Causes of decline and conservation solutions for Corn Buntings *Emberiza calandra* in eastern Scotland

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DECLARATION

I hereby declare that I alone composed this thesis, that the work is my own, except for contributions from colleagues clearly stated at the start of each chapter, and that this work has not been submitted for any other degree or professional qualification.

Date: 16-08-12

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ABSTRACT

The Corn Bunting *Emberiza calandra* is one of the most severely declining farmland birds across Europe. In the UK, numbers fell by 86% between 1967 and 2008. Corn Buntings favour open landscapes, nest on or close to the ground, are often polygynous, double-brooded, and have a seed-based diet supplemented in summer by invertebrates. This study investigated the recent causes of decline in arable and mixed farmland in eastern Scotland, and sought to identify potential conservation solutions that could be delivered through agri-environment schemes (AES). Combining new data with analyses of existing long-term datasets, I investigated habitat associations during summer and winter, the timing and success of nesting attempts, and measured reproductive and population responses to AES.

Corn Buntings declined almost to extinction in one study area where, over 20 years, the main recorded intensifications of farming were reduced weed abundance within crops and removal of boundaries to make bigger fields. Territory locations, late-summer occupancy and polygyny were all strongly associated with weedy fields. There were also positive associations with overhead wires and in early summer with winter barley and forage grasses. Late-summer occupancy was associated with spring-sown cereals, crops that are amongst the last to be harvested. Changes in habitat associations and to aspects of the mating system as the population declined and agriculture intensified are discussed.

Intensive monitoring showed that Corn Buntings laid clutches from mid-May to mid-August, mostly in fields of forage grasses and autumn-sown cereals in early summer, and spring-sown cereals in late summer. A preference for nesting in dense swards explained this seasonal variation. Breeding success in forage grasses was poor, due to high rates of nest loss during mowing. However, in experimental trials, nest success in fields with delayed mowing was fivefold that of control fields. With sufficient uptake through AES, delayed mowing could raise productivity to levels required to reverse population declines. In winter, cereal stubbles and AES unharvested crop patches were the main foraging habitats used. Unharvested crops with abundant cereal grain in their first winter of establishment were favoured.

Population monitoring over seven years and 71 farms revealed increases on farms with AES targeted at Corn Buntings, no significant change on farms with general AES, and declines on control farms. In arable-dominated farmland, management that increased food availability reversed declines, but on mixed farmland where Corn Buntings nested in forage grasses, delayed mowing was essential for population increase. This study has already influenced the design of AES targeted at Corn Buntings in Scotland, and I make further recommendations for the species' conservation and design of AES that are applicable to farmland throughout Britain and Europe.

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CHAPTER 1. INTRODUCTION – CHANGES IN AGRICULTURE AND FARMLAND BIRD POPULATION DECLINES

The focus of this thesis is one of Scotland's fastest declining bird species, the Corn Bunting *Emberiza calandra*. As the name suggests, it is a farmland specialist strongly associated with cereal cultivation throughout Europe, but populations are in decline across the continent. Detailed summaries of Corn Bunting ecology, population size, trends, and conservation status in Scotland, the UK and Europe are given in Chapter 2, along with an outline of the overall aims of my studies and the content of each subsequent chapter (Chapters 3–8).

First, by way of an introduction, and to put my studies of Corn Buntings into the context of wider issues, Chapter 1 covers some general background information that will lead the reader into understanding the rationale behind the main hypothesis-testing elements of the thesis. I begin with a short review of the changes in agriculture that have taken place in recent decades, both in the UK and elsewhere in Europe, and of the cost this has had to farmland biodiversity. I then summarise recent changes in farmland bird populations at a UK and European level, give a brief assessment of associations between species' ecological and life-history traits and their population trends, and review published studies from across Europe that link agricultural changes to bird population trends. Finally, I briefly outline the history and success of agri-environment schemes in the UK and Europe (the main policy response to farmland biodiversity loss), before ending with a short section to explain why the Corn Bunting in eastern Scotland makes a good case study, both for testing solutions to improve the effectiveness of such schemes, and in the context of the type of farming that still predominates in this region.

1.1. Agricultural intensification

During the last 50 years, global food crop yield per unit area more than doubled, helping to feed a growing human population (Green *et al*. 2005). Increasingly efficient farming methods made this possible, underpinned by mechanisation, plant breeding and the development of agrochemicals to improve crop growth and protection from disease, drought and pests. In Europe, where farmland is the single largest habitat occupying almost half of the continent's land area, this intensification of agriculture has been one of the main environmental changes in recent decades (Donald *et al*. 2006, Reidsma *et al*. 2006). In member states of the European Union (EU), a major driver accelerating agricultural intensification has been a Common Agricultural Policy (CAP) whose subsidies promoting technological innovations and the improvement of land for farming encouraged increased production of crops and livestock. This has enabled the EU to become a net exporter of most of its agricultural products (Buckwell & Armstrong-Brown 2004), with, for example, wheat yield per unit area in France, Germany and the UK increasing by 108%, 95% and 91%, respectively, between 1970 and 2000 (FAOSTAT 2012).

1.1.1. Changes to farming in the UK

In the UK, the main components of agricultural intensification are as follows. The use of chemical products to fertilise the soil and control plant and insect pests and disease (herbicides, insecticides and fungicides, respectively) has increased substantially, in terms of both the area of land treated and the number of applications per growing season. For example, between 1970 and 2000, there was a tripling of the pesticide area treated (Buckwell & Armstrong-Brown 2004), although their persistence in the environment and toxicity to vertebrates has declined with the progressive switch from organochlorines to organophosphates, carbamates and pyrethroids since the 1960s (Shrubb 2003). The use of inorganic fertilisers has replaced traditional methods such as the spreading of manure from livestock, and the growing of nitrogen-fixing crops such as clover in 3-year crop rotations. Furthermore, crop rotations involving root crops and spring and autumn cultivations for weed and disease control were no longer necessary once effective fungicides and herbicides became available (Shrubb 2003). Thus, freed from the necessity of keeping livestock and growing grass and arable crops in rotation, many farmers became more specialised, growing a smaller variety of crops and focusing solely on arable or livestock production. In many parts of the UK, the once abundant mixed farm growing crops alongside cattle, sheep or poultry is now a rarity. Farms in the drier eastern regions have tended to focus on arable production, whilst those in the cooler, wetter west and in marginal upland areas have specialised in grassland-based rearing of livestock for meat or milk (Newton 2004).

Economies of scale have also driven the polarisation of arable and grassland systems, whereby economic returns are greater when farming is simplified and focused on a single product. Such economic factors have frequently resulted in the amalgamation of small family-run farm units into large-scale agri-businesses covering many hundreds of hectares and employing few workers. Mechanisation has allowed this process, as the large, powerful tractors, combine harvesters, and other machinery available today enables a single worker to carry out farming operations quickly over a large area. Mechanisation has also contributed to the loss of non-cropped habitats with the removal of field boundaries to make bigger arable fields, giving more land for crops and allowing large machinery to cover the ground more efficiently. Features lost during this process include hedgerows, at an estimated rate of 5000 km per year between 1947 and 1990 (Shrubb 2003), ditches, banks, walls and other uncultivated patches of land. The overall effect of these losses is the creation of a more homogenous landscape dominated by large contiguous blocks of a single crop type.

One of the most fundamental changes within arable farming systems has been the development of new varieties of cereals that allows the sowing of crops in the autumn instead of spring. In England, 80–90% of cereals are now autumn-sown, compared with 20–30 % in the 1960s (Wilson *et al*. 2009). Many fields therefore have growing crops in them for almost twelve months of the year, in contrast to traditional spring-sown systems where land would remain fallow over the autumn and winter months, often as unploughed stubble. Autumn-sown crops also receive more pesticide inputs than their spring-sown counter-parts, and harvesting is earlier. The change in timing of sowing has also brought about a switch between the main types of cereals grown. Wheat requires a long growing season so is particularly well-suited to autumn-sowing, and in England its area almost doubled between 1970 and 2010, where it now represents 70% of the 2.5 million ha of cereals grown, compared with just 30% of the 3 million ha grown in 1970 (DEFRA 2012). This increase in wheat has been at the expense of barley, which represented 60% of cereals grown in England in 1970, but less than a quarter in 2010 following a two-thirds decline in area (DEFRA 2012). Similarly, the area of oats in England has also more than halved since 1970.

In addition, the development of new varieties of crops resistant to a wider range of environmental conditions has seen the northward spread of ‘new’ crops. Maize, for example, covered just 500 ha in the UK in 1970, but by 2007 its area had increased to 145 000 ha (Wilson *et al*. 2009, FAOSTAT 2012). By contrast, some crops became widespread due to high CAP subsidies for planting them, such as oilseed rape whose area in England increased from less than 4000 ha in 1970 to more than 0.5 million ha in 2007 (Wilson *et al*. 2009).

In pastoral systems, grassland management has changed just as radically. There has been widespread re-seeding of meadows with grass varieties (mainly ryegrasses *Lolium* spp.) that grow rapidly in response to large inputs of inorganic nitrogen fertiliser, whose use doubled between 1970 and 1986 (Vickery *et al*. 2001). Inputs of phosphorus and potassium have also increased, and the effect of these high inputs of inorganic fertilisers combined with re-seeding has been the replacement of slower growing species-rich swards with rapid growing, uniformly dense species-poor swards. This allows meadows to support higher grazing densities of cattle and sheep, and permits earlier and more frequent mowing of grass to make silage, rather than taking a single late cut to make hay. The agronomic benefits of making silage instead of hay are that it produces a greater quantity of fodder, the nutritional quality is better because the grass is cut at an earlier growth stage (Rinne *et al*. 1999), and the crop is less susceptible to damage by wet weather at harvest time. Between 1962 and the mid-1990s, the percentage of forage grass cut for silage increased from 10% to 80%, encouraged in later years by the development in the late 1980s of a method for using plastic to wrap silage in bales (Shrubb 2003).

Grazing regimes have also changed, with a general increase in livestock densities, particularly in the uplands. The number of sheep in the UK rose by 50% between 1976 and 1997, whilst the number of cattle fell by 18% over the same period, with most of the decline involving dairy cattle (Vickery *et al*. 2001). Sheep numbers have declined in recent years, but are still 10–15% above the mid-1970s total (DEFRA 2012). Intensification in livestock systems has also involved keeping animals in larger herds, often kept indoors year-round. For example, whilst the number of pigs in the UK has fallen sharply in recent years to just under 4.5 million in 2010, from 9 million in the early 1970s and 7.5–8 million in the 1990s, 80% of animals in England are currently in herds of 1000 or more (DEFRA 2012). Similarly, poultry rearing for eggs and meat is now largely on an industrial scale, with 99.5% of broilers (birds reared for meat) in England currently in flocks greater than 10 000 (DEFRA 2012).

In both grassland and arable fields, another major aspect of intensification during the 1970s and 1980s was the grant-aided drainage of wet areas using under-field pipes. Half of all land drainage in Britain during the 20th century occurred during this period, with, for example, 20% of farmland in Essex and Lincolnshire drained during the 1970s (Peach *et al*. 2004). Drier soils improve grass swards and crop growth, reduce crop disease, and allow heavy machinery to access land throughout the year.

By the mid-1980s, intensification combined with CAP subsidies for growing certain crop types had led to gross over-production of food within the EU. The policy response was to encourage farmers to take arable land out of production and leave it fallow. ‘Set-aside’, as this policy measure became known, was introduced first as a voluntary measure in 1988, and then became compulsory in 1992 following a change in subsidy payments for cereal and protein crops from a yield to an area basis (Gillings *et al*. 2010). Farmers growing these crops were required to put a proportion (average 10% p.a.) of their land into set-aside each year to qualify for these payments. In its first year as a compulsory measure, set-aside covered more than 0.6 million ha in the UK and 6.4 million ha across the EU (Wilson *et al*. 2009). There were two general types of set-aside. Rotational set-aside involved different patches of land left as fallow each year, and non-rotational set-aside involved the same piece of land taken out of production for several years. Thus, the introduction of set-aside went some way towards reversing intensifications that had removed non-cropped habitats, summer fallows and over-winter stubbles. However, in 2008 following a period of high commodity prices and falling production, the compulsory set-aside scheme ended.

Finally, mention must be made of the rise of 'organic' farming in recent decades. Organic farming is underpinned by the principles of human health being inseparable from the health of ecosystems, and that "organic agriculture should be based on living ecological systems and cycles, work with them, emulate them and help sustain them" (http://www.soilassociation.org/whatisorganic/organicprinciples. Accessed 26 April 2012). It is characterised by the use of traditional methods such as mechanical weeding, crop rotations, under-sowing and biological control instead of pesticides to control weeds and insect pests, animal dung and green manure instead of inorganic fertilisers to replenish the soil, and minimum-tillage methods of cultivation to maintain good soil structure (Hole *et al*. 2005). Consequently, organic farms tend to be similar to the traditional mixed farm of the pre-intensification era, with arable and grassland managed for livestock, spring-sown crops, small fields with sensitive field-boundary management, and patches of non-cropped land. In response to increased consumer demand for food produced using methods with high environmental and animal-welfare standards, and for food perceived to have fewer health risks, the area of farmland under certified organic management across Europe increased from 0.3 million ha in 1990 to 7 million ha in 2006 (Wilson *et al*. 2009). In 2011, the UK's organic land area was 0.7 million ha (4.2% of farmland), but has declined in recent years as sales of organic produce have fallen, associated with higher living costs and consumers becoming less willing or able to pay for expensive organic food (Soil Association 2012).

