes. Bars indicate net number of plots gained or lost from each community type. From data in Barr *et al*. (1993).

nature reserves (Krebs *et al*. 1999). Farm management has undergone major changes since the 1940s, partly in response to government policy objectives and partly through technological advances, creating a complex array of factors that might influence biodiversity (Fig. 1).

In Britain, modern agriculture is largely a consequence of the 1947 Agriculture Act, which sought to attain self-sufficiency in food production, a trend which accelerated with the country's accession to the European Union (EU) in 1973 (Potter 1997). As a result, the average cereal yield in Britain has increased continuously since the 1940s (Fig. 2a), even though the workforce has declined by 77% in the same period (DEFRA statistics). This increase in efficiency has resulted in fields becoming increasingly devoid of biodiversity, with few non-crop plants being tolerated (Fig. 2b). Consequently, there has been much concern about the status of a wide range of species on farmland (Donald 1998; Krebs *et al*. 1999). We attempted to quantify changes in population status of the flora and fauna occurring on lowland farmland and the management that has impacted on them. We focused primarily on currently widespread taxa in Britain, which represent a major proportion of the sum of Britain's wildlife. However, many of the same conditions are likely to apply elsewhere in Europe (Pain & Pienkowski 1997) and, to a certain extent, North America (Askins 2000).

Table 1. Population changes in the wildlife of arable farmland in Great Britain. The percentage of species studied (*n*) increasing or decreasing is given; where these were identified as 'significant', the percentage showing no change is also given. Only studies dealing with national populations are included (local/regional studies are discussed in the text); studies listed in bold measured populations on farmland, those in plain type national populations of species occurring on farmland

	Years	Measure	*n*	% decreasing	No change	% increasing	Reference
Lower plants	Pre/post-1950	Range	38	79	21	0	Hill, Preston & Smith (1994)
Plants*	1960–87	Range	46	67	–	33	Rich & Woodruff (1996)
Plants	Current	Opinion	46	41	37	22	Grime, Hodgson & Hunt (1988)
Plants	1960–2000	Literature review†	22	56	–	45	Sotherton & Self (2000)
Plants‡	**1978–90**	**Frequency**	**45**	**33**	**59**	**9**	**Smart *et al.* (2000)**
Plants§	**Long-term**	**Number of studies**	**45**	**60**	**38**	**2**	**Wilson *et al.* (1999)**
Dragonflies	Pre-1975–1988	Range	32	56	16	28	DoE (1998)
Grasshoppers	Pre-1975–1988	Range	22	50	9	41	DoE (1998)
Butterflies	Pre-1970–1982	Range	54	56	7	37	DoE (1998)
Butterflies	1982–95	Range	25	4	42	54	Asher *et al.* (2001)
Butterflies	1976–2000	Index	21	14	43	43	Greatorex-Davies & Roy (2001)
Moths	1968–84	Index	45	16	56	28	Woiwod & Dancy (1986)
Amphibians	1970–80	Opinion	5	40	60	0	Cooke & Scorgie (1983)
Reptiles	1970–80	Opinion	4	75	25	0	Cooke & Scorgie (1983)
Birds	1940–69	Opinion	28	54	7	39	Gibbons, Avery & Brown (1996)
Birds	1970–90	Range	28	86	–	14	Gibbons, Reid & Chapman (1993)
Birds	**1970–99**	**Index**	**18**	**78**	**5**	**17**	**Updated from Fuller *et al.* (1995)**
Mammals	1965–95	Various	12	50	33	17	Harris *et al.* (1995)

*Only significant changes reported.
†Based on data from limited study areas.
‡Data from lowland arable and pastoral areas.
§Includes data from Europe.

Trends in populations

Despite the acknowledged importance of long-term monitoring studies (Woiwod 1991), there are relatively few such projects in Britain and fewer elsewhere. In Britain, national schemes monitor birds (Baillie *et al.* 2001), butterflies (Pollard, Moss & Yates 1995), moths and aphids (Woiwod & Harrington 1994). On a local scale, the Game Conservancy Trust (GCT) has monitored a wide range of plants and insects on 62 km^2 of Sussex downland in southern England since 1970 (Aebischer 1991). The declines in farmland taxa have been widespread, with around half of plants, a third of insects and four-fifths of bird species experiencing a population decline (Table 1). Consequently, farmland probably holds more scarce and threatened plant species than any other habitat (Rich & Woodruff 1996). Furthermore, 12 of 36 bird species listed as of highest conservation concern in Britain (Gregory *et al.* 2000) occur on farmland, all but one (corncrake *Crex crex* L.) because of a recent severe (> 50%) population decline.

PLANTS

Much flower-rich farmland habitat, such as hay meadows, has disappeared in recent decades (Wilson 1992); however, trends in individual species are less well documented. Many farmland plants contracted markedly in range through the 1950s and 1960s, for example species once regarded as arable weeds, such as corn marigold *Chrysanthemum segetum* L. and corn buttercup *Ranunculus arvensis* L., are now very localized (Stewart, Pearman & Preston 1994). More recently, Barr *et al.* (1993) noted that the number of species in their Countryside Survey arable plots declined by 38% between 1978 and 1990, with a 10% decline in pastoral areas; annual weeds showed by far the biggest decline. While there is a tendency for most species to decrease in frequency of occurrence, some show local increases and a few species, such as cleavers *Galium aparine* L. and blackgrass *Alopecurus myosuroides* Huds., are becoming increasingly dominant (Smart *et al.* 2000; Sutcliffe & Kay 2000). Declining population trends of plants in arable habitats have been reported elsewhere in Europe, for example Denmark, Finland, the Netherlands and Italy (Andreasen, Stryhn & Striebig 1996; Campbell *et al.* 1997), but not in Hungary (Toth, Benecsne & Balazs 1997). The GCT study noted little overall change in plant numbers over 30 years, with some species increasing (notably goosefoots *Chenopodium* sp. and poppies *Papaver* sp.) and others declining [for example chickweed *Stellaria media* (L.) Vill.] (Aebischer 1991; Ewald & Aebischer 1999).

Although interpretation is complicated by improvements in seed-cleaning technology and sampling problems, analysis of the impurities (i.e. weed seeds) in seed cereals submitted for statutory testing since the 1930s (Don 1997) shows most species are occurring less frequently, indicating lower abundance; a few species, such as knotgrass *Polygonum aviculare* L., that are particularly difficult to control remain relatively common. There is also evidence that the overall size of the seed

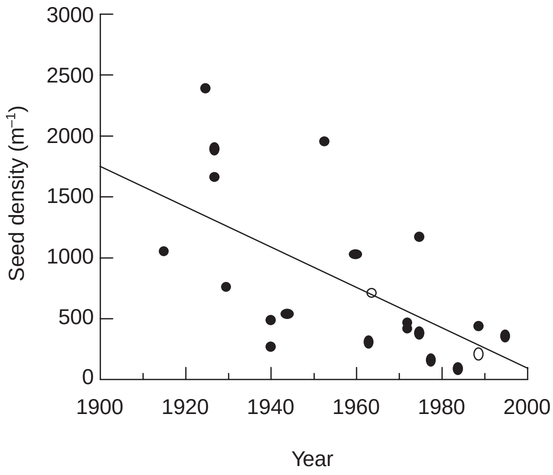

Fig. 3. Published estimates of seed density in arable soils. Points represent densities of dicotyledonous seed in the top 1 cm of soil in arable fields in Britain (filled symbols; from sources listed in Robinson 1997) and Denmark (open symbols; Jensen & Kjellsson 1995). Studies are included only if they sampled the entire seed bank between September and November and the fields had been part of a cereal-based rotation for at least 5 years; results from adjacent fields and years have been averaged. Slope of regression through British data: –17 seeds m^{-2} $year^{-1}$, $R^2 = 0{\cdot}35$.

bank in arable soils has declined markedly in Britain (Fig. 3). A similar trend has been reported in Denmark (Jensen & Kjellsson 1995), where the density of viable seeds in arable fields halved between 1964 and 1989; particularly large declines (> 90%) were recorded for chickweed and corn spurrey *Spergula arvensis* L.