1.1.2. Changes to farming in other parts of Europe

Most of the changes associated with intensification in the UK have also occurred across the rest of Europe. However, regional differences in agricultural systems, crops grown, and socio-economics mean that other changes in farming practice are associated with intensification. For example, crops rarely or never grown in the UK but widespread in southern Europe include sunflowers, olives, grapes, citrus fruits and rice. Intensification in vineyards, orchards and olive groves includes increased use of pesticides and clearance of ground vegetation from beneath trees (Fournier & Arlettaz 2001, Brambilla *et al*. 2008), whilst the mechanisation of seeding practices has led to changes in the flooding regimes of rice fields (Fasola & Ruíz 1997). Sunflowers have become more widespread, with their area across the three main growers of this crop in the 'old' EU (France, Spain and Italy) increasing ninefold between 1970 and 2000, from 0.2 million ha to 1.8 million ha (FAOSTAT 2012). Similarly, over the same period the area of rice within these three countries increased by 42% (FAOSTAT 2012).

In drier regions of Europe such as central Iberia, intensification of arable land often involves the introduction of irrigation schemes (Súarez *et al*. 1997). Combined with increased use of agro-chemicals, this allows the growing of crops new to those areas such as maize and alfalfa, replacing traditional mosaics of low-input cereals, fallows and grazed meadows (Ursúa *et al*. 2005). In some of these areas, the planting of permanent crops such as olives and vines has also transformed farming systems (e.g. Silva *et al*. 2007).

Land abandonment, the polar opposite to intensification, is also widespread across Europe, especially in regions where environmental constraints such as poor soils or climate restrict farming productivity (e.g. Laiolo *et al*. 2004, Wretenberg *et al*. 2006, Fonderflick *et al*. 2010, Reino *et al*. 2010). Such land eventually reverts to scrub through natural regeneration, or sometimes planted with trees and converted to forestry. One region where abandonment of agricultural land has been widespread is central and eastern Europe. Following the fall of communism, state support for agriculture fell, leading to a drop in pesticide and fertiliser use and a general decline in farming intensity, often involving conversion of arable land to meadows, and afforestation or abandonment of previously cultivated land (Reif *et al*. 2008).

Agricultural intensification and crop yields in central and eastern Europe have lagged behind the 'old' EU countries. Most central and eastern European farms are under 5 ha, and on average, agriculture employs 10% of the national population, compared with 2% in western European countries (Tryjanowski *et al*. 2011). Such small-scale farming, using traditional low-technology methods, maintains an intricate mosaic of crop diversity among a dense network of semi-natural habitats that form the field boundaries. In some countries, however, are large areas of semi-natural grasslands (dry steppes and wet meadows) where extensive grazing is practised (Báldi *et al*. 2005). Large farms using intensive, high-input mechanised methods also occur, especially in those countries ruled under communism up until the late 1980s, where 'collectivist' agriculture gave rise to large state-run farms.

In recent years, many of these countries have joined the EU (e.g. Bulgaria, Czech Republic, Hungary, Poland, Romania), and so their agriculture is now supported and governed by the CAP. Consequently, crop and livestock production is expected to increase with the widespread adoption of intensive farming methods that have previously transformed agriculture across the rest of the EU. The area of high-input crops such as oilseed rape, sunflower and maize is likely to increase at expense of traditional low-input cereals and mixed farming, with further abandonment of farming on marginal land (Nagy *et al*. 2009, Tryjanowski *et al*. 2011). Indeed, between 1995 and 2010, the area of maize increased fourfold in the Czech Republic, and in Romania, the area of oilseed rape expanded massively from 300 ha to more than 0.5 million ha (FAOSTAT 2012).

1.1.3. The cost to biodiversity

Whilst the CAP has delivered cheap food and self-sufficiency for a region with a growing human population, one major consequence of agricultural intensification has been the widespread loss of farmland biodiversity (Krebs *et al*. 1999; Reidsma *et al*. 2006). There have been severe declines in the abundance and diversity of plants and invertebrates (Andreasen *et al*. 1996, Wilson *et al*. 1999, Sutcliffe & Kay 2000, Zechmeister *et al*. 2003), which in turn have reduced the populations of animals that depend upon them for food. These include butterflies (Feber *et al*. 2007, Van Dyck *et al*. 2009), moths (Conrad *et al*. 2006), bees (Carvell *et al*. 2007), mammals (Pena *et al*. 2003, Wickramasinghe *et al*. 2003, Smith *et al*. 2004) and birds (Tucker & Heath 1994, Krebs *et al*. 1999, Donald *et al*. 2001, 2006). It is this last group, birds, in which population monitoring is especially strong, and declines across Europe have been most severe in those species considered farmland specialists (Gregory *et al*. 2005, Voříšek *et al*. 2010).

1.2. Changes in farmland bird populations

1.2.1. UK trends

In the UK, bird populations have been monitored annually since the mid-1960s by the British Trust for Ornithology (BTO) through their Common Bird Census (CBC), and since 1994 their Breeding Bird Survey (BBS). The BBS involves two early-morning visits by a volunteer surveyor during April–June each year to record the number of birds seen or heard along two 1-km transects within a 1-km square (Risely *et al*. 2011). To avoid biased selection by the surveyor, the BTO randomly selects the 1-km squares. In the CBC, which ran from 1962 to 2000, plots were not selected at random, and the other main difference from the BBS was that surveyors made 8–10 visits each year to produce territory maps from multiple registrations of bird locations (Baillie *et al*. 2010). Since the BBS began, the number of 1-km squares surveyed across the UK has more than doubled, from 1570 in 1994 to 3239 in 2010, enabling the calculation of UK population trends for more than 100 species (Risely *et al*. 2011). The increase in coverage has also allowed calculation of trends for an increasing number of species at a country and regional level, and for different habitat categories. In addition, two national breeding bird atlases from surveys undertaken in 1968–72 (Sharrock 1976) and 1988–91 (Gibbons *et al*. 1993) have enabled the measurement of range changes between these two periods, and fieldwork for a third atlas covering wintering and breeding birds in 2007–2011 has just completed.

These two sources of information have revealed severe population declines and range contractions within the UK for several farmland species. Species whose population or breeding range has declined by more than 50% over 25 years are of high conservation concern, and placed on the ‘red list’ of Bird of Conservation Concern (Eaton *et al*. 2009). They are joined by species that have undergone ‘historical decline’ (a severe decline between 1800 and 1995 without substantial recovery), and those listed as ‘globally threatened’ by Birdlife International using IUCN criteria (Eaton *et al*. 2009). Fifteen of the 52 species currently on the ‘red list’ are farmland specialists, as are another 14 of the 126 given ‘amber list’ status (25–50% decline in population or range, historical decline with recent recovery, nationally rare with fewer than 300 breeding pairs or a restricted distribution, internationally important population, or of conservation concern within Europe).

By combining the trends of species with shared habitat preferences, BBS/CBC data are also used to produce composite trends for bird populations in farmland, woodland, wetland, and the marine environment. The UK government uses these wild bird indicators for measuring sustainable development and trends in biodiversity. The farmland bird indicator, or Farmland Bird Index (FBI), incorporates the population trends of 19 species that remain relatively common and widespread throughout the UK, and shows that their combined population has halved since 1970, the biggest decline of any group (Fig. 1.1). The main period of decline appeared to be from the late 1970s to the early 1990s. Table 1.1 shows the estimated population size, distribution, and trends over the long and short term for each of the 19 FBI species. Ten are on the ‘red list’, four on the ‘amber list’ and five on the ‘green list’. For some species, population declines appear to have halted or even shown signs of recovery (e.g. Tree Sparrow, Reed Bunting), whilst others continue to decline (e.g. Grey Partridge, Turtle Dove). Some species trends also vary between the UK countries, perhaps linked to regional variation in land use. Additionally, some species declines have been so severe that they are too rare for BBS monitoring, requiring periodic dedicated surveys to measure populations. These include Corncrake *Crex crex*, Stone Curlew, Cirl Bunting, Red-billed Chough *Pyrrhocorax pyrrhocorax* and Montagu's Harrier *Circus pygargus*.

1.2.2. European trends

Population monitoring schemes similar to those in the UK now operate in most other European countries, although only a few (e.g. Czech Republic, Denmark, Finland, France, Sweden) have data earlier than 1990. Co-ordinated by the European Bird Census Council (EBCC) and BirdLife International, the Pan-European Common Bird Monitoring Scheme (PECBMS) combines data from national schemes to produce European-wide population trends. The PECBMS currently involves 25 countries, including the UK, and has derived long-term population trends for 116 common species (23 farmland) from the early 1980s to 2009, and short-term trends for 145 species (36 farmland) from the 1990s to 2009 (PECBMS 2011). A farmland bird indicator at the European scale has been produced (Fig. 1.2), but note that because the PECBMS includes UK data, the European and UK indicators are not completely independent. Of the 36 species classified as farmland specialists, 20 have declined, six increased, six remained stable and four have an uncertain trend (Table 1.2). Since 1980, their composite population index has fallen by 48%, and their overall biomass has more than halved (Voříšek *et al*. 2010, PECBMS 2011). However, similar to the UK, a period of relative stability appears to have followed steep decline during the 1980s (but note that data from few countries other than the UK contribute to the trend during the 1980s). Other declining European farmland birds with small or restricted distributions (< 50 000 breeding pairs) and not covered by PECBMS include Corncrake, Montagu's Harrier, Great Bustard *Otis tarda*, Little Bustard *Tetrax tetrax*, Lesser Kestrel *Falco naumanni* and European Roller *Coracias garrulus*.

Figure 1.1. *Population trends of common UK birds between 1970 and 2007, subdivided into species guilds according to the main habitat occupied. The lowest (yellow) line represents the Farmland Bird Index (19 species), compared with, from top to bottom, seabirds (blue line, 19 species), all species (red line, 115 species), and woodland species (green line, 38 species).*

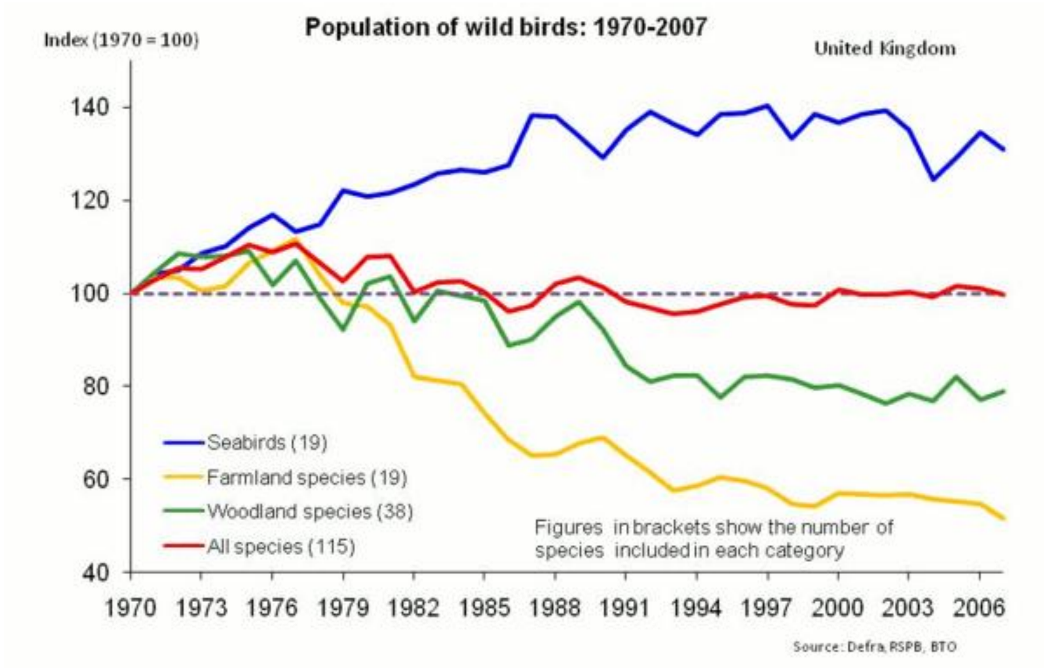


Figure 1.2. *Composite population trend of 36 species of common European farmland birds between 1980 and 2009.*
Source: EBCC/RSPB/BirdLife/Statistics Netherlands
PECBMS (2011), http://www.ebcc.info/index.php?ID=459

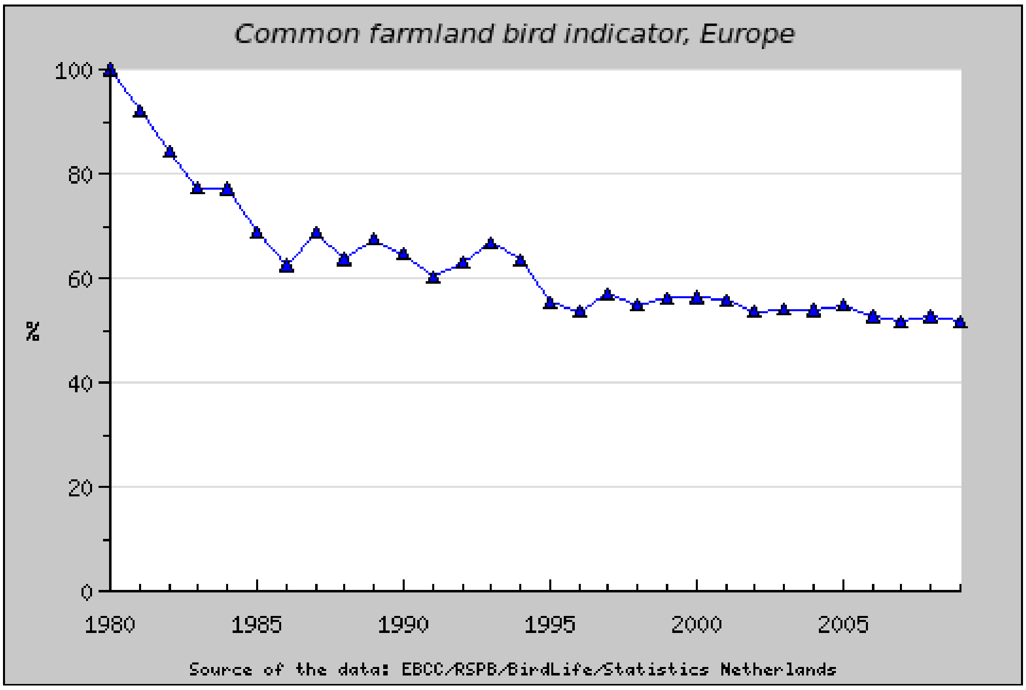


Table 1.1. *Estimated size and change in population (number of breeding pairs) and distribution for each of the 19 species whose trends contribute to the UK Farmland Bird Index. UK-wide and individual country trends are shown. Species are listed in descending order of magnitude of long-term decline. Italics = not classified as a farmland bird at a European scale.* ***Bold*** *= significant trend (95% confidence limits do not overlap zero). na = not applicable due to insufficient data.* [a] *1985–99,* [b] *across 2830 10-km squares in Great Britain and the Isle of Man. UK conservation status shown as superscript next to species name:* [R,A,G] *red, amber and green list, respectively (Birds of Conservation Concern).*
Sources: population size, trends and conservation status from Baillie et al. (2010) and Risely et al. (2011); breeding distribution from Gibbons et al. (1993).

Species	**UK population in year 2000 (1000s)**	**Breeding distribution [b] (occup. 10-km squares)**		**Change in breeding population index (CBC/BBS)**						
				1967–2008		**1998–2008**				
		1988–91	**% change since 1968–72**	**UK**	**Eng**	**UK**	**Eng**	**Scot**	**Wal**	**NI**
Eurasian Tree Sparrow [R]	68	1346	-19.6	na	**-97**	**+56**	**+25**	na	na	na
Grey Partridge [R]	70–75	1629	-18.7	**-90**	**-89**	**-39**	**-29**	na	na	na
European Turtle Dove [R]	44	940	-24.9	na	**-89**	**-64**	**-64**	na	na	na
Corn Bunting [R]	8.5–12.2	921	-32.2	**-86**	**-84**	-12	**-6**	na	na	na
Common Starling [R]	804	2620	-3.6	na	**-85**	**-35**	**-40**	**-21**	**-46**	-16
Yellow Wagtail [R]	11.5–26.5	1047	-9.4	**-77**	**-75**	**-42**	**-43**	na	na	na
Eurasian Linnet [R]	556	2268	-4.6	na	**-76**	**-16**	**-23**	+10	**-37**	**+63**
Common Whitethroat [A]	945	2186	-6.7	**-62**	**-63**	**+13**	**+12**	**+45**	**-16**	na
Eurasian Skylark [R]	1800	2729	-1.6	na	**-61**	-5	**-9**	+11	-17	**-43**
Yellowhammer [R]	792	2224	-8.8	**-56**	**-59**	**-9**	**-16**	+17	**-25**	na
Northern Lapwing [R]	137–174[a]	2340	-9.0	-31	-14	-3	+7	-12	na	na
Reed Bunting [A]	192–211	2188	-11.8	-17	-21	**+37**	**+42**	+49	na	+4
Rook [G]	1100–1400	2237	-0.4	na	na	**-12**	-7	-14	-18	**-32**
Common Kestrel [A]	36.8	2481	-4.1	na	+4	-8	+6	**-41**	na	na
European Greenfinch [G]	734	2323	-2.7	+7	+15	-1	+1	-4	-9	0
European Goldfinch [G]	313	2209	+5.2	na	**+39**	**+52**	**+49**	**+67**	**+21**	**+303**
Eurasian Jackdaw [G]	555	2344	-3.0	**+103**	**+91**	**+22**	**+28**	+9	+11	**+29**
Stock Dove [A]	309	1821	-6.8	na	**+160**	-5	**-8**	na	na	na
Common Woodpigeon [G]	2600–3200	2510	-2.3	**+160**	**+186**	**+27**	**+32**	-1	**+22**	**+30**

Table 1.2. *Estimated size (excluding Russia and Turkey) and change in population (number of breeding pairs) and distribution (within 35–70ºN and 10ºW–30ºE) for each of the 36 species whose trends contribute to the European Farmland Bird Index. A further six species in italics contribute to the UK Farmland Bird Index, but are not considered farmland specialists at a European scale. Species are listed in descending order of magnitude of long-term decline.* ***Bold*** *= regular breeder within UK. n = number of countries contributing to trend, long = 1980–2009, short = 1990–2009. na = not applicable due to insufficient data. Population trend as defined by PECBMS, whereby the category depends on the overall slope estimate and its 95% confidence intervals: D = decline; I = increase; m = moderate (significant trend, but not significantly more than 5% pa because 1.00<lcl<1.05 for increase, or 0.95<ucl<1.00 for decline); s = strong (significant trend, and more than 5% pa because lcl>1.05 for increase, or ucl<0.95 for decline); St = stable (no significant increase or decline, and trend definitely <5% pa because 95% cl enclose 1, lcl>0.95 and ucl<1.05); U = uncertain (no significant increase or decline, but not certain if trends are <5% pa because 95% cl enclose 1, and lcl<0.95 or ucl>1.05).* [a] *index for early period may be unrepresentative due to limited geographical coverage. European conservation status shown as superscript next to species name:* [D] *declining,* [H] *depleted,* [V] *vulnerable,* [S] *secure,* [2] *unfavourable status and concentrated in Europe,* [3] *unfavourable status, not concentrated in Europe,* [E] *favourable status and concentrated in Europe,* [0] *favourable status, not concentrated in Europe. Sources: population size and European conservation status from BirdLife International (2004), population trend information from PECBMS (2011), and distribution from Snow & Perrins (1998).*

Species		**Population (millions)**		**Mean annual % change in population index**		**Population trend**		**Latitudinal range (ºN)**	**Longitudinal range (º)**
Common name	**Latin name**		**n**	**long**	**short**	**long**	**short**		
Crested Lark [H3]	*Galerida cristato*	1.5–3.3	15	[a] -11.80	+2.94	Dm	U	35-55	10W-30E
Grey Partridge [V3]	*Perdix perdix*	1.0–2.3	12	-6.38	-6.43	Dm	Dm	40-65	10W-30E
Ortolan Bunting [H2]	*Emberiza hortulana*	0.7–1.0	9	[a] -6.21	-0.80	Ds	St	35-65	10W-30E
European Turtle Dove [D3]	*Streptopelia turtur*	2.2–3.8	21	-3.89	-0.93	Dm	Dm	35-60	10W-30E
Corn Bunting [D2]	*Emberiza calandra*	4.8–12.7	18	-3.53	-1.70	Dm	Dm	35-60	10W-30E
Eurasian Linnet [D2]	*Carduelis cannabina*	7.5–17.0	23	-3.49	-4.81	Dm	Dm	35-65	10W-30E
Black-tailed Godwit [V2]	*Limosa limosa limosa*	0.1	2	-3.10	-3.64	Dm	Dm	45-60	5W-30E
European Serin [SE]	*Serinus serinus*	7.8–18.0	16	[a] -3.07	-2.72	Dm	Dm	35-60	10W-30E
Northern Lapwing [V2]	*Vanellus vanellus*	1.1–1.7	17	-3.06	-1.76	Dm	Dm	35-70	10W-30E
Yellow Wagtail [S0]	*Motacilla flava*	3.7–6.4	20	-3.04	-1.06	Dm	St	35-70	10W-30E
Meadow Pipit [SE]	*Anthus pratensis*	6.0–13.5	16	-2.67	-3.84	Dm	Dm	45-70	10W-30E
Common Starling [D3]	*Sturnus vulgaris*	19.5–42.0	25	-2.01	-0.77	Dm	Dm	40-70	10W-30E
Eurasian Tree Sparrow [D3]	*Passer montanus*	16.0–27.9	22	-1.93	-1.23	Dm	St	35-65	10W-30E

Table 1.2 cont.

Species		Population (millions)		Mean annual % change in population index		Population trend		Latitudinal range (ºN)	Longitudinal range (º)
Common name	**Latin name**		**n**	**long**	**short**	**long**	**short**		
Whinchat [SE]	*Saxicola rubetra*	3.4–5.0	20	-1.87	+0.04	Dm	St	40-70	10W-30E
Eurasian Skylark [H3]	*Alauda arvensis*	24.1–43.2	25	-1.81	-1.42	Dm	Dm	35-70	10W-30E
Yellowhammer [SE]	*Emberiza citrinella*	14.0–25.0	23	-1.56	-1.01	Dm	Dm	40-70	10W-30E
Eurasian Jackdaw [SE]	*Corvus monedula*	3.2–6.0	21	[a] -1.22	-2.70	Dm	Dm	35-65	10W-30E
Common Kestrel [D3]	*Falco tinnunculus*	0.3–0.4	22	-0.72	-2.76	Dm	Dm	35-70	10W-30E
Reed Bunting [S0]	*Emberiza schoeniclus*	3.4–6.3	17	-0.69	-0.96	Dm	Dm	35-70	10W-30E
Barn Swallow [H3]	*Hirundo rustica*	12.5–26.5	25	-0.59	-1.84	St	Dm	35-70	10W-30E
Lesser Grey Shrike [D2]	*Lanius minor*	0.4–1.0	4	na	-4.71	na	U	35-55	0-30E
Calandra Lark [D3]	*Melanocorypha calandra*	1.0–4.0	3	na	-4.66	na	Dm	35-45	10W-30E
Black-eared Wheatear [H2]	*Oenanthe hispanica*	0.6–0.9	4	na	-1.86	na	Dm	35-45	10W-30E
Woodchat Shrike [D2]	*Lanius senator*	0.4–1.1	6	na	-1.29	na	Dm	35-50	10W-30E
Tawny Pipit [D3]	*Anthus campestris*	0.7–1.3	7	na	[a] -1.12	na	U	35-60	10W-30E
Greater Short-toed Lark [D3]	*Calandrella brachydactyla*	2.3–3.0	3	na	+0.28	na	St	35-45	10W-30E
Common Stonechat [S0]	*Saxicola torquatus*	1.7–3.9	16	na	+0.41	na	St	35-60	10W-30E
Stone Curlew [V3]	*Burhinus oedicnemus*	< 0.1	3	na	+1.19	na	St	35-50	10W-30E
Rock Sparrow [S0]	*Petronia petronia*	0.9–1.4	4	na	+1.28	na	St	35-45	10W-25E
Spotless Starling [SE]	*Sturnus unicolor*	2.1–3.1	3	na	+1.52	na	Im	35-45	10W-15E
Thekla Lark [H3]	*Galerida theklae*	1.5–2.1	2	na	+2.58	na	Im	35-45	10W-5E
Cirl Bunting [SE]	*Emberiza cirlus*	1.9–5.0	6	na	+3.36	na	Im	35-50	10W-30E
Black-headed Bunting [H2]	*Emberiza melanocephala*	0.2–0.6	4	na	+4.87	na	U	35-45	10W-30E
Red-backed Shrike [H3]	*Lanius collurio*	3.9–7.2	21	+0.52	+0.95	St	St	35-65	10W-30E
European Greenfinch [SE]	*Carduelis chloris*	12.4–27.5	25	+0.62	-0.55	Im	St	35-70	10W-30E
Stock Dove [SE]	*Columba oenas*	0.5–0.7	18	+0.85	+1.11	St	St	35-65	10W-30E
Rook [S0]	*Corvus frugilegus*	4.7–7.4	14	+1.16	+0.57	Im	St	35-60	10W-30E
Common Whitethroat [SE]	*Sylvia communis*	8.7–16.1	24	+1.19	+0.75	Im	Im	35-65	10W-30E
Common Woodpigeon [SE]	*Columba palumbus*	8.0–14.5	25	+1.85	+1.79	Im	Im	35-70	10W-30E
European Goldfinch [S0]	*Carduelis carduelis*	9.5–22.5	23	+2.01	+0.24	Im	St	35-60	10W-30E
Eurasian Hoopoe [D3]	*Upupa epops*	0.7–1.1	12	[a] +3.70	+0.05	U	St	35-60	10W-30E
White Stork [H2]	*Ciconia ciconia*	0.2	10	[a] +3.90	+2.01	Im	Im	35-60	10W-30E

1.3. Ecological and life-history traits of common European farmland birds

Across Europe, 36 species classified as farmland specialists are sufficiently common and widespread to be included in the PECBMS, and a composite of their population trends forms the European farmland bird indicator (Fig. 1.2). A further six common and widespread species are considered within the UK to be farmland specialists, and contribute to the UK FBI (Table 1.1). Overall, these 42 species (Table 1.2) encompass 19 families, from some of the continent's largest birds, the storks *Ciconiidae*, to small passerines such as the finches *Fringillidae*. Amongst them is a great diversity of ecological and life-history traits (Table 1.3). In this section, I compare the population trends of these 42 species with their ecological and life-history traits to look for general patterns between the two, because this may help to explain why some species have increased whilst others have declined during the recent era of agricultural intensification. However, this is not a key topic of my thesis, so I do not conduct a detailed statistical analysis, but simply comment on the more obvious general relationships between species trends and traits that are apparent from informal inspection of the information presented in the tables and in Figure 1.3.