INVERTEBRATES

National monitoring of Lepidoptera suggests that those species that occur widely on farmland generally appear not to be declining, whereas those with restricted distributions, often typical of unimproved grassland, are declining (Table 1; Pollard, Moss & Yates 1995; Asher *et al*. 2001). However, this may reflect the nature of the monitoring programmes, as relatively few stations are located on farmland (Pollard, Moss & Yates 1995). A study comparing trends in moth numbers in different habitats at Rothamsted (Hertfordshire, England) shows a general decline in farmland populations, but little change in woodland populations (Woiwod & Harrington 1994). Cowley *et al*. (1999), however, found that both common and localized butterfly species in pastoral north Wales declined in frequency between 1901 and 1997. Declines in butterfly populations have been reported elsewhere in Europe, although they seem less severe in southern Europe, which generally has a less intensively managed landscape (Van Swaay 1990; Pavlicek-van Beek, Ovaa & van der Made 1992).

Data from the Rothamsted insect survey suggest aphid populations have shown little marked change since the 1960s, although a few species appear to have increased (Woiwod 1991). These samples are drawn from an air column using suction traps, and are thus likely to draw individuals from a range of habitats. The GCT study in Sussex, in contrast, has recorded a dramatic decline in aphid numbers since 1970. These samples were taken from within cereal fields, where the effects of arable management are most apparent. A number of field studies across Europe have shown declines in many carabid species, with a few species becoming more common and dominating the assemblage (Luff & Woiwod 1995; Kromp 1999). There have also been declines in many bumblebee *Bombus* species throughout Europe; in Britain, particularly in eastern and central England (Williams 1986; Corbet, Williams & Osborne 1991).

The most detailed study of long-term trends in invertebrate abundance in Britain comes from the work of the GCT in Sussex (Aebischer 1991; Ewald & Aebischer 1999). This study shows that most groups have declined in numbers, with some showing little change, notably Collembola (the most common group) but also carabids and some other predatory insects (Aebischer 1991; Sotherton & Self 2000).

AMPHIBIANS AND REPTILES

Although there is no systematic monitoring of amphibian and reptile populations, a survey of local fieldworkers suggested large declines in all the common British species in the 1970s, particularly in arable-dominated eastern Britain (Cooke & Scorgie 1983). In the 1980s, national declines amongst amphibians, but not reptiles, slowed and in some areas populations started to increase, particularly in south-east England, possibly because of an increase in the number of garden ponds (Beebee 1996). While the herpetofauna will have suffered as part of the rationalization of farm structure (for example from the loss of small ponds on farmland), arable habitat is unlikely to be used extensively, so these are not considered further.

MAMMALS

There is no general monitoring of mammal populations, and for many species even estimates of population size are scarce because of the difficulties in censusing them directly (Harris *et al*. 1995). Gamekeepers' records have been collated since at least 1960 and suggest that populations, or at least gamekeeper 'bags', of hedgehogs *Erinaceus europaeus* L. and weasels *Mustela nivalis* L. have decreased, while stoats *M. erminea* L., rabbits *Oryctolagus cuniculus* L. and foxes *Vulpes vulpes* L. have increased (Tapper 1992). However, these trends are confounded by changes in effort (McDonald & Harris 1999). Brown hare *Lepus europaeus* Pallas populations declined between the 1960s and 1980s, both in Britain and elsewhere in Europe, but may have remained relatively stable since (Hutchings & Harris 1996). Bat populations are particularly difficult to census, but roost counts suggest populations are generally decreasing (Stebbings 1988; Yalden 1999).

Small mammal populations fluctuate widely, however; both harvest mouse *Micromys minutus* Pallas and field vole *Microtus agrestis* L. are thought to have declined over the last 40 years, whereas wood mouse *Apodemus sylvaticus* L. populations, the most common small mammal of cereal fields, have apparently shown little change (Harris *et al*. 1995). In a review of the diet of the barn owl *Tyto alba* Scopoli, which forages largely on farmland, Love *et al*. (2000) noted a marked increase in the proportion of wood mice and pygmy shrews *Sorex minutus* L. but a decrease in common shrews *S. araneus* L. between the 1970s and 1990s, although to what extent this reflects changes in prey density is uncertain.

BIRDS

In the immediate post-war period, many populations of farmland birds were recovering from low-points following declines during the 1920s and 1930s, with peak populations reached in the early 1970s (Gibbons, Avery & Brown 1996). Currently, about half of the species occurring on farmland (not just farmland specialists) have declined to some extent since 1968 (29% significantly; Siriwardena *et al*. 1998). The mean decline in population size for 17 farmland specialists has been 26%; birds that forage on seeds in winter seem particularly affected ($n = 8$, mean decline 39%; other nine species, mean decline 4%). In general, although population declines have been greatest in arable areas of eastern Britain, local extinction (and hence range contraction) has been greatest in the largely pastoral west because of lower initial populations there (Chamberlain & Fuller 2001).

Elsewhere in Europe, birds of farmland ecosystems have declined consistently in the last 25 years, with the declines being greatest in the more intensively farmed areas of north-western Europe and least in eastern Europe, where the largest populations remain (Donald, Green & Heath 2001). In North America, many species characteristic of farmland or grassland habitats have also declined over the last three decades (Peterjohn & Sauer 1999).

Recent changes in agricultural practices

Before discussing the ways in which agricultural change has impacted on wildlife, we outline the nature and extent of the major changes that have occurred. We focus on arable farming, although changes in lowland pastoral and upland agriculture are likely to be at least as great (Vickery *et al*. 2001). The policies underlying these changes are discussed by Robson (1997) and their wider impacts by Harvey (1997) and Pretty (1998).

Following removal of government support in the mid-19th century, which allowed cheap imports of grain from North America, much arable land fell into disuse or was converted to pasture as dairy and sheep farming were more profitable (O'Connor & Shrubb 1986; Stoate 1995). Because of this reduced capacity and the unreliability of imports, there was a shortage of food, especially cereals, during the Second World War. A consequent desire for self-sufficiency and demand for an increased standard of living after the austerity of the war has driven British post-war agricultural policy, initially through the Agriculture Act of 1947. This introduced widespread price maintenance for crops and capital grants and subsidies to encourage investment in agriculture, with a resultant increase in the area of arable land (Fig. 4). Many of the aims of the Act are also encompassed in the Common Agriculture Policy (CAP; originally established in 1962), with the result that trends in agricultural practice in Britain are reflected throughout much of western Europe (Potter 1997; Schifferli 2000).

Measured in the terms of the 1947 Act, post-war agriculture has been almost too successful, resulting in overproduction of many commodities, by as much as 20% to 30% annually in the 1980s. Because of these surpluses, and its high cost (currently around half the EU core budget), the CAP has increasingly needed reform. In 1992, the MacSharry reforms introduced measures to curb overproduction and direct subsidies (Robson 1997; Potter & Lobley 1998). More recently, reforms have incorporated environmental and rural development issues and prepare for the future expansion of the EU (Potter & Lobley 1998; Bignal 1999).

MECHANIZATION AND HEDGEROW LOSS

The Agriculture Act introduced financial incentives that encouraged an increase in the amount of machinery used on farms (Fig. 5). Farm machinery is capital intensive and its increasing use has accelerated the tendency towards large farms; the number of farms declined by 35% between 1949 and 1999 and increased in size, in 1949 only 1% of farms covered 500 acres (200 ha) or more, by 1999 this had risen to over 6% (DEFRA statistics). Farms also increasingly tend to specialize in either arable crops or livestock, resulting in a geographical separation between tillage and pastoral systems (Fig. 6).