Comparative studies that identify groups of species with shared life-history and ecological traits which correlate with population trend (declines versus stable or increase) can be useful for prioritising conservation research (Bennett & Owens 2002). They can also help to identify species that might be vulnerable to environmental change, but whose populations are poorly monitored (Thaxter *et al*. 2010). Several such studies of birds have recently been published (e.g. Hewson & Noble 2009, Amano & Yamaura 2007, Reif *et al*. 2010, Thaxter *et al*. 2010, Van Turnhout *et al*. 2010). These studies have shown various general associations between population trend and body size, lifespan, fecundity, migratory strategy, distribution, diet, nest location, and degree of habitat specialisation. However, limitations of such studies include lack of data for certain traits (e.g. lifespan), and for continent-wide studies, variation in ecology (e.g. migratory tendency, habitat use) and threats across a species' range (Hewson & Noble 2009, Van Turnhout *et al*. 2010). A further complexity is that, arguably, bird species should not be treated as independent sampling units because they are related evolutionarily. In comparative studies, phylogenetic generalised least squares regression, an extension of Felsenstein's (1985) "independent contrasts" method, is often used to control for phylogenetic dependence in the data (Amano & Yamaura 2007, Thaxter *et al*. 2010), although some authors choose to simplify data analyses by avoiding the complexities and drawbacks of correcting for phylogenetic relatedness (e.g. Van Turnhout *et al*. 2010).

Among the 42 farmland species considered here, it appears that declines are more prevalent among ground-nesting species (13 of 17) than in those nesting high in bushes, trees or buildings (eight of 14), and a higher proportion of open-cup (20 of 33) than hole-nesters (three of seven) have declined. Similar associations between declines and ground-nesting were found among 170 breeding bird species in the Netherlands (Van Turnhout *et al*. 2010) and 59 breeding species in England (Thaxter *et al*. 2010), but there was no strong association between nest location and trend among 49 species in English woodlands (Hewson & Noble 2009).

In Table 1.3, species with a specialised diet (either invertebrates only or seeds only) have shown a greater propensity for decline than those with a broader diet. Nine of 13 species with a specialised adult diet have declined, as have 19 of 25 with a specialised chick diet. This compares with declines in 16 of 29 species with a generalist adult diet, and six of 17 with a generalist chick diet. Hewson & Noble (2009) also found greater declines among seed-eaters than generalist plant-eaters that also take foliage and fruit, and Reif *et al*. (2010) found greater declines in seed-eaters than insectivores among 68 passerines in the Czech Republic.

I could find little difference between migratory and non-migratory species, but as previously explained, for many species the migration strategy differs across Europe. However, Sanderson *et al*. (2006) did find greater declines among long-distance migrants (those wintering in sub-Saharan Africa) than non-migratory European species, irrespective of their breeding habitat. Hewson & Noble (2009) also found more declines among long-distance migrants, but Reif *et al*. (2010) found no such relationship among Czech passerines. Thaxter *et al*. (2010) found a more complex pattern, with declines generally greater in Afro-tropical migrants than resident species, but with differences in the timing of declines according to the overwintering bioclimatic zone used (earlier declines in species wintering in arid savannah, and later declines in those wintering further south in humid West African forests and savannah).

There appear to be no clear associations between population trend and body size, annual adult survival rate, maximum number of broods per year, clutch size, combined incubation and fledging period, or fecundity (Figure 1.3). Other studies have found larger declines among passerines with small body mass, short lifespan, high fecundity and large clutch size (Reif *et al*. 2010), and conversely among birds with medium body size and low fecundity (Amano & Yamaura 2007).

Table 1.3. *Ecological and life-history traits of 42 European farmland bird species, sub-divided by population trend (declining over the long or short term, and stable or increasing). Mean values for adult body mass and annual adult survival rate are from Robinson (2005), and all other information is from Snow & Perrins (1998). Max broods = maximum number of broods per female per year. Incub+chick period = number of days from start of incubation to chicks fledging. Fecundity = maximum annual number of young reared per female per year (max broods x mean clutch size). na = data not available.*

Species	Body mass (g)	Adult survival	Max broods	Mean clutch size	Incub + chick period	Fecundity
Declining						
Crested Lark	45	na	3	4	28	12
Grey Partridge	390	0.55	1	15	39	15
Ortolan Bunting	20	na	2	5	25	10
European Turtle Dove	140	0.50	3	2	35	6
Corn Bunting	40	0.58	2	5	26	10
Eurasian Linnet	20	0.37	3	5	28	15
Black-tailed Godwit	310	0.94	1	4	50	4
European Serin	15	na	2	4	28	8
Northern Lapwing	230	0.71	1	4	65	4
Yellow Wagtail	20	0.53	2	5	28	10
Meadow Pipit	20	0.54	2	4	26	8
Common Starling	80	0.69	2	5	33	10
Eurasian Tree Sparrow	25	0.43	3	5	30	15
Whinchat	15	0.47	2	5	31	10
Eurasian Skylark	40	0.51	4	4	30	16
Yellowhammer	30	0.54	3	4	25	12
Eurasian Jackdaw	220	0.69	1	5	50	5
Common Kestrel	205	0.69	1	5	58	5
Reed Bunting	20	0.54	3	5	24	15
Barn Swallow	20	0.37	2	5	35	10
Lesser Grey Shrike	45	na	1	6	32	6
Calandra Lark	60	na	2	5	32	10
Black-eared Wheatear	15	na	1	5	25	5
Woodchat Shrike	35	na	2	6	32	12
Tawny Pipit	30	na	2	5	26	10
Stable or increasing						
Greater Short-toed Lark	25	na	2	4	26	8
Common Stonechat	15	na	3	5	27	15
Stone Curlew	470	0.83	1	2	64	2
Rock Sparrow	30	na	2	5	32	10
Spotless Starling	90	na	2	5	33	10
Thekla Lark	35	na	2	4	28	8
Cirl Bunting	25	na	3	4	24	12
Black-headed Bunting	30	na	1	5	29	5
Red-backed Shrike	30	na	2	5	29	10
European Greenfinch	30	0.44	2	5	29	10
Stock Dove	300	0.55	4	2	42	8
Rook	310	0.79	1	4	50	4
Common Whitethroat	15	0.39	2	5	22	10
Common Woodpigeon	450	0.61	2	2	45	4
European Goldfinch	15	0.37	3	5	29	15
Eurasian Hoopoe	70	na	2	7	43	14
White Stork	3300	na	1	4	94	4

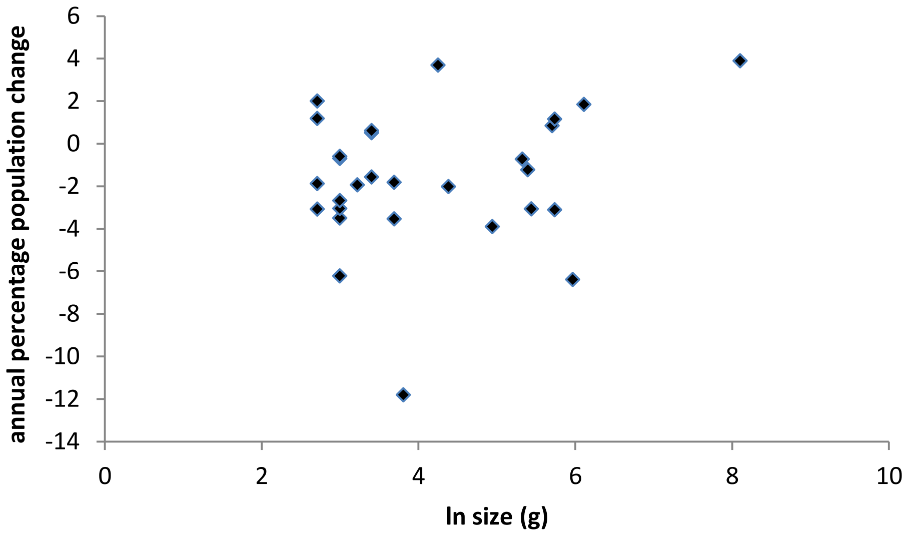
Table 1.3 cont.

Migrant status L = long-distance migrant (vacates breeding areas in winter, and main wintering grounds are in Africa or Asia), S = short-distance migrant (vacates breeding areas in winter, but main wintering grounds are within Europe); Diet Seed = seeds, Veg = seeds and non-seed plant matter including, leaves and fruits, Arth = arthropods (mostly insects and spiders), Omniv = animal and plant matter, Anim = invertebrates and vertebrates, Aerial = aerial insects taken in flight; Foraging location herb = low herbaceous vegetation; Nest type open = open cup, encl = enclosed/hole; Nest location low = < 2 m above ground, mid = 2–3 m, high = >3m. All other categorisations are self-explanatory.

Species	Migrant	Adult diet	Chick diet	Foraging location	Nest type	Nest location
Declining						
Crested Lark	No	Omniv	Arth	ground	open	ground
Grey Partridge	No	Veg	Omniv	ground	open	ground
Ortolan Bunting	Yes L	Omniv	Arth	variable	open	ground
European Turtle Dove	Yes L	Seed	Seed	ground	open	high
Corn Bunting	Some S	Omniv	Omniv	ground	open	ground
Eurasian Linnet	Some S	Seed	Seed	ground/herb	open	low
Black-tailed Godwit	Yes L	Arth	Arth	ground	open	ground
European Serin	Some S	Seed	Seed	ground/herb	open	high
Northern Lapwing	Yes S	Arth	Arth	ground	open	ground
Yellow Wagtail	Yes L	Arth	Arth	ground/aerial	open	ground
Meadow Pipit	Some S	Omniv	Arth	ground	open	ground
Common Starling	Some S	Omniv	Arth	ground	encl	variable
Eurasian Tree Sparrow	No	Omniv	Omniv	ground	encl	high
Whinchat	Yes L	Omniv	Arth	ground/herb	open	ground
Eurasian Skylark	Some S	Veg	Arth	ground	open	ground
Yellowhammer	Some S	Omniv	Omniv	ground	open	low
Eurasian Jackdaw	Some S	Omniv	Omniv	ground	encl	high
Common Kestrel	Some S	Anim	Anim	ground	variable	high
Reed Bunting	Some S	Omniv	Arth	variable	open	low
Barn Swallow	Yes L	Aerial	Aerial	aerial	open	high
Lesser Grey Shrike	Yes L	Arth	Arth	ground	open	high
Calandra Lark	No	Omniv	Arth	ground	open	ground
Black-eared Wheatear	Yes L	Omniv	Arth	ground	variable	ground
Woodchat Shrike	Yes L	Arth	Arth	ground/aerial	open	high
Tawny Pipit	Yes L	Omniv	Arth	ground	open	ground
Stable or increasing						
Greater Short-toed Lark	Yes L	Omniv	Omniv	ground	open	ground
Common Stonechat	Some S	Omniv	Arth	ground	open	ground
Stone Curlew	Most L	Arth	Arth	ground	open	ground
Rock Sparrow	No	Omniv	Arth	ground	encl	variable
Spotless Starling	No	Omniv	Arth	ground	encl	high
Thekla Lark	No	Omniv	Omniv	ground	open	ground
Cirl Bunting	No	Omniv	Omniv	ground	open	low
Black-headed Bunting	Yes L	Omniv	Omniv	variable	open	low
Red-backed Shrike	Yes L	Anim	Anim	variable	open	mid
European Greenfinch	Some S	Seed	Omniv	variable	open	variable
Stock Dove	Some S	Veg	Veg	ground	encl	high
Rook	Some S	Omniv	Omniv	ground	open	high
Common Whitethroat	Yes L	Omniv	Arth	herb/bush	open	low
Common Woodpigeon	Some S	Veg	Veg	ground	open	high
European Goldfinch	Some S	Seed	Omniv	herb/trees	open	variable
Eurasian Hoopoe	Most L	Arth	Arth	ground	encl	variable
White Stork	Most L	Anim	Anim	ground	open	high

Figure 1.3. *Life-history traits plotted against long-term European population trend for 29 farmland bird species.*

a) *Body mass (natural log)*



b) *Annual adult survival*

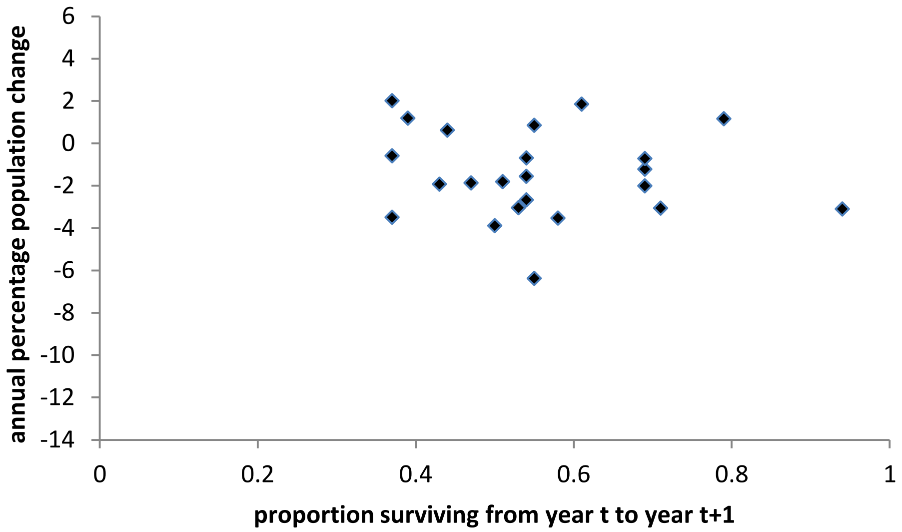
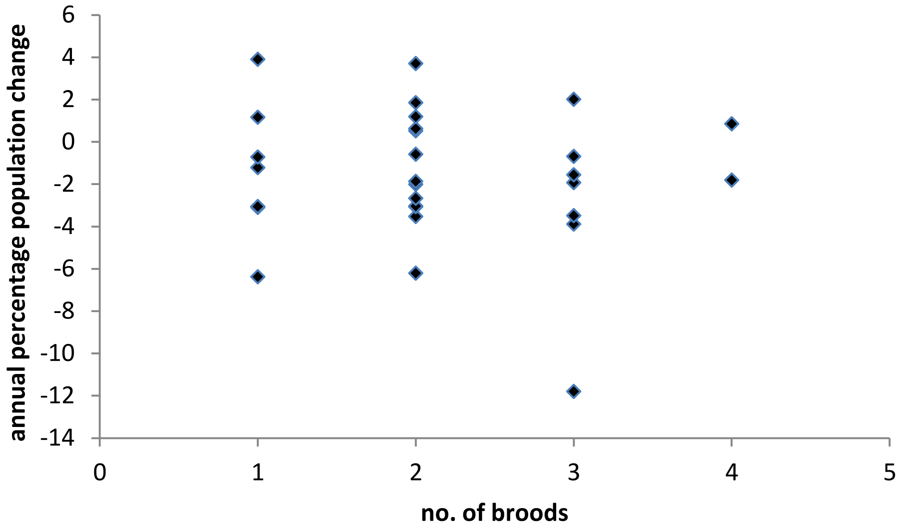


Fig. 1.3 cont.

c) *Maximum number of broods per year*



d) *Mean number of eggs per clutch*

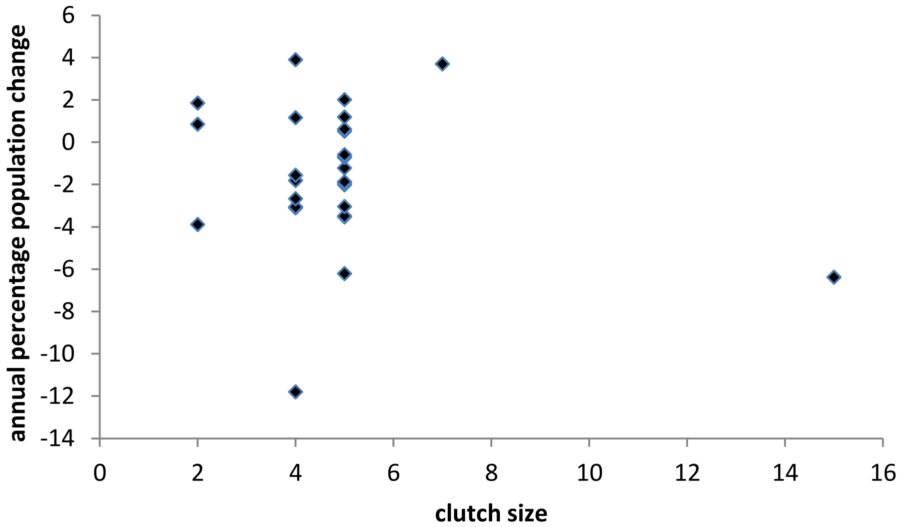
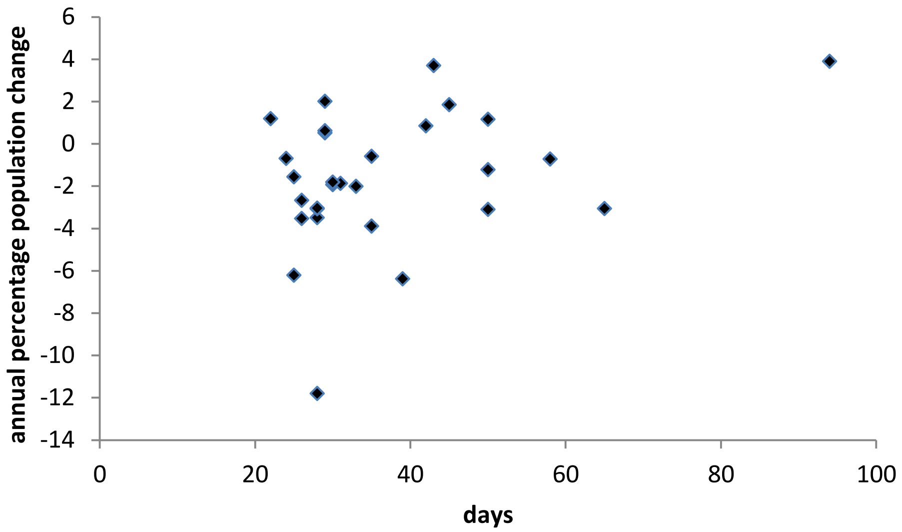
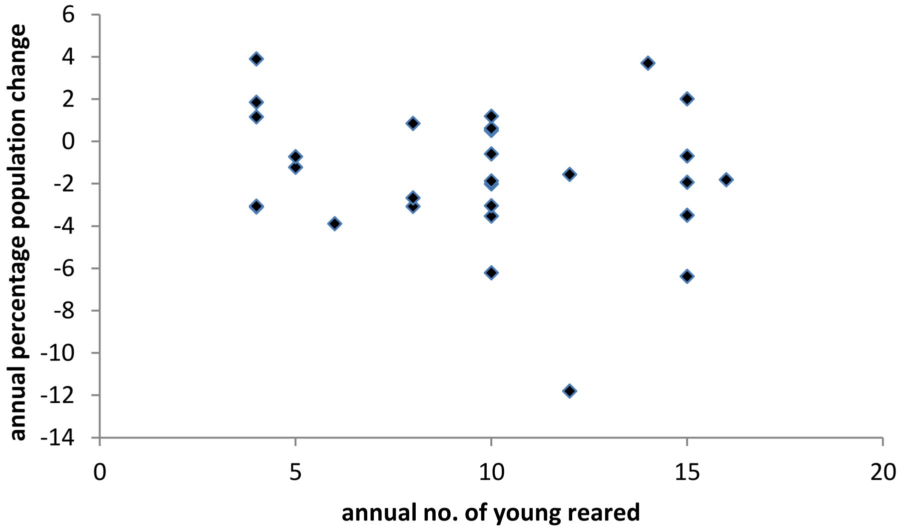


Fig. 1.3 cont.

e) *Nest and chick period*



f) *Fecundity*



From this simple, informal assessment, therefore, it appears that ecological rather than life-history traits are more closely associated with population trends of common European farmland birds, with trends differing between species according to preferred nest location and the types of food resources they depend upon. In reality, relationships are more complex, involving interactions between life-history and ecological traits that affect demography, and I do not explore these here. Instead, in the next section I review more targeted local studies from across Europe to summarise relationships between these traits and recent changes in agriculture that have affected demographic rates and population trends of the 42 species considered.

1.4. Agricultural changes driving population trends of farmland birds

Changes in farming can affect fecundity and annual survival rates of birds in several ways. Table 1.4 shows some of the mechanisms by which changes in land management can interact with specific traits to influence population trends. Not all effects are negative or consistent, as environmental change may adversely affect one species but benefit another. In some cases, one aspect of a species' lifecycle may improve in response to land use change whilst another could deteriorate. For example, the introduction of oilseed rape may increase fecundity via increased seed food availability during the breeding season, but the loss of over-winter stubble associated with an autumn-sown crop could also reduce survival via a reduction in winter seed food.

In Table 1.5, I summarise the findings of studies that aimed to understand the habitat preferences and ecology of the 42 species considered, and how changes in land use may have affected their population trend. The effects (positive, negative and mixed) of several aspects of agricultural intensification on population trend are listed, and where possible I have attempted to distinguish between effects on fecundity and survival. However, few studies have actually measured survival, because a robust assessment depends upon capturing and marking individual birds. Consequently, most studies suggesting an effect of changes in agricultural land use on annual survival are based on foraging habitat preferences and changes in food availability.

1.4.1. Arable management intensity

Of 25 species where studies have shown that their population trends were likely to have been affected by arable intensification, effects were entirely negative for 19 species, and mixed (or inconsistent between survival and fecundity) for another six species. Changes with wholly negative effects on birds include the increased use of herbicides and insecticides that remove weed plants and invertebrates from the food chain. This can adversely affect adult survival rates and fecundity of seed-eating specialists such as Turtle Dove and Linnet through loss of weed seed food (Browne & Aebischer 2003, Moorcroft *et al*. 2006), and reduce fecundity in species that feed seeds or invertebrates to their chicks. For example, studies of Grey Partridge (Rands 1985), Yellowhammer (Hart *et al*. 2006) and Corn Bunting (Brickle *et al*. 2000) have all shown that greater use of pesticides can reduce chick condition and survival through loss of invertebrate food. During winter, Cirl Buntings, Reed Buntings, Yellowhammers and Skylarks selected overwinter stubbles with high weed seed densities, where the preceding crop had received fewer herbicide applications than normal (Bradbury *et al*. 2008, McKenzie *et al*. 2011). Stricter regulation and testing, however, mean that pesticides are now far less toxic to non-target animals and less persistent in the environment than they were in the past (Shrubb 2003). During the 1950s and 1960s, direct toxicity effects of organochlorine pesticides on birds was a major driver of population declines in pigeons, buntings and finches through increased mortality (poisoning) or reduced reproductive success (eggshell thinning and embryotoxicity) from consuming seeds dressed with insecticides (O’Connor & Mead 1984, Crick 1997). Accumulation of organochlorines within the food chain led to similar effects and population declines in birds of prey (Newton 2004).

The switch from spring to autumn sowing of cereals has been detrimental for many species, especially those that forage for seeds on the ground during winter. Larks, finches and buntings make great use of crop stubbles outside of the breeding season, but autumn-sowing replaces stubbles with crop swards that are seed-deficient and avoided by most birds (e.g. Wilson *et al*. 1996, Moorcroft *et al*. 2002). Thus, the widespread loss of overwinter stubbles is likely to have been instrumental in population declines by reducing the availability of winter seed food and suppressing annual survival rates. Reduced survival is sufficient to have caused population declines in several UK species, including Reed Bunting (Peach *et al*. 1999, Siriwardena *et al*. 1999).

Recent studies have shown positive associations between the area of overwinter stubble and population trends of Skylark and Yellowhammer (Gillings *et al*. 2005), and that provision of supplementary food in winter can have a positive effect on local trends in House Sparrow *Passer domesticus* and Yellowhammer (Hole *et al*. 2002, Siriwardena *et al*. 2007).

For some species, sowing crops in autumn instead of spring can also reduce fecundity. The loss of spring-cultivated land reduces the availability of short, sparse swards favoured by nesting Lapwings (Sheldon *et al*. 2005), whilst rapid growth of autumn cereal swards forces Skylarks to nest close to 'tramlines' (narrow linear pathways through the crop made by tractors) later in the breeding season, where nest predation rates are high (Donald *et al*. 2002). Tall dense swards can also restrict access to a variety of species that forage on the ground, both large and small (e.g. Mason & Macdonald 2004, Menz *et al*. 2009a,b). For crop-nesting multi-brooded species such as the Corn Bunting, earlier harvesting of autumn-sown crops can also shorten the breeding season, restricting females to just one brood (Brickle & Harper 2002). However, autumn-sown cereals can be beneficial. For example, earlier ripening cereals provide an additional source of food for buntings provisioning chicks, especially during cold or wet periods when invertebrates are scarce (Evans *et al*. 1997, Stoate *et al*. 1998, Brickle & Harper 1999, Douglas *et al*. in press), whilst Lapwings select autumn-sown cereal fields for feeding during winter (Gillings *et al*. 2007).

1.4.2. Grassland management intensity

For many species, the effects of grassland management intensity are mixed. Brotons *et al*. (2005), for example, found that densities of Tawny Pipit and Greater Short-toed Lark on natural steppe grasslands were greatest in areas next to improved pastures, probably because insect abundance was higher in the farmed habitats, but neither species occurred on pastures in the absence of steppe. In particular, the effect of grazing intensity is complex, differing between the type of livestock involved, and with preferences varying between bird species. In a study of bird assemblages in Hungarian grasslands, Báldi *et al*. (2005) found that 13 species (e.g. Whinchat, Yellow Wagtail) were more abundant on extensively grazed grassland (0.5 cows ha^{-1}), 18 (e.g. Lapwing, Stone Curlew) were more abundant on intensively grazed grassland (>1 cow ha^{-1}), and for 15 species (e.g. Kestrel, Tawny Pipit) results were inconsistent between regions. However, what is categorised as 'intensive' in Hungary is 'extensive' in countries such as the UK, where high grazing densities increase the risk of nest losses to trampling in ground-nesting birds such as Lapwing (e.g. Hart *et al*. 2002). Wakeham-Dawson *et al*. (1998) also showed that intensive grazing may deter ground-nesting birds from settling (Skylarks in this case), by lowering sward heights to below that needed for good nest concealment and for chick-food invertebrates to thrive. At the other end of the scale, under-grazing can lead to scrub encroachment, and ultimately to the loss of grassland altogether. In some regions this is a serious problem, and has led to local declines of grassland birds such as larks, pipits, wheatears, shrikes and buntings (e.g. Laiolo *et al*. 2004, Brambilla *et al*. 2007, Tsiakiris *et al*. 2009).