Cultivation with chisel or tine ploughs is now favoured over mouldboard ploughs, resulting in less soil disturbance. Direct drilling (i.e. without removing the previous crop stubble) was popular in the 1970s and 1980s, but problems with weeds, such as blackgrass, may have limited its appeal. Harvesting is now accomplished more quickly and with less wastage, as several operations (harvesting, binding and threshing) occur in one stage. In 1970, losses from cereal harvesters averaged 2–3%, mostly from the front cutter bar (NIAE 1972). With an average wheat yield of around 4 tonnes ha^{-1} this would result in about 250 grains m^{-2} being left on the field. Currently, acceptable losses are

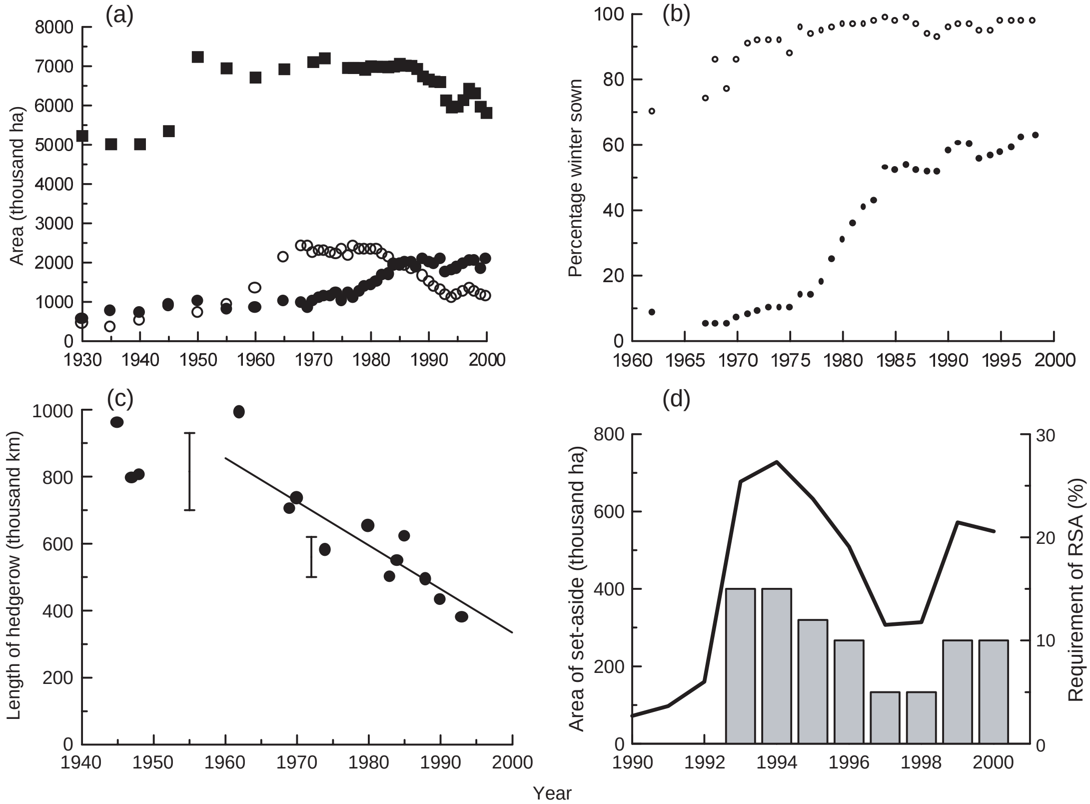

Fig. 4. Changes in farmland landscape structure. (a) Area covered by all cereals (filled squares), wheat (open circles) and barley (filled circles) in the UK. (b) Proportion of wheat (open circles) and barley (filled circles) sown in winter in the UK. (c) Published estimates of hedgerow length in England and Wales; dots indicate (mean) estimates, bars ranges. Slope of regression (1960–98): –13 000 km year^{-1}, R^2 = 0·74. (d) Extent of rotational set-aside (RSA) in the UK (line) and proportion of arable land required to be entered as RSA to qualify for arable subsidies (bars). Data from DEFRA statistics, except (c) from sources in Robinson (1997).

typically around 0·25% (Pearce 1990); even with a high yield of 10 tonnes ha^{-1}, this results in only around 60 m^{-2} being left on the field. Robinson (1997) recorded densities of only 20 m^{-2} in Norfolk stubbles in November 1995. Only in years of bad weather is a significant amount of grain likely to be spilt (Costigan & Biscoe 1991). Modern harvesters and the use of drying barns have also removed the need to stack grain in fields, once a widespread practice (Shrubb 1997); accidental spillage is also less common.

Field size has increased through removal of hedgerows, many (though by no means all) created by the 18th century Enclosure Acts (Pollard, Hooper & Moore 1974). Although some were advocating widespread hedgerow removal in the 1800s to allow for more efficient use of (horse-drawn) ploughs, it was not until the 1960s and widespread tractor/combine use that hedgerow removal began on a large scale (Fig. 4). In pastoral Somerset and Dorset, average field size increased from 5·5 ha to 9·5 ha between 1945 and 1994, whereas in arable Cambridgeshire, where stock-proof boundaries are generally no longer required, the increase has been from 6·5 ha to 16 ha; hedgerow density in arable counties may now only be 20–30% of that in pastoral counties (Barr *et al.* 1993; Westmacott & Worthington 1997). Many farmers manage their hedgerows to keep them 'tidy' (Macdonald & Johnson 2000), and through the 1980s hedgerows suffered from very intensive management, with most being heavily flailed or short-trimmed, although this may be becoming less prevalent (Joyce, Williams & Woods 1988; Westmacott & Worthington 1997) and rates of hedgerow removal and planting may now balance, at least in terms of overall length (Haines-Young *et al.* 2000).

CHEMICAL USAGE

Before 1930, pesticides were mostly preparations of lime, copper or sulphur. The first hormonal herbicide, methoxone (MCPA), was introduced in the 1940s, and by 1955 37 different compounds had been approved for agricultural use. In 1970, this had increased to 136, and by 1997 344 pesticide compounds were available (DEFRA statistics); the range of species that can be controlled has similarly increased (Ewald & Aebischer 1999). Modern pesticides are generally less persistent and more efficient (with less direct toxicity to non-target organisms), requiring smaller amounts of active ingredient (Campbell *et al.* 1997; Skinner *et al.* 1997). Although the weight of active chemical used has declined, the number and extent of applications (a better measure of environmental impact) has increased (Fig. 5).