Earlier and more frequent mowing of grass associated with the switch from making hay to silage can have serious negative effects for meadow-nesting species such as Whinchat and the globally-threatened Corncrake, by destroying nests, killing chicks and sometimes killing incubating females (Green *et al*. 1997, Müller *et al*. 2005, Grüebler *et al*. 2008). Schekkerman *et al*. (2009) also found that Black-tailed Godwit chicks suffered higher rates of predation in newly mown fields with reduced vegetation cover. Conversely, more frequent mowing may benefit ground-foraging species, by increasing accessibility to invertebrate food. Rooks and Jackdaws forage mainly in grasslands (Mason & Macdonald 2004), and in two studies favoured intensively managed fields with short swards (Barnett *et al*. 2004, Atkinson *et al*. 2005). Starlings also make great use of newly cut meadows, where their foraging efficiency is greater than in taller swards (Devereux *et al*. 2006). White Stork breeding success was greater when mowing of grasslands was asynchronous, giving a continuous supply of newly cut fields where chick-food availability was high (Johst *et al*. 2001). In Italy, radio-tracking showed that Stone Curlews foraged at night in newly cut grass fields where soil invertebrates were easier to access, and thus benefited from frequently mown fast-growing swards (Caccamo *et al*. 2011). However, for Whinchat, frequent cutting reduced the abundance of its foliage-dwelling invertebrate prey, lowering chick survival (Britschgi *et al*. 2006).

Other aspects of grassland intensification include re-seeding with ryegrasses and increased use of chemical fertilisers. This creates rapid-growing dense swards, which can reduce access to food for ground-foraging birds (Atkinson *et al*. 2005). Studies have shown negative associations between sward density and foraging habitat selection or efficiency for Meadow Pipit (Vandenberghe *et al*. 2009), Yellowhammer (Douglas *et al*. 2009), Lesser Grey Shrike (Wirtitsch *et al*. 2001), Common Kestrel (Aschwanden *et al*. 2005, Garratt *et al*. 2011), Barn Owl (Arlettaz, *et al*. 2010b) and Black-tailed Godwit chicks (Kleijn *et al*. 2010).

1.4.3. Crop diversity

There were positive associations with crop diversity for 27 of the 42 species, and no negative associations. One of the main aspects of reduced crop heterogeneity associated with species declines is the loss of arable or grassland from farms as production becomes more specialised. In the UK, for example, Robinson *et al*. (2001) using BBS data found that several species including Grey Partridge, Tree Sparrow, Yellowhammer, Reed Bunting and Corn Bunting became less numerous in grassland dominated landscapes as the proportion of arable decreased. All five feed predominantly on crop and weed seeds associated with arable cultivations, especially during winter when invertebrate availability is low. They are among those species whose range has contracted the most since the 1970s (Table 1.1), with local extinctions in northern and western parts of Britain coinciding with a loss of arable from those regions. Conversely, some species have declined in arable dominated landscapes with the loss of grassland and livestock. Studies from across Europe have shown that breeding success in Swallows is higher on farms with cattle and grazed pastures, due to greater abundance of aerial insects. Effects include increased clutch size, greater chick survival and more second broods when cattle are present, and population declines following the loss of cattle (Møller 2001, Ambrosini *et al*. 2002, 2011, Evans *et al*. 2007, Grüebler *et al*. 2010). Grassland is also important for breeding waders. Although some species such as Lapwing and Stone Curlew occupy arable areas and frequently nest in fields of spring-sown crops where they favour the short sparse swards, they also require invertebrate-rich grassland nearby as chick-rearing areas (Galbraith 1988, Green *et al*. 2000).

Heterogeneity of crop types can also increase fecundity in multi-brooded ground-nesting species, because differential growth rates of crop swards allows birds to switch habitats between nesting attempts. Examples of species that make sequential use of different crops as the breeding season progresses include Skylark (Wilson *et al*. 1997) and Yellow Wagtail (Gilroy *et al*. 2010, Kragten 2011).

1.4.4. Area of fallow or semi-natural habitat

For all 31 species associated with the area of fallow or semi-natural habitat, the relationship was positive. Such land is relatively unaffected by agricultural activities like pesticide use, cutting and ploughing, and can act as a refuge for plants and animals. Invertebrate communities tend to be larger, and plant communities more mature, giving greater sward heterogeneity. For ground-nesting species, there is less of a threat of agricultural operations destroying nests, or rapid, homogenous crop growth creating dense swards and restricting accessibility. Some of the strongest relationships are found amongst species characteristic of low-intensity farmland in southern, central and eastern Europe, including those associated with pseudo-steppes such as the larks, Stone Curlew and bustards where long-term fallows are important nesting and foraging habitats (e.g. Tella *et al*. 1996, Moreira 1999, Moreira *et al*. 2005).

Table 1.4. *Some changes in agricultural practice associated with intensification, and their potential effects on the fecundity and survival of farmland birds. These are hypothesised relationships, but the studies cited in Table 1.5 confirm that most of them are real effects.*
Reduced/increased fecundity = fewer/more young reared per female per year, due to higher/lower rates of nest failure, lower/higher chick survival, or fewer/more nesting attempts made. Reduced/increased survival = lower/higher probability of an adult or juvenile surviving until the following breeding season, due to a greater/lower risk of mortality from starvation, predation or disease associated with reduced/increased availability or accessibility of food, or greater/lower risk of direct mortality caused by predation (associated with loss of cover), or by agricultural operations.

Broad change in agriculture	**Specific change in farming**	**Hypothesised demographic effect**	**Mechanism**
Arable intensification	Increased use & efficacy of pesticides	Reduced fecundity	Increased chick mortality due to less invertebrate and weed seed food. Increased nest predation due to less weed cover for ground-nesters. Greater risk of nest destruction with increased farming operations in fields.
		Reduced survival	Increased mortality due to less invertebrate and weed seed food.
	Switch from spring to autumn sowing of crops	Reduced fecundity	Greater risk of nest destruction with earlier crop harvest. Fewer nesting attempts possible with earlier crop harvest.
		Reduced survival	Increased mortality with loss of over-winter stubbles and associated seed food.
		Increased fecundity	Less chick mortality in early broods with greater availability of ripening crop seeds.
		Increased survival	Reduced mortality with greater availability of crop vegetation food during autumn and winter.
	Development of fast-growing & high yield crop varieties	Reduced fecundity	Fewer nesting attempts possible with denser swards and rapid crop growth.
	Use of larger & more powerful machinery	Reduced survival	Increased mortality as more efficient harvesting removes crop seed food.
Grassland intensification	Switch from a single cut for hay to multiple cuts for silage	Reduced fecundity	Increased nest destruction with earlier and more frequent cutting. Increased chick mortality due to less invertebrate food. Fewer nesting attempts possible with earlier and more frequent cutting.
		Increased fecundity	Less chick mortality with increased access to food in recently cut swards. More nesting attempts possible for species that favour short swards.
		Reduced survival	Increased mortality of incubating females with earlier and more frequent mowing.
		Increased survival	Reduced mortality with increased access to food in recently cut swards.

Table 1.4 cont.

Broad change in agriculture	Specific change in farming	Demographic effect	Mechanism
Grassland intensification	Increased use of inorganic fertilisers & re-seeding with high-yield grasses	Reduced fecundity	Increased nidifugous chick mortality caused by reduced access to food with dense and rapid sward growth.
		Reduced survival	Increased mortality with reduced accessibility to (and loss of) seed, invertebrate and animal food in less diverse swards.
		Increased survival	Less mortality with greater availability of plant food.
	Increased livestock grazing densities	Reduced fecundity	Greater risk of nest destruction through trampling by livestock. Higher nest predation rates in shorter swards. Grazing prevents development of scrub, limiting nesting opportunities.
		Increased fecundity	Less chick mortality with increased access to food in short swards. Grazing maintains breeding habitat by preventing scrub encroachment.
Increase in field size	Loss of field boundaries	Reduced fecundity	Fewer nesting opportunities for hedgerow/field edge nesting species. Increased chick mortality due to less invertebrate and weed seed food.
		Increased fecundity	Nests further from field boundaries have lower predation rates.
		Reduced survival	Increased mortality with loss of rough grassland and associated prey.
		Increased survival	Reduced predation risk in open habitats.
Introduction of new crops	Oilseed rape	Reduced fecundity	Fewer nesting attempts possible with taller swards and rapid crop growth.
		Increased fecundity	Less chick mortality with greater availability of ripening crop seeds. Increased nesting opportunities in tall, dense swards.
		Increased survival	Reduced mortality with greater availability of crop vegetation food during autumn and winter.

Table 1.5. *Associations between aspects of agriculture subjected to recent change, and the fecundity and annual survival of 42 species of European and British farmland birds. F = fecundity; S = survival; G = general association; n = nest survival; c = chick survival; a = annual nesting attempts per pair;* **-** *negative effect; + positive effect; m mixed effect. Gaps in table = species studied but no clear effect shown, or association not studied for that species.*

Species	Arable intensity [a]	Grassland intensity [b]	Crop diversity [c]	Area of fallow or semi-natural habitat [d]	Field wetness [e]	Area of scrub or woodland [f]	Field size [g]	New crops [h]
Declining								
Crested Lark	-G	+G		+G	-G	-G	+G	
Grey Partridge	-Fc -S		+G	+Fanc +S		-G	-Fanc	
Ortolan Bunting	-Fac	mFac	+Fac	+Fac		mG	-Fa	
European Turtle Dove	-Fac mS	-Fc	+Fac	+Fac		mFa	-Fa	+Fc
Corn Bunting	-Fanc -S	mG	+Fac +S	+Fac		mG	-G	-Fn +S
Eurasian Linnet	-Fac -S	mFac	+Fac +S	+Fac +S		mFa	-Fa	+Fac +S
Black-tailed Godwit		-Fnc mG			+G	-G	+G	
European Serin						mG		+G
Northern Lapwing	-Fan +S	-Fanc mS	+Fac	+Fanc	+G	-G	+Fn	
Yellow Wagtail	-Fanc	-Fanc	+Fac	+Fac	+G	-G	mF	-Fan
Meadow Pipit		-Fanc -S	+G	+Fac +S	+G	-G	+G	
Common Starling	-G	mFac mS	+Fac	+Fac	+G	mFa		-G
Eurasian Tree Sparrow	-Fac -S		+Fac +S	+Fac	+Fc	mFa	-Fa	+S
Whinchat		-Fnc -S	+Fac	+Fa		mG	-G	
Eurasian Skylark	-Facn -S	-Fanc -S	+Fa	+Fac +S		-G	+Fn	-Fa +S
Yellowhammer	-Fnc -S	-G	+Fac +S	+Fac +S		mG	-Fa	+S
Eurasian Jackdaw		+G +S	+G	+S		mG	-Fa	
Common Kestrel		mFac mS	+G	+Fac +S		-G	-Fac -S	
Reed Bunting	-Fc -S	-G	+Fac +S	+Fanc +S	+G	mG	-Fa	mF +S
Barn Swallow		mFac mS	+Fac			mG	-Fc -S	
Lesser Grey Shrike		mFac	+Fac	+Fac +S		mG	-G	
Calandra Lark		mG		+G	-G	-G	+G	
Black-eared Wheatear		+G			-G	mG		
Woodchat Shrike		mG		+G		mG	-G	
Tawny Pipit	-G	mF		+G	-G	-G	+G	

Table 1.5 cont.

Species	Arable intensity [a]	Grassland intensity [b]	Crop diversity [c]	Area of fallow or semi-natural habitat [d]	Field wetness [e]	Area of scrub or woodland [f]	Field size [g]	New crops [h]
Stable or increasing								
Greater Short-toed Lark	-G	mG		+G	-G	-G	+G	
Common Stonechat		-G		+Fac		mG	-G	
Stone Curlew	-Fanc mS	-Fanc	+Fac	+Fanc +S	-G	-G	+G	
Rock Sparrow								
Spotless Starling		+G				-G		
Thekla Lark		-G		+G	-G	mG	-G	
Cirl Bunting	-Fc -S	-Fc	+Fac +S	+Fac		mG	-Fa	
Black-headed Bunting	-G					mG	-G	
Red-backed Shrike		-Fac	+Fa	+Fac		mG	-G	-Fa
European Greenfinch	-Fc -S	+G				mG	-Fa	
Stock Dove	mS					mG	-Fa	
Rook	mG	mG +Fc	+Fc		+Fac	mFa	-Fa	
Common Whitethroat			+Fa	+Fac		mG	-Fanc	+Fa
Common Woodpigeon	mS	mG	+S	+G		mG	-Fa	+S
European Goldfinch	-S	+G		+S		mG	-Fa	
Eurasian Hoopoe	-Fac -S	-G	+Fac	+Fac		mG	-Fac	
White Stork		mFac	+Fac	+Fac	+G	-G		+S

[a] *increased intensity with greater use of agro-chemicals, autumn-sowing, less over-winter stubble, and earlier harvest.*
[b] *increased intensity with greater use of agro-chemicals, re-seeding, multiple and earlier cuts, higher livestock grazing densities.*
[c] *increased diversity with more arable crop types and mixed arable/grass systems.*
[d] *refers to summer fallow, set-aside, and also includes heathland, wetland, and semi-natural grassland.*
[e] *intensification can reduce field wetness through drainage, and increase wetness through irrigation of arable crops.*
[f] *includes the effects of land abandonment and encroachment of scrub, and afforestation.*
[g] *bigger fields associated with more open landscape and fewer field-boundary features such as hedgerows, trees, ditches, grass margins.*
[h] *includes the effects of increased area of maize, oilseed rape, rice and arable silage.*