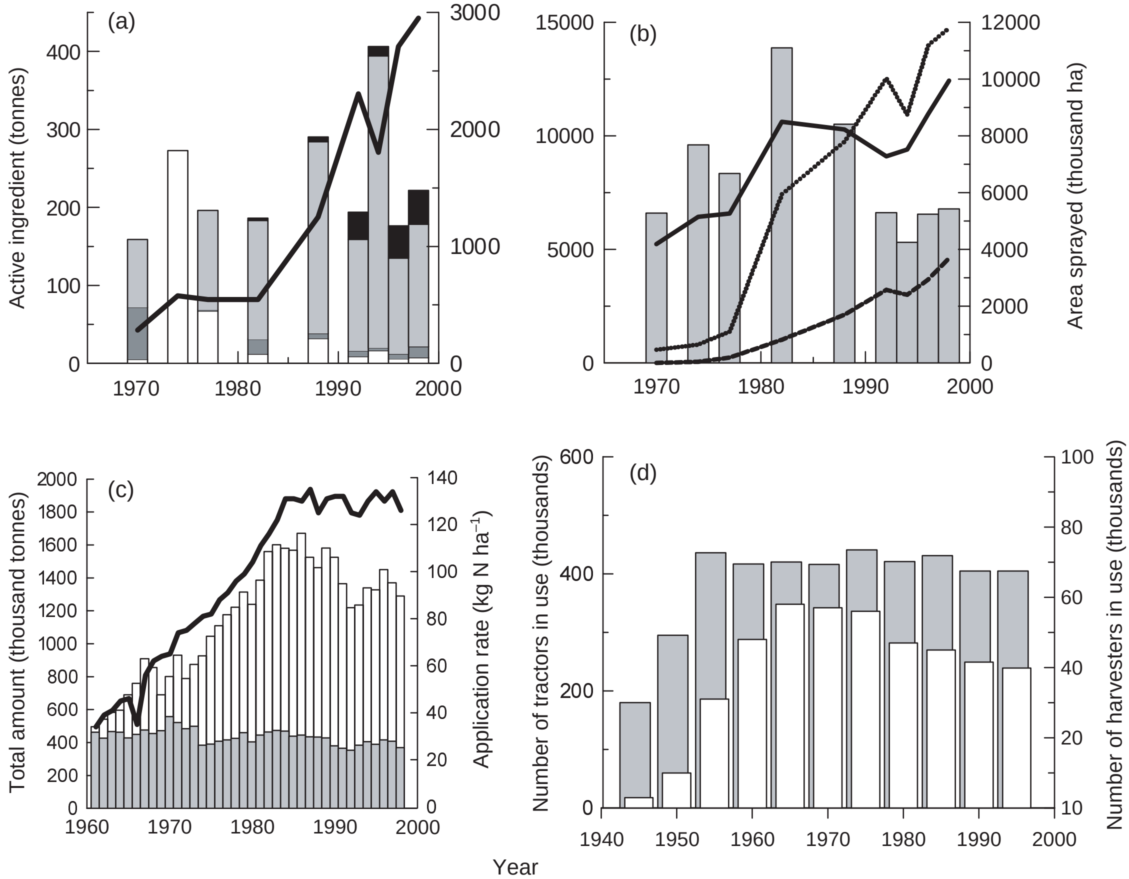

Fig. 5. The increasing inputs in farming. (a) Area sprayed with insecticide in England and Wales (line and right axis) and amount (active ingredient) of each type of insecticide used (bars and left axis): carbamates (white), organo-chlorines (heavy stippling), organo-phosphates (light stippling) and pyrethroids (black), this breakdown not available for 1974. (b) Area sprayed (right axis) with herbicides in England and Wales (solid line), fungicides (dotted line) and sown with chemically treated seeds (dashed line). Bars (and left axis) indicate weight of herbicide (active ingredient) applied. (c) Total amount of fertilizer applied in Britain (bars and left axis): nitrogen (open), phosphate (filled) and average application rates of nitrogen to all crops (line and right axis). (d) Numbers of tractors (filled bars and left axis) and combine harvesters (open bars and right axis) in use in the UK. All data from DEFRA statistics.

Most fields receive multiple treatments, although this varies between crops, winter cereals are usually sprayed more than spring cereals, potentially with up to a dozen different chemicals (Thomas, Garthwaite & Banham 1996). Usage of rodenticides has declined from 90% of arable farms in 1988 to 78% of farms in 1996, with usage becoming increasingly concentrated around buildings and during the autumn and winter (De'Ath, Garthwaite & Thomas 1998).

Some form of nutrient input is essential in farming, in order to replace nutrients removed from the system when the crop is harvested. Organic manures have long been used to maintain fertility but, since the 1940s, inorganic fertilizers have increasingly been used (Fig. 5). This allows greater concentrations of nutrients to be applied, with a quicker release time into the soil, but reduces the amount of organic matter in the soil and may also affect soil chemistry (Chalmers, Kershaw & Leech 1990). The increase in nitrogen use partly reflects the increase in planting of winter cereals (application rates are typically 50–80% higher than on spring cereals) and the production of silage rather than hay. The application of sulphur is also increasing as atmospheric concentrations decline through reduction in air pollution (Withers 1993).

CROP ROTATIONS

In some ways, modern farming in England began with the advent of the 'Norfolk' four-course rotation in the 19th century. This added a root crop to rotations involving two cereals and a grass ley (which was often undersown into the previous cereal) to improve weed control and fertility and allow crops and livestock to be raised on the same farm; with regional adaptations it continued as the basis of much farming until the 1940s (Stoate 1995). Although winter cropping was possible in southern England, usually as a catch crop of vetch *Vicia sativa* L. or fodder rye *Secale cerale* L. the post-war period has seen a great increase in the level of autumn sowing of crops, particularly barley *Hordeum vulgare* L. (Fig. 4; Stoate 1995, 1996). There is substantial regional variation in this though, most of the spring-sown barley, for example, is still sown in Scotland.

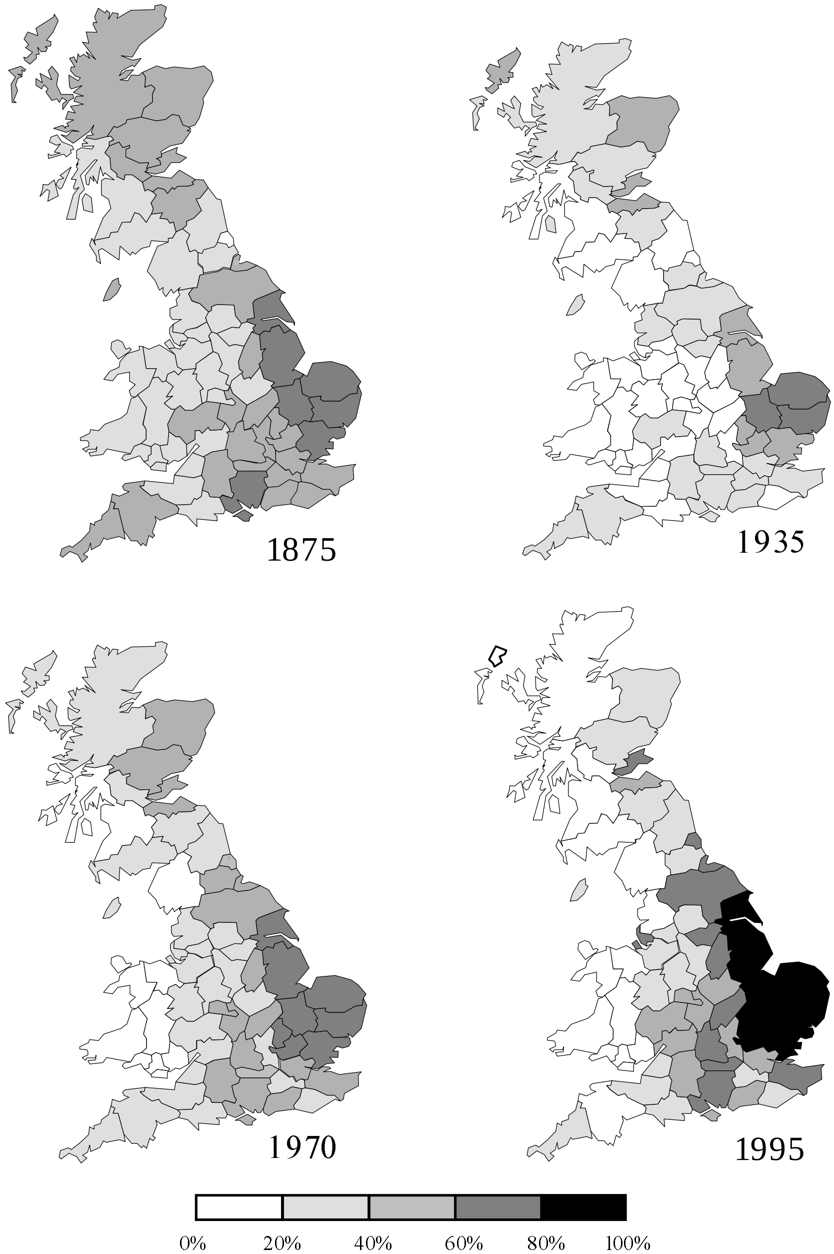

Fig. 6. The amount of annually tilled land, as a percentage of total farmed area by county in 1875, 1935, 1970 and 1995. Based on DEFRA statistics.

Before the widespread use of pesticides, weed control was achieved through tillage and the use of rotations that included non-cereal crops to prevent pest build-up and legume crops to maintain soil fertility, which were sometimes (25–30% of fields) undersown in the previous cereal crop (O'Connor & Shrubb 1986; Stoate 1995). The widespread use of chemicals, particularly pre-emergent herbicides, which allow grass weeds to be sprayed off before the crop is sown, has removed this dependency, allowing continuous cereal cropping. This encouraged the proliferation of arable cropping in eastern Britain, where climatic conditions are most suitable (Fig. 6).

With continuous cropping of largely autumn-sown cereals, undersown grass leys are much less frequent and fewer fields are left fallow over the winter. These stubbles can allow non-crop plants present to set seed and germinate. The introduction of set-aside (a production control measure) as part of CAP reforms has temporarily increased the amount of stubble, primarily in south-eastern England. Initially this measure was voluntary, but in 1992 arable subsidies were linked to farmers entering land as set-aside, which temporarily removes land from production for a period of up to 5 years (Robson 1997; Firbank 1998), greatly increasing its extent (Fig. 4). In Britain, around two-thirds of rotational set-aside (i.e. for one season) is left to naturally regenerate a green cover, effectively an overwinter stubble (Thomas, Garthwaite & Banham 1996); non-rotational set-aside forms a much more pasture-like habitat.

Although most stubble fields are ploughed in during the autumn, a few remain longer, although the frequency with which this occurs has decreased. Elliott, Cox & Simonds (1968) noted that 10·2% of cereal fields in East Anglia remained as stubbles in December, whereas a 1997 survey found that only 6·2% of the cereal area was stubble in December, virtually all of which could be accounted for as set-aside (Gillings & Fuller 2001). Similarly, Inglis *et al.* (1990) recorded stubbles on about 10% of their Cambridgeshire study area in December in the early 1970s; by the late 1980s, none were present at this time.

LANDSCAPE DIVERSITY

A major consequence of the intensification of agriculture has been a dramatic reduction in landscape diversity. At the turn of the century, much of British agriculture was of a mixed character, with both stock and crop husbandry occurring on the same farm. In the agriculture boom of the 1950s, a number of factors led to the polarization of agriculture in Britain, with arable farming predominating in the east and pastoral farming in the west (Fig. 6). Increasing use of machines meant animal labour, and hence fodder cropping (root crops and oats), was no longer required; the advent of chemical pest control and increased availability of inorganic fertilizers reduced the need for crop rotations so land could be devoted to a cash crop all year, rather than having to lie fallow for part of the season. Individual farms became larger and are continuing to grow, particularly in the arable east of the country, with the result that larger blocks of land are being managed with the same aim. Even in mixed farming areas, adjacent farms are likely to specialize in either crop or animal husbandry. However, the introduction of new crops, such as oil-seed rape *Brassica rapa* L. (in the late 1960s, covering 332 000 ha in 2000) and linseed *Linum usitatissimum* L. (in the early 1990s, covering 72 000 ha in 2000) will have contributed to habitat diversity.

Within farms, habitat diversity has been reduced, for example in the loss of non-cropped areas, hedgerows (Fig. 4), field margins, ponds and copses, although no national data on the extent of this exist. Surveys by Macdonald & Johnson (2000) in 1981 and 1998 show that management of these habitats partly reflects economic motivations, but mainly farmer attitudes; they also suggest loss of such habitats is becoming less prevalent. In the last 10 years, the number of new ponds and farm woods being created has outweighed those lost, increasing habitat diversity, although not necessarily restoring it (Haines-Young *et al.* 2000). In addition, with increasingly tighter legislative controls, farms are becoming increasingly hygienic, with less tolerance of waste products, such as dung or spilt grain.

Causes of population declines

Although the changes outlined above have been continuous since the 1940s, the period of greatest change was in the 1970s and early 1980s, when the widespread declines in farmland wildlife first received attention (Potts & Vickerman 1974; Chamberlain *et al.* 2000). Although there are many local studies of the biodiversity impacts of agriculture, discussion of the factors affecting the declines of specific taxa, however, is often hampered by the lack of data at a national (or even regional) scale. Local studies, usually at a plot or field scale, are usually difficult to interpret as their results may not be applicable at broader scales (Duffield & Aebischer 1994; Siriwardena *et al.* 2000a). Changes in agricultural practice have largely been monotonic and in the same direction, towards increasing yields. Distinguishing between the various factors and attributing causal relationships to a particular combination of factors is very difficult; simple temporal correlations are likely to be misleading (Chamberlain *et al.* 2000).

One reason often proposed for the wide range of declines in farmland taxa is the increased use of pesticides (Campbell *et al.* 1997). Although pesticides are undoubtedly effective at controlling their target species and often affect non-target organisms (Wilson *et al.* 1999), evidence of population changes from studies at scales larger than individual plots can be equivocal. Often this is done by comparing farms employing conventional or intensive management techniques with those employing lower intensity or organic practices. Farms with lower chemical inputs generally hold greater numbers, and higher diversity of species, of a broad range of taxa, although the reported statistical significance of these results can be weak and some find no differences (Braae, Nøhr & Petersen 1988; Holland *et al.* 1994; Moreby *et al.* 1994; BTO/IACR 1995; Campbell *et al.* 1997; Greenwood 2000). It is often difficult to attribute the greater numbers or diversity of species on organic farms solely to chemical use, because although farms are usually paired, the effects of other factors, such as hedgerow density and rotations, can be at least as important as any differences in chemical applications (Chamberlain, Wilson & Fuller 1999). For example, Yeates *et al.* (1997) recorded increased numbers of mites and nematodes and lower earthworm numbers on organic farms; however, this is likely to reflect differences in soil structure caused by differing tillage practices, rather than pesticide use.

PLANTS

A number of potential factors for the declines in arable plants can be identified: habitat drainage; changes in soil nutrient status; changes in the timing of cereal sowing; and the application of chemicals, either directly or indirectly (Hampicke 1978; Chancellor, Fryer & Cussans 1984; Wilson 1992). Detailed autecological work has elucidated the declines of some of the rarer species, for example, the corncockle *Agrostemma githago* L. declined due to improved seed-cleaning techniques, which are also likely to have reduced weed populations generally (Firbank 1988; Wilson 1992). The increase in autumn cereal sowing (Fig. 4b) has contributed to the declines of spring-germinating plants, such as the corn marigold, and earlier harvesting of winter crops will also have affected late autumn-germinating species, such as corn buttercup and shepherd's-needle *Scandix pecten-veneris* L. (Wilson 1992; Hald 1999).

Many species that remain common on farmland are either resistant to, or difficult to target with, herbicides, or have prolific, persistent, seed banks. This tends to suggest herbicides may be responsible for the declines of the remaining species, as does the lack of a decline in frequency of weed occurrence between 1969 and 1987 reported by Whitehead & Wright (1989), who recorded plant densities before herbicide application. Long-term studies of individual fields show that timing of sowing is critical, as are past management practices (Chancellor 1985). Ewald & Aebischer (1999) also noted that timing of application was important, with spring herbicide spraying having the greatest impact on plant populations. Minimum tillage favours species that have an ephemeral seed bank, particularly grasses, limiting the uptake of reduced cultivation techniques (Bobbink 1991; Wilson *et al*. 1999).

In parallel with the increase in pesticide use, inorganic fertilizers are also applied much more frequently, particularly in grass leys. Although increased nutrient status is a major problem in grassland habitats (Marrs 1993), there seems to be little evidence that increased nitrogen use affects diversity or numbers of non-crop plants directly in current high-intensity arable systems (McCloskey *et al*. 1997; Kleijn *et al*. 2001). Spray drift into field margins reduces plant density and diversity there (Marshall 1988). Many plant species that are currently increasing are associated with elevated nutrient status, and fertilizer-intolerant species may have disappeared in the 1950s and 1960s (Wilson 1992; Smart *et al*. 2000). At least some of the effect of fertilizers comes from indirect effects, such as shading from species that can use the nutrients most efficiently (Bobbink 1991). Organic manures potentially have the same effects as inorganic fertilizers, but nutrients are usually much less concentrated and released more slowly, reducing their deleterious impacts (Simpson & Jefferson 1996). With decreased atmospheric sulphur dioxide levels, we might expect lichen numbers to be increasing, but there is some evidence that increased ammonia emissions from farmland may be slowing this process (Coppins, Hawksworth & Rose 2001).