Supporting references for Table1.5:

Crested Lark:	Tucker & Heath 1994, Suárez *et al*. 2004, Báldi *et al*. 2005, Reino *et al*. 2009
Grey Partridge:	Rands 1985, 1986, Potts & Aebischer 1995, Bro *et al*. 2000, 2001, Robinson *et al*. 2001, Aebischer & Ewald 2004
Ortolan Bunting:	Golawski & Dombrowski 2002, Kujawa 2004, Fonderflick *et al*. 2005, 2010, Vepsalainen *et al*. 2005, Berg 2008, Brotons *et al*. 2008, Menz *et al*. 2009a,b, de Groot *et al*. 2010
European Turtle Dove:	Browne & Aebischer 2003, 2004, Browne *et al*. 2004, 2005
Corn Bunting:	Crick *et al*. 1994, Donald & Evans 1994, 1995, Donald & Forrest 1995, Donald & Aebischer 1997, Brickle & Harper 1999, 2002, Brickle *et al*. 2000, Mason & Macdonald 2000, Stoate *et al*. 2000, Robinson *et al*. 2001, Golawski & Dombrowski 2002, Kujawa 2002, Taylor & O'Halloran 2002, Báldi *et al*. 2005, Scozzafava & De Sanctis 2006, Lilleør 2007, Wilson *et al*. 2007a, Fox & Heldbjerg 2008, Kopij 2008, Brambilla *et al*. 2009, Reino *et al*. 2009, Tsiakiris *et al*. 2009, Davey *et al*. 2010a, Peach *et al*. 2011, Setchfield *et al*. 2012
Eurasian Linnet:	Berg & Part 1994, Eybert *et al*. 1995, Siriwardena *et al*. 1999, Siriwardena *et al*. 2000a,b, Mason & Macdonald 2000b, Fuller *et al*. 2001, Moorcroft *et al*. 2002, 2006, Bradbury *et al*. 2003, Laiolo *et al*. 2004, Báldi *et al*. 2005, Woodhouse *et al*. 2005, Reino *et al*. 2009, Davey *et al*. 2010a, Fonderflick *et al*. 2010, Peach *et al*. 2011
Black-tailed Godwit:	Kleijn & van Zuijlen 2004, Báldi *et al*. 2005, Schekkerman & Beintema 2007, Schekkerman *et al*. 2008, 2009, Kleijn *et al*. 2010
European Serin:	Súarez *et al*. 1997
Northern Lapwing:	Beintema & Müskens 1987, Galbraith 1988, Baines 1990, Peach *et al*. 1994, Hart *et al*. 2002, Henderson *et al*. 2002, Taylor & Grant 2004, Sheldon *et al*. 2005, 2007, Milsom 2005, Ottvall & Smith 2006, Gillings *et al*. 2007, Eglington *et al*. 2008
Yellow Wagtail:	Bradbury & Bradter 2004, Henderson *et al*. 2004a, Kujawa 2004, Báldi *et al*. 2005, Wilson & Vickery 2005, Sage *et al*. 2006, Bátary *et al*. 2007, Gilroy *et al*. 2008, 2009, 2010, 2011, Morris & Gilroy 2008, Neumann *et al*. 2009, Kovacs-Hostyanszki *et al*. 2011, Kragten 2011, Peach *et al*. 2011
Meadow Pipit:	Woodhouse *et al*. 2005, Evans *et al*. 2006, Vandenberghe *et al*. 2009
Common Starling:	Tiainen *et al*. 1989, Solonen *et al*. 1991, Whitehead *et al*. 1995, Barnett *et al*. 2004, Devereux *et al*. 2004, 2006, Laiolo 2005, Robinson *et al*. 2005, Freeman *et al*. 2007, Rintala & Tiainen 2008, Davey *et al*. 2010
Eurasian Tree Sparrow:	Robinson *et al*. 2001, Field & Anderson 2004, Siriwardena *et al*. 2007, Field *et al*. 2008, Peach *et al*. 2011
Whinchat:	Berg & Part 1994, Henderson *et al*. 2004a, Kujawa 2004, Orlowski 2004, 2010, Báldi *et al*. 2005, Müller *et al*. 2005, Britschgi *et al*. 2006, Grüebler *et al*. 2008, Broyer 2009, 2011, Tome & Denac 2012

Eurasian Skylark:	Green 1978, Wilson *et al*. 1997, Poulsen *et al*. 1998, Wakeham-Dawson *et al*. 1998, Chamberlain & Gregory 1999, Donald & Vickery 2001, Donald *et al*. 2002, Eraud & Boutin 2002, Gillings *et al*. 2005, Field *et al*. 2007, Morris & Gilroy 2008, Peach *et al*. 2011
Yellowhammer:	Kyrkos *et al*. 1998, Stoate *et al*. 1998, Bradbury *et al*. 2000, 2008, Morris *et al*. 2001, 2005, Robinson *et al*. 2001, Golawski & Dombrowski 2002, Perkins *et al*. 2002, Henderson *et al*. 2004a, Gillings *et al*. 2005, Whittingham *et al*. 2005, Hart *et al*. 2006, Scozzafava & De Sanctis 2006, Birrer *et al*. 2007, Siriwardena *et al*. 2007, 2008, Douglas *et al*. 2009, 2010, Batáry *et al*. 2010b, Dunn *et al*. 2010, McKenzie *et al*. 2011, Peach *et al*. 2011
Eurasian Jackdaw:	Andren 1992, Gregory & Marchant 1996, Barnett *et al*. 2004, Atkinson *et al*. 2005
Common Kestrel:	Aschwanden *et al*. 2005, Butet *et al*. 2010, Garratt *et al*. 2011
Reed Bunting:	Burton *et al*. 1999, Peach *et al*. 1999, 2011, Mason & Macdonald 2000, Robinson *et al*. 2001, Fuller *et al*. 2002, Moorcroft *et al*. 2002, Brickle & Peach 2004, Surmacki 2004, Gruar *et al*. 2006, Orlowski & Czarnecka 2007, Bradbury *et al*. 2008, Siriwardena *et al*. 2008, Davey *et al*. 2010a, Sage *et al*. 2010
Barn Swallow:	Møller 2001, Ambrosini *et al*. 2002, 2011, Evans *et al*. 2003, 2007, Henderson *et al*. 2007, Gruebler *et al*. 2010
Lesser Grey Shrike:	Kristin 1995, Lefranc 1997, Kristin *et al*. 2000, Isenmann & Debout 2000, Wirtitsch *et al*. 2001, Lepley *et al*. 2004, Giralt *et al*. 2008
Calandra Lark:	Moreira 1999, Suárez-Seoane *et al*. 2002, Moreira *et al*. 2005, Brotons *et al*. 2005, Reino *et al*. 2009, 2010, Morgado *et al*. 2010
Black-eared Wheatear:	Mestre *et al*. 1987, Tucker & Heath 1994
Woodchat Shrike:	Schaub 1996, Lefranc 1997, Bechet *et al*. 1998, Isenmann & Fradet 1998
Tawny Pipit:	Tucker & Heath 1994, Brotons *et al*. 2005, Grzybek *et al*. 2008, Fonderflick *et al*. 2010
Greater Short-toed Lark:	Tucker & Heath 1994, Suárez-Seoane *et al*. 2002, Brotons *et al*. 2005, Reino *et al*. 2009, 2010
Common Stonechat:	Woodhouse *et al.* 2005, Birrer *et al*. 2007, Revaz *et al*. 2008, Reino *et al*. 2009
Stone Curlew:	Green & Griffiths 1994, Tella *et al*. 1996, Green *et al*. 2000, Caccamo *et al*. 2011
Spotless Starling:	Renard *et al*. 1998
Thekla Lark:	Tucker & Heath 1994, Reino *et al*. 2009, 2010
Cirl bunting:	Evans & Smith 1994, Evans *et al*. 1997; Peach *et al*. 2001, Stevens *et al*. 2002, Bradbury *et al*. 2008, Brambilla *et al*. 2008
Black-headed Bunting:	Tucker & Heath 1994
Red-backed Shrike:	Lefranc 1997, Vanhinsbergh & Evans 2002, Karlsson 2004, Scozzafava & De Sanctis 2006, Birrer *et al*. 2007, Brambilla *et al*. 2007, 2010, Golawski & Golawska 2008, Golawski & Meissner 2008, Tsiakiris *et al*. 2009
European Greenfinch:	Fuller *et al*. 2001, Báldi *et al*. 2005, Gil-Delgado *et al*. 2009, Davey *et al*. 2010

Stock Dove:	O'Connor & Mead 1984, Davey *et al*. 2010
Rook:	East 1988, Marchant & Gregory 1999, Griffin & Thomas 2000, Kasprzykowski 2003, 2007, Barnett *et al*. 2004, Mason & Macdonald 2004, Atkinson *et al*. 2005, Gimona & Brewer 2006, Olea & Baglione 2008
Common Whitethroat:	Berg & Part 1994, Green *et al*. 1994, Mason & Macdonald 2000b, Fuller *et al*. 2001, Robinson *et al*. 2001, Stoate & Szczur 2001, Birrer *et al*. 2007, Tsiakiris *et al*. 2009
Common Woodpigeon:	Inglis *et al*. 1990, 1997, Inglis *et al*. 1994a,b, Isaacson *et al*. 2002, Barnett *et al*. 2004, Davey *et al*. 2010
European Goldfinch:	Siriwardena *et al*. 1999, Fuller *et al*. 2001, Báldi *et al*. 2005, Reino *et al*. 2009
Eurasian Hoopoe:	Fournier & Arlettaz 2001, Báldi *et al*. 2005, Barbaro *et al*. 2008, Arlettaz *et al*. 2010b
White Stork:	Carrascal *et al*. 1993, Johst *et al*. 2001, Nowakowski 2003, Latus & Kujawa 2005, Massemin-Challet *et al*. 2006, Rendón *et al*. 2008, Olsson & Rogers 2009

Another group strongly associated with semi-natural habitats are the shrikes. Giralt *et al*. (2008), for example, found that fledging success in Lesser Grey Shrikes was greater when territories included areas of shrub and fallow, because of high arthropod availability in these habitats, and similar relationships have been found for Red-backed Shrike (e.g. Golawski & Meissner 2008).

In the UK, summer densities of gamebirds, pigeons, Skylarks and other ground-foraging seed-eating passerines were higher in set-aside than in all other field types, and all of these groups preferred rotational to non-rotational set-aside, probably because swards of the former were less dense and uniform, giving greater foraging access (Henderson *et al*. 2000). In winter, species that showed a preference for set-aside included Grey Partridge, Jackdaw, Woodpigeon, Meadow Pipit, Goldfinch, Skylark, Linnet, Yellowhammer and Cirl Bunting, with most species showing stronger selection of first-year stubbles than fallows in their second year or older (Buckingham *et al*. 1999). Shortly before compulsory set-aside ended, Vickery *et al*. (2007) showed a weak, positive correlation between the area of set-aside and between-year changes in the UK FBI.

In other semi-natural habitats, Reed Buntings nesting in small-scale wetlands experienced lower nest predation and better chick survival than those on adjacent farmland, because denser vegetation gave better nest concealment, and invertebrates were more abundant (Brickle & Peach 2004, Surmacki 2004). Similarly, in a nest-box colonisation experiment, Tree Sparrow selected sites next to wetland habitats where they preferentially foraged for invertebrates when provisioning chicks (Field & Anderson 2004).

1.4.5. Field wetness

Sixteen species were associated with field wetness, split almost evenly between a preference for wet and dry habitats. Agricultural intensification can change field wetness in two ways. Drainage reduces soil moisture whereas irrigation has the opposite effect. Species that prefer moist soils include those that feed on soil-dwelling invertebrates such as Black-tailed Godwit and Lapwing, and breeding densities are higher in wet fields with patches of surface water (Kleijn & van Zuijlen 2004, Eglington *et al*. 2008). Soil-surface feeders such as Yellow Wagtail also favour damp soils with high penetrability and abundant invertebrate food (Gilroy *et al*. 2008). Drainage of wet grassland has contributed to historical declines in White Stork (Carrascal *et al*. 1993), and Nowakowski (2003) showed that pairs nesting closer to wet meadows had higher breeding success.

Species adapted to dry conditions include most of the larks, Tawny Pipit, Black-eared Wheatear and Stone Curlew. In their Iberian stronghold, irrigation has negatively affected these and other species (e.g. Great Bustard, Little Bustard, Montagu's Harrier and European Roller) by replacing dry cereal cultivation with intensively managed crops (Súarez *et al*. 1997, Brotons *et al*. 2004a, Pinto *et al*. 2005). However, some species have benefitted from new foraging habitats associated with irrigation, including Lesser Kestrel and Rook (Ursúa *et al*. 2005, Olea & Baglione 2008).

1.4.6. Area of scrub/woodland

Associations with the area of scrub or woodland were negative for 14 species and mixed for 26 species. For bush and tree-nesters such as finches, starlings, shrikes, pigeons, corvids and some of the buntings, patches of scrub and woodland provide nest sites. They also provide elevated song-posts for territorial species, and hunting perches for shrikes, chats and raptors. Trees and bushes also provide food for species that feed on buds and fruits (e.g. Greenfinch, Goldfinch), and invertebrate-rich foraging habitat for warblers, chats, and some of the buntings and shrikes. Several studies have shown the importance of larger patches of scrub or woodland (i.e. not including hedgerows, which in effect are linear patches of scrub) within breeding territories. These include studies of Turtle Dove (Browne *et al*. 2004), Hoopoe (Barbaro *et al*. 2008), Red-backed Shrike (Vanhinsbergh & Evans 2002), Lesser Grey Shrike (Kristin 1995), Woodchat Shrike (Isenmann & Fradet 1998) and Ortolan Bunting (Berg 2008, de Groot *et al*. 2010).