INVERTEBRATES

In general, many insect populations, particularly butterflies and orthopterans, are expanding as the British climate is becoming milder (Menzel & Fabian 1999; Roy *et al*. 2001); however, changes in habitat management have had a large, mostly negative, impact (Thomas, Morris & Hambler 1994; Warren *et al*. 2001). For example, many butterfly populations have suffered from the loss of unimproved pasture (Rands & Sotherton 1986; Asher *et al*. 2001), sawfly numbers from a reduction in the number of longer-term grass leys (Barker, Brown & Reynolds 1999), and insect diversity generally is lower in more intensively managed fields (di Giulio, Edwards & Meister 2001). Local studies have shown that some species are likely to have been affected by the loss of non-cropped habitats such as field margins, which can act as a reservoir of individuals (Dennis, Thomas & Sotherton 1994; Lee, Menalled & Landis 2001), although Jones (1976) noted greater numbers and diversity of arthropods in winter wheat plots than adjacent fallow plots. Diversity did, however, decrease through the 8 years of this study with the simplified rotation.

Chemical control of insects can have long-term consequences, with effects evident several years after the initial application (Aebischer 1990), although some species can recover very rapidly (Brown, White & Everett 1988). A large number of studies has quantified the short-term effects of insecticides on populations of both target and non-target species (reviewed in Wilson *et al*. 1999). These have generally been at a small scale and yielded varying results: most show declines in insect numbers following insecticide applications, but others show little change, or even increases. At least some of these differences in population trends are likely to be due to varying susceptibility to insecticide control; the timing of spraying is important and particular life histories will determine which species are most susceptible (Burn 1989; Vickerman 1992; Marc, Canard & Ysnel 1999).

Pesticides will also have indirect effects on invertebrate populations. A number of experimental studies has shown that they are dependent on the abundance of their food plants, for example staphylinids (Aebischer 1991), dipteran (Mowat 1974) and butterfly larvae (Feber, Smith & Macdonald 1996) and isopods (Paoletti & Hassall 1999). The decline of the chrysomelid beetle *Gastrophysa polygoni* L. has been linked to reductions in its food plant knotgrass *Polygonum aviculare* (Sotherton 1982). Moreby & Southway (1999) also present results of experiments showing that increased herbicide use reduces insect numbers. Increased fertilizer use reduces numbers of Orthoptera due to increased shading from the crop (Van Wingerden, van Kreveld & Bongers 1992).

The soil fauna, particularly the microfauna, represents a major part of the biodiversity of agro-ecosystems, though their study presents formidable challenges (Paoletti 1999). In general, greater soil disturbance reduces their numbers. The amount of organic matter (i.e. food) is the most important determinant of earthworm abundance, and although light soil cultivation can be beneficial, deep ploughing can reduce numbers by up to 50% (Edwards & Bohlen 1996). Increased fertilizer use is generally beneficial to the soil fauna, although concentrated liquid forms (slurry) may not be (Paoletti 1999). Holland & Luff (2000) found different practices favoured different carabid species, although the presence of non-cropped habitats, minimum tillage and some fertilization seemed to offer the broadest benefits.

MAMMALS

In their review of the population status of British mammals, Harris *et al*. (1995) noted that the most common causes of population change are likely to include habitat change (believed to affect 48% of the 65 species covered), use of chemicals (including pesticides, affecting 38%, mostly bats and rodents) and deliberate killing (28%, primarily carnivores and species perceived as pests), although few explicit data were available.

While a number of factors have been identified as potential causes of the decline in brown hare numbers, a reduction in landscape diversity has been suggested as being most important, at least in arable areas; in pastoral areas, increases in stocking densities and the increasing use of grass for silage may be more important (Hutchings & Harris 1996). Rabbit populations are recovering from a catastrophic decline caused by the myxoma virus in the early 1950s, and this has been linked to the decline and subsequent recovery of the stoat (for whom rabbits are an important prey species) and the population increase and current decline in the weasel (rabbit grazing reduces the amount of long grass for voles, which are a weasel's primary prey) (Tapper 1992; Yalden 1999). Numbers of bats are likely to have suffered from destruction of roost sites, loss of foraging habitat and a general decline in their insect prey (Walsh & Harris 1996; Yalden 1999). For example, the greater horseshoe bat *Rhinolophus ferrumequinum* Schreber may have declined with the loss of hedgerows as feeding habitat (Mitchell-Jones 1998).

Population trends of small mammals on farmland are unclear, but are likely to be affected by food supply of both insects and seeds, which have declined (Tew, Macdonald & Rands 1992); they may also be particularly susceptible to direct mortality from rodenticides or eating seeds treated with molluscicides, although immigration from other habitats can occur rapidly (Shore *et al*. 1997; McDonald *et al*. 1998). The harvest mouse has been adversely affected by loss of grassland habitats and increased planting of winter cereals (Perrow & Jowitt 1995), and common vole *Microtus arvalis* Pallas numbers have declined in areas where their preferred grassland habitat has been converted to agriculture (Gorman & Reynolds 1993). There is, however, little quantitative evidence for the relative importance of changes in habitat structure, harvesting, chemical usage and food supply.

BIRDS

Changing agricultural practices have had a major impact on bird communities, and the reasons for the declines in some species can be pinpointed through population monitoring and autecological studies (Aebischer, Green & Evans 2000). For example, corncrake populations have fallen due to increasingly intensive grassland management, in particular a loss of hay meadows (Green & Stowe 1993), and the breeding populations of some waders, such as redshank *Tringa totanus* L. and lapwing *Vanellus vanellus* L., are declining with the loss of unimproved grassland and increased stocking densities (O'Brien & Smith 1992; Peach, Thompson & Coulson 1994). Agricultural changes have benefited some species, for example woodpigeon *Columba palumbus* L., because of increased oil-seed rape planting (Inglis *et al*. 1990). Declines in seed-eating passerines (larks, finches, buntings and sparrows) have received much attention recently and we review them in some detail, as a case study of the processes involved.

In general, British bird species are nesting earlier and with increased success because of changes in climatic variables (Crick & Sparks 1999), although amongst seed-eaters this may also be a density-dependent response to lower population levels (Siriwardena *et al*. 2000b). While productivity per nesting attempt has not declined (except in the case of the linnet *Carduelis cannabina* L.), there may have been a decline in the number of breeding attempts, particularly among open-field nesters. For example, skylarks *Alauda arvensis* L. generally prefer to nest in short vegetation, such as spring-sown cereal, particularly where food densities are high, but frequently abandon nests in winter cereal (the crop grows too fast), reducing opportunities for subsequent broods (Wilson *et al*. 1997; Wakeham-Dawson *et al*. 1998; Chamberlain *et al*. 1999). Corn buntings *Miliaria calandra* L. nest much later than other species and nest success may have decreased, because of both harvesting operations (Crick *et al*. 1994) and reduced insect abundance (Brickle *et al*. 2000). However, responses to cropping regime are likely to be complex; both corn buntings and skylarks nest at higher densities and with greater success in areas of high crop diversity (Ward & Aebischer 1994; Chamberlain & Gregory 1999).

Several studies have suggested the potential importance of hedgerow features for farmland birds, although there is often disparity in the preferences reported (Lack 1992; Parish, Lakhani & Sparks 1994, 1995; Macdonald & Johnson 1995). Characteristics such as verge width and adjacent land use may be more important, particularly in winter (Green, Osborne & Sears 1994; Parish, Lakhani & Sparks 1995). Comparisons of bird numbers between farms that have lost 60–70% of their hedgerows reveal little difference in breeding bird densities (Lack 1992; Gillings & Fuller 1998), although differences may be stronger at lower population densities (O'Connor 1986).

For most passerines that eat seeds in winter, mortality appears to be relatively low in the early winter period and much higher in late winter when food resources have been depleted, with late winter mortality relatively more important since 1970 (Crick, Donald & Greenwood 1991). Adult and juvenile survival rates of most species decreased during their population declines, and

Table 2. Population and dietary characteristics of British farmland seed-eating passerines. Population changes on farmland Common Birds Census plots between 1976 and 1995 from Siriwardena *et al.* (1998) (95% confidence interval in parentheses). Relative rates of survival (S), daily nest failure rates (NFR) and brood size (BS) are given for periods of increasing (I), stable (S) or declining (D) population trends, asterisks indicate where the change is sufficient to have caused the population change (Siriwardena, Baillie & Wilson 1998; Siriwardena *et al.* 2000b). Figures in bold indicate statistically significant results, and hyphens insufficient data. The use of gardens is given by: †recorded < 10% gardens; ‡recorded in 25% of gardens; §recorded in 80% of gardens during the late winter (Cannon 2000)

Species	Index	S	NFR	BS	Gardens?	Wintering area
Tree sparrow	–84 (–97, +0·02)	S > D$^{(-)}$	I > D	**D > I**	†	Mostly sedentary
Corn bunting	–66 (–, –)	–	I > D	**D > I**	†	Sedentary
Skylark	**–49 (–57, –39)**	–	S > D > I	**D > S > I**	†	Within Britain
Reed bunting	**–46 (–69, –0·08)**	I = S > D$^{(*)}$	**I > D = S**	**S > I > D**	†	Mostly sedentary
Linnet	**–38 (–55, –14)**	D > I	**D > I$^{(*)}$**	**I > D**	†	Mostly France/Spain
Yellowhammer	**–36 (–42, –10)**	I > S > D$^{(-)}$	**I > S > D**	**D > S > I**	†	Mostly sedentary
Greenfinch	–1 (–21, +34)	**S > D > I**	S > D = I	I > S > D	§	Britain (France, Belgium)
Goldfinch	+7 (–20, +52)	**I > D$^{(*)}$**	**D > I**	D > I	‡	Mostly France/Spain
Chaffinch	**+19 (+7, +31)**	**I > S**	I > S	I = S	§	Within Britain

for reed bunting *Emberiza schoeniclus* L., goldfinch *Carduelis carduelis* L. and house sparrow *Passer domesticus* L. these changes are sufficient to account for the declines (Siriwardena, Baillie & Wilson 1998, 1999; Peach, Siriwardena & Gregory 1999). In parallel with the declines in farmland birds, populations of predators, particularly sparrowhawks *Accipiter nisus* L. and magpies *Pica pica* L., have increased. This is very unlikely to have caused the population declines of farmland birds (Thomson *et al.* 1998), although predation may have increased because of habitat simplification (Donald & Vickery 2000), a subject that needs further study. Intriguingly, recent work has suggested that parasites transmitted from captive-reared pheasants *Phasianus colchicus* L. may be having an adverse impact on grey partridge *Perdix perdix* L. populations (Tompkins *et al.* 2000).

While the breeding season and winter cannot be considered in isolation, the decline of farmland passerine seed-eaters can be broadly related to their foraging on seeds in winter and changes in survival (Table 2), although the importance of changes in number of nesting attempts remains to be quantified. For example, most of the British populations of linnet and goldfinch winter in France and Iberia, where seed densities can be approximately five times those currently found in British soils (Díaz & Tellería 1994). Chaffinches *Fringilla coelebs* L. and greenfinches *Carduelis chloris* L. forage extensively in gardens, which are also becoming increasingly important for other species, particularly in the late winter period (Cannon 2000). Similarly, while set-aside has many potential benefits for wildlife (Firbank 1998), current management practices are likely to limit the extent to which weed seeds are present, which may explain why populations of farmland seed-eaters in Britain did not increase with the widespread introduction of set-aside in 1992.

The population decline of grey partridge in Britain has been linked to a decrease in the number of invertebrates available during the nesting season, and hence chick survival (Potts 1986), although the same may not be true in France (Bro *et al.* 2000). The linnet is the only bird where changes in breeding season have been shown to be sufficient to account for changes in population (Table 2). This suggests that declines in the summer insect fauna may not have affected granivorous passerines to the same extent (partridge chicks are nidifugous and forage independently). The potential importance of winter food for passerines is highlighted by the cirl bunting *Emberiza cirlus* L. (Evans 1997; Aebischer, Green & Evans 2000). The cirl bunting was once common over much of southern England, but by the 1980s had become restricted to the south-west with a population of just 118 pairs. When farmers were paid to leave weed-rich winter stubbles, the population increased to around 500 pairs in less than 10 years.

Bird populations elsewhere in north-west Europe are likely to be affected by agricultural intensification in a similar way; however, those in southern and eastern Europe are more often affected by loss of unimproved habitat (Schifferli 2000; Donald, Green & Heath 2001), and numbers of little bustard *Tetrax tetrax* L. have increased with low-intensity cultivation of steppe habitats (Wolff *et al.* 2001). In North America, many species are thought to be declining because of farmland abandonment, particularly in the east, farmland elsewhere being increasingly intensively managed (Askins 2000). The precipitous decline of the dickcissel *Spiza americana* Gmelin has been related to changes in food supply in their wintering grounds (Fretwell 1986).

Discussion

CURRENT PATTERNS

Frequently, agricultural land is regarded solely as a matrix between habitat islands of woodland, wetland or heath. Currently, such land supports many species of conservation concern, and this review demonstrates that widespread declines in wildlife populations have occurred because of agricultural intensification. Although humans have altered and influenced the

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environment throughout most of Europe's post-glacial history, the current declines are much greater and more widespread than any recorded previously (Campbell *et al*. 1997).

A major factor in the decline in farmland biodiversity has been the loss of more specialized taxa. Thus, many of the birds (Gibbons, Avery & Brown 1996), plants (Wilson 1992) and butterflies (Asher *et al*. 2001) that declined markedly in the period prior to 1970 were dependent on areas of extensive low-input cultivation or the presence of non-cropped habitat. In general, the plants currently common on arable land are found in a wide range of other habitats, particularly disturbed anthropogenic habitats. Similarly, butterflies now typical of farmland in Britain (Warren *et al*. 2001), carabid beetles in the Netherlands (Turin & den Boer 1988) and heteropteran bugs in Switzerland (di Giulio, Edwards & Meister 2001) are those that tend to be habitat generalists, as do the non-declining farmland birds in Britain (Siriwardena *et al*. 1998). Until recently, conservation management and policy has focused on (scarce) species with relatively specific conservation requirements in more or less well-defined areas, facilitating straightforward policy measures, such as designation of protected areas or promoting local changes in land management. Although such efforts have proven very successful in achieving conservation targets for farmland birds with localized distributions, such as the cirl bunting (Evans 1997), they have been less successful for widespread species (Gregory *et al*. 2000).

Although habitat loss remains an important driver of biodiversity loss, particularly for plants and invertebrates (but not birds; Gillings & Fuller 1998), more intensive field management, degradation in habitat quality, and increasing habitat homogeneity (across all scales) are currently more important. Increased use of pesticides is resulting in smaller species-depauperate plant and invertebrate communities (Wilson *et al*. 1999). This in turn has consequences for species higher up the food chain, with populations of at least some predatory insects, mammals and birds being adversely affected by reduced food resources. The shift from spring- to autumn-sowing of cereals has also had widespread consequences. These factors are likely to interact, for example populations are likely to be less resilient to chemical control (intentional or not) in areas of reduced habitat heterogeneity (Lee, Menalled & Landis 2001). Although this review has concentrated on arable systems, the biodiversity impacts of pastoral systems are at least as great (Vickery *et al*. 2001).

FUTURE TRENDS

In the medium term, agriculture throughout Europe will face a number of challenges. Changes in the CAP to reduce its cost and comply with world trade agreements are likely to be a major driver of agricultural change, particularly with the proposed expansion of the EU to include eastern European countries (Potter 1997; Bignal 1999). Current world trade negotiations are focusing on the removal of direct production and export subsidies (including 'Blue Box' payments, such as the EU's arable area payments and livestock premiums), which represent at least a half of 'total farm income' in Britain (DEFRA statistics). Reductions in production subsidies are likely to fuel further the trend towards increasingly large farm enterprises and profitability (yield ha^{-1}), which will probably exacerbate current trends of biodiversity loss. However, non-market distorting payments, such as those for crop research, conservation or environmental aims ('Green Box' payments) are currently excluded from subsidy reduction commitments. The use of the Rural Development Regulations and other environmental measures has the potential significantly to diversify farm income streams, and could replace the loss of production subsidies (Bignal 1998, 1999).

In general, farming practices in eastern Europe are less intensive and declines in biodiversity appear to be less widespread, at least amongst birds (Donald, Green & Heath 2001). In eastern Europe, farmland wildlife is threatened by the abandonment of agricultural land (9% between 1990 and 1999) and subsequent scrub encroachment following the collapse of communism and state support, although this benefits other forms of biodiversity. Abandonment is also a problem in other areas of low-intensity farming, particularly southern Europe (Osterman 1998). The future expansion of modern production systems as eastern European countries are assimilated into the EU is likely to result in widespread declines in farmland biodiversity, as has been witnessed in north-western Europe; Britain acceded to the EU in 1973, coinciding with the greatest changes in agriculture (Chamberlain *et al*. 2000). In Spain and Portugal, which joined the EU in 1986, production subsidies provided through the CAP have had a major detrimental effect on wildlife (Suárez, Naveso & de Juana 1997).

The development of genetically modified (GM) plants in 1982 has stimulated much debate and controversy, although environmental concerns have largely focused on crops resistant to herbicides or insect pests (Burke, Seidler & Smith 1994; Beringer 2000). At present, there is little direct evidence to suggest that such crops are highly invasive or that pollen transfer to non-crop plants will be rife (Crawley *et al*. 2001), and although tests have shown adverse impacts on non-target organisms (Donegan *et al*. 1999; Losey, Rayor & Carter 1999) there is some doubt as to their impact on field populations (Sears *et al*. 2001; Trewavas & Leaver 2001). The major impacts on biodiversity are likely to be indirect, for example by changing cropping patterns (continuous cropping will be encouraged) or changing agrochemical inputs, and dependent on interactions between agronomic and economic pressures (Hails 2000; Watkinson *et al*. 2000). Although fewer pesticide applications will be needed, control may be more complete and the impacts on biodiversity greater. However, the impacts may not all be negative, greater pest control in fields could reduce the need to spray non-cropped areas and remove the need for deep ploughing, benefiting soil insects (although soil organic matter would also decline). The opportunity to minimize biodiversity loss with the introduction of this and other novel practices, rather than trying to reverse them afterwards, should not be missed.

In the longer term, technological advances in agriculture will be set against a backdrop of a changing climate (Rosenzweig & Parry 1994). Most current climate models predict Britain will experience warmer, drier summers and warmer, wetter winters; climatic variability will also be greater, leading to more extreme weather events (Hulme & Jenkins 1998). Cereal yields are likely to decrease with increasing temperature, yields will also be susceptible to high temperatures at critical periods (Wheeler *et al.* 1996; 2000). This is likely to lead to a north- and westward shift in arable cultivation; grassland systems may spread south and east, and crops such as sunflowers *Helianthus annuus* L. and fodder maize *Zea mays* L. may become more widespread, dependent on the global economic situation (MAFF 1999; Parry *et al.* 1999). The possibility also exists for agricultural land to be used to mitigate climate change, for example using minimum tillage regimes (to reduce carbon release or combat soil erosion) or growing biofuels such as short-rotation coppice (Parry 2000). In the shorter term habitat diversity may increase; however, the long-term impacts on farmland biodiversity are likely to be indirect, through, for example, increased water abstraction or changes in cultivation schedules.

Research and management directions

Land in Europe is required to serve multiple uses, with a diverse range of interest groups drawing benefits from it, of which agriculture may not be the most important in economic terms (Pretty 1998). Land managers have a continuum of options, from incorporating social, environmental and agronomic needs on the same land (Bignal 1998), to decoupling them entirely, with a smaller area of land managed for maximum production, allowing some areas to serve other objectives (Avery 1995). This latter approach may allow a range of problems to be tackled, including habitat loss, climate change, improving water quality in drinking water catchments and providing environmental amenity for residents and tourists, but does not address the underlying issues of general ecosystem health. Analyses integrating social, economic and environmental objectives are needed to determine the optimal trade-offs between these in each region, to create a landscape that satisfies many different needs in a relatively restricted area. Incorporating broader environmental objectives within the farming remit could reconcile these multiple requirements.

Many farmers are concerned with 'tidiness' and perceived efficiency, with uncropped areas seen as wasted land (Macdonald & Johnson 2000). Altering this attitude may well be crucial for enhancing farmland biodiversity. More generally, farmers' attitudes are likely to have a major influence on how new technologies are applied, and hence their biodiversity impacts (Watkinson *et al.* 2000). Understanding these processes is likely to be just as important for biodiversity as autecological studies.

Much concern has focused on the disappearance of farmland birds (Krebs *et al.* 1999) and the UK government has pledged to reverse the trends of 20 common farmland specialists by 2020. Given the apparent importance of food resources, measures that increase populations of plants and insects, and allow them to complete their life cycles, are likely to be most effective. This could be achieved by the management of non-cropped habitats for wildlife coupled with a responsive, targeted use of agrochemicals, and could include the use of GM technology, for example for selective control of pernicious weeds such as blackgrass. Pretty (1998) shows that while yields from such systems may be slightly lower (by up to 15%), gross margins are often higher because variable costs are reduced (by up to 50%). More generally, agri-environment schemes currently achieve such objectives through prescriptions on practices that should be employed, rather than the results that are achieved. Providing support for explicit environmental outcomes, achieved in a manner determined by the farmer according to local conditions, may yield more sustainable results and encourage regional habitat heterogeneity (Musters *et al.* 2001).

It is, however, critical to ensure that such measures address the underlying causes of the declines (Kleijn *et al.* 2001). For example, the amount of pesticide active ingredient applied to fields has decreased markedly from rates of 4 kg ha^{-1} for the early hormonal herbicides to 10–25 g ha^{-1} for those in use today. Although this has some benefits, such as reduced pesticide run-off (Skinner *et al.* 1997), the effects on biodiversity are not diminished. Similarly, plant and insect control on organic and farms with reduced inputs ('integrated management') is often as efficient as on conventional farms, although using seemingly less intensive methods (Liebman & Davis 2000; Bond & Grundy 2001); in this case the benefits in terms of increased biodiversity will not be marked. The use of set-aside to reverse farmland bird trends is also unlikely to be successful if other agricultural practices mean no seed resources are available there.

The impacts on biodiversity of arable management are scale-dependent, for example insecticides have marked effects on invertebrate communities at a plot scale, but not necessarily at larger scales (Duffield & Aebischer 1994; Campbell *et al.* 1997). Understanding how, indeed if, results from the many plot- or field-scale studies apply at regional or national scales, and the scaling processes involved, is a key area where research is required. Interactions between factors, for example predation and the structural complexity of habitats

(Donald & Vickery 2000) and regional variation in habitat use (Robinson, Wilson & Crick 2001), also need to be considered. Such distinctions will be critical for agri-environment schemes, which need to develop regionally, and locally, flexible approaches (Bignal, Jones & McCracken 2001). With so many factors impacting on biodiversity, the need for large-scale monitoring to inform environmental management has never been greater.

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