However, densities of most farmland species decline as the proportion of woodland in the landscape increases, because by definition farmland is their primary habitat. For example, pigeons and corvids readily nest in woodlands, but are heavily dependent on food resources and foraging habitats in farmland (Inglis *et al*. 1994b, Mason & Macdonald 2004). Using CBC data from farmland and woodland plots in England and Wales, Fuller *et al*. (2001) classified Whitethroat, Linnet, Goldfinch, Greenfinch and Yellowhammer as hedgerow specialists because they were strongly associated with farmland hedges and scrub, but scarce in woodlands, and predicted that populations would eventually decline if hedgerows were replaced with farm woodlands.

In other parts of Europe, steppe species such as Calandra Lark that require open fields with no shrubs or trees are highly sensitive to landscape fragmentation through afforestation or scrub encroachment into grasslands (Morgado *et al*. 2010), and this is a particular problem in marginal areas with widespread land abandonment (Laiolo & Tella 2006). For example, Fonderflick *et al*. (2010) showed that over 15–20 years, a shift towards intensive farming on productive land resulted in abandonment of extensive pastures, leading to scrub encroachment and declines in open-country species such as Greater Short-toed Lark, Tawny Pipit, Linnet and Ortolan Bunting. Following small-scale afforestations (shelterbelts 5–21 m wide and 380–1100 m long) in Polish farmland, Kujawa (2004) found that breeding densities of Yellow Wagtails, Skylarks and Whinchats declined, but Yellowhammers, Corn Buntings and Red-backed Shrikes increased.

1.4.7. Field size

Species relationships with field size generally reflect their associations with woodland and scrub. Field enlargement may benefit open-country ground-nesting species, whereas it will be detrimental to those that use non-cropped field boundary habitats such as hedgerows, scrub and herbaceous vegetation for foraging or nesting. In the UK, loss of hedgerows has been linked to declines in several boundary-nesting species, including Grey Partridge (Rands 1986), Turtle Dove (Browne *et al*. 2004), Common Whitethroat (Stoate & Szczur 2001) and Yellowhammer (Bradbury *et al*. 2000). Hedgerows are also important foraging habitats for a variety of species, including less obvious ones such as Swallow. Evans *et al*. (2003) showed that Swallows selectively foraged over field boundary features because aerial insects were more abundant than over fields, favouring hedgerows and trees especially during cold, wet and windy conditions.

In Switzerland, studies on Hoopoe demonstrate the effect that nest site limitation can have on breeding productivity. Fournier & Arlettaz (2001) found that some Hoopoes foraged up to 1 km from their nest site when provisioning chicks, because of a lack of nesting holes close to favoured foraging areas due to removal of hedgerows and tall trees. These birds had lower breeding success than ones nesting closer to foraging areas. Following the provision and colonisation of nest-boxes in foraging areas, over a ten year period the annual number of successful broods increased from around 20 per year to more than 100 (Arlettaz *et al*. 2010a). By contrast, removal of field boundaries can increase fecundity in some species. For example, Morris & Gilroy (2008) found that Skylarks and Yellow Wagtails nesting close to field boundaries suffered higher nest losses to mammalian predators than those nesting in field centres.

1.4.8. New crops

There were a few examples of bird relationships with new crop types associated with agricultural intensification worthy of note. The area of oilseed rape has increased massively in the UK and other parts of Europe, and several species have benefitted from this. The oil-rich seeds of rape provide a valuable source of food for Linnets during the breeding season, helping to buffer the effects of loss of weed seed food from other management intensifications (Moorcroft *et al*. 2006). Reed Buntings frequently nest in rape fields, which provide a seed-rich and invertebrate-rich foraging habitat and possibly better nest concealment from predators (Brickle & Peach 2004, Gruar *et al*. 2006), and Whitethroats may behave similarly (Mason & Macdonald 2000b). Finally, the population increase in Woodpigeon has been partly due to reduced overwinter mortality from starvation following the introduction of rape, whose leaves are now the preferred winter food for this species (Inglis *et al*. 1997). By contrast, Skylarks avoid rape fields during the breeding season because the tall dense swards restrict access to the ground (Wilson *et al*. 1997).

Combined with increased foraging on rubbish dumps, the expansion of rice cultivation around the Mediterranean has benefited White Storks by allowing an increasing proportion of the population to overwinter in southern Europe, reducing annual mortality (Barbraud *et al*. 1999, Rendón *et al*. 2008). However, although these 'resident' adults returned to their breeding sites earlier and laid bigger clutches, large broods had a higher mortality rate through chick starvation, so fledging success was no greater than in the later nesting migrants (Massemin-Challet *et al*. 2006).

Finally, the increasing trend for growing cereals for arable silage may benefit some species (e.g. Linnet, Tree Sparrow, Yellowhammer and Reed Bunting) by providing weed seed food in over-winter stubbles in landscapes otherwise dominated by seed-poor habitats such as intensive grassland or maize (Peach *et al*. 2011). However, it could also act as a trap for crop-nesting species such as Skylark, Yellow Wagtail and Corn Bunting because harvesting is early, before grains ripen, and there may be insufficient time for broods to fledge (Peach *et al*. 2011).

1.4.9. Summary

This review has demonstrated how complex agricultural intensification is, and how its various components affect species differently. For some species, the effects of certain aspects of land use change can vary across its annual lifecycle or European range. Whilst some aspects of intensification have had universal negative effects (e.g. loss of crop diversity and loss of semi-natural habitats), others are less consistent, and vary between species (e.g. aspects of grassland intensity such as mowing regimes and livestock grazing densities). Clearly, conservation solutions will vary between species, and between farming systems across Europe. Whilst the number of published studies referred to (which is not a comprehensive list!) shows how our knowledge base of farmland bird ecology has increased in recent years, these are unevenly distributed across Europe with a bias towards the north and west, and especially the UK (Tryjanowski *et al*. 2011). Further studies are needed in southern, central and eastern Europe, whose farmland bird populations have thus far not declined at the same rate as in northern and western Europe. With several central and eastern European countries recently joining the EU, perhaps the greatest current threat to farmland birds in Europe is the collapse of populations in these countries under the weight of CAP-driven intensification (Voříšek *et al*. 2010).

1.5. Measures to restore farmland biodiversity – agri-environment schemes

Currently, the main policy mechanism for attempting to halt and reverse farmland biodiversity losses across Europe is the ‘agri-environment scheme’, whereby farmers and landowners are paid to manage areas of their land for wildlife or other environmental objectives. Management options for farmland birds within these schemes have arisen mainly from the findings of studies reviewed in the previous section. Some of the reviewed studies were in fact experimental trials testing the efficacy of management solutions on demographic rates, whilst others involved monitoring population responses in the wider countryside following the deployment of agri-environment schemes. As knowledge of farmland bird ecology continues to improve, schemes are adapted to include new management options, and refinements made to existing options to increase their effectiveness.

1.5.1. A brief history of UK agri-environment schemes

Within the UK, nature conservation has traditionally had a site-based focus, involving the designation, protection and management of nature reserves and Sites of Special Scientific Interest (SSSIs). Only after the excesses of agricultural intensification had brought about the huge food surpluses of the 1980s that led to the introduction of set-aside (see section 1.1.1) was there any move towards wildlife conservation within the wider farmed landscape (Wilson *et al*. 2009). In 1987, the UK government introduced its first agri-environment scheme, the Environmentally Sensitive Area (ESA) scheme. For the first time, farmers were encouraged, and funded, to adopt measures for environmental protection and enhancement, and by 2000, ESAs covered approximately 10% of English and 20% of Scottish farmland (Wilson *et al*. 2009). Some considered this to be the dawn of a new era in agriculture – the agri-environment era whereby farmers are expected, and paid, to maintain production, but in ways that also deliver other 'public goods' such as environmental resource protection, attractive landscapes and biodiversity (Buckwell & Armstrong-Brown 2004).

EU-funded agri-environment schemes became available to all member states in 1992 with the 'MacSharry' reforms of the CAP (Buckwell & Armstrong-Brown 2004). In Britain, each country had its own scheme (England – Countryside Stewardship Scheme; Scotland – Countryside Premium Scheme; Wales – Tir Cymen). Unlike the ESAs, these schemes were not geographically restricted, although the focus was still on semi-natural habitats and landscape features, and they were not applicable to intensive arable farming systems (Wilson *et al*. 2009). Therefore, a pilot Arable Stewardship Scheme containing management options tailored to arable land was trialled in 1998–2000 in two English lowland regions – East Anglia and the West Midlands (Stevens & Bradbury 2006). Following the trial, successful arable options became available in the Countryside Stewardship Scheme and some ESAs. In 1999, further CAP reforms (the 'Agenda 2000' reforms) saw agri-environment schemes become compulsory for all EU member states, with increased funding made available by allowing countries to divert money into schemes from their production subsidy budget (Buckwell & Armstrong-Brown 2004). Soon afterwards, Scotland introduced a new scheme, the Rural Stewardship Scheme (which included several arable options), to replace its Countryside Premium Scheme (SEERAD 2003). In many countries, further decoupling of subsidies from production has since followed, such that funding for, and the area of land in agri-environment schemes, has never been higher than at present.

Further development of agri-environment schemes in Britain over the last decade has led to the current three-tiered pyramid approach (Wilson *et al*. 2009). To receive their main CAP subsidy payment (the 'Single Farm Payment'), farmers must adhere to certain standards that maintains their land in 'good agricultural and environmental condition' – this forms the base of the pyramid. The middle tier is a group of simple, inexpensive entry-level agri-environment measures available to all (England – Entry Level Stewardship; Scotland – Land Managers' Options; Wales – Tir Cynnal). Finally, agri-environment measures in the top tier are more costly and targeted, and only available by competitive application (England – Higher Level Scheme; Scotland – Rural Priorities; Wales – Tir Gofal).

1.5.2. Effectiveness of agri-environment schemes

Evaluations of the effectiveness of schemes to halt and reverse declines of farmland birds have yielded mixed results. In the UK for example, only two of the species (Reed Bunting, Corn Bunting) whose trends contribute to the FBI showed a positive population response to measures implemented over a five-year period through the Arable Stewardship Pilot Scheme (Stevens & Bradbury 2006). In ESA schemes, only on those sites deploying the most expensive options were breeding wader declines halted (Wilson A. *et al*. 2007). Ultimately, the FBI continues to fall, albeit at a slower rate than during the pre agri-environment era. Furthermore, although billions have been spent on agri-environment schemes across Europe during the last three decades, a recent review of studies measuring their effectiveness found relatively few that demonstrated clear biodiversity benefits, and worse still, many schemes lacked robust monitoring to make such assessments possible (Kleijn & Sutherland 2003).

Since Kleijn and Sutherland's review, however, a number of published studies have demonstrated benefits for a wide range of taxa (e.g. Kleijn *et al*. 2006, Knop *et al*. 2006, Pywell *et al*. 2006, Maes *et al*. 2008). A small number of UK studies have also demonstrated considerable success in reversing farmland bird declines (Aebischer *et al*. 2000, Peach *et al*. 2001, O'Brien *et al*. 2006). These schemes were successful because they each targeted an individual species with key resources identified from previous research (Stone Curlew – safe-nesting fallow plots on arable land and restoration of grazing on semi-natural grasslands; Cirl Bunting – provision of rough grassland, hedgerow/scrub management and low-input spring cereals with overwinter stubbles; Corncrake – provision of early-season cover and delayed mowing of hay meadows).

Key ingredients for successful agri-environment schemes are, therefore, a thorough understanding of the mechanisms driving a species' decline (see section 1.4), and effective design and targeting of conservation solutions. The remainder of this thesis focuses on one species – the Corn Bunting – with the main aim of testing and improving management solutions to design an agri-environment scheme capable of halting and reversing the species' population decline.

1.6. Corn Buntings in eastern Scotland – a case study

The Corn Bunting has undergone one of the largest population declines among UK and European farmland birds and is consequently of high conservation concern (Tables 1.1 and 1.2). Across most of Europe, it is a true farmland specialist, and traits such as a preference for nesting in crops combined with a late breeding season make it particularly sensitive to agricultural intensification (Tables 1.3 and 1.5). Therefore, the Corn Bunting makes a good case study for testing conservation solutions designed to improve the effectiveness of agri-environment schemes.

Conducting the study in eastern Scotland is also valuable because Scottish studies are currently under-represented in the UK literature on farmland birds, especially in lowland arable farming systems. This is despite Scotland holding a significant proportion of the UK populations of several red and amber-listed species typical of arable farmland (e.g. Skylark 31%, Reed Bunting 24%, Yellowhammer 18%, Linnet 16%, Grey Partridge 14%, Common Whitethroat 14%, Tree Sparrow 12%, Corn Bunting 11% – Forrester *et al*. 2007). For example, a search of the 'Web of Science' database (accessed 19 July 2012) using the topic keywords "birds" and "farmland" and "scotland" revealed just 34 publications, compared with 329 for "birds" and "farmland" and "england".

Given that recent population change has been more positive in Scotland than England in all five UK FBI species (Skylark, Reed Bunting, Yellowhammer, Linnet and Common Whitethroat) with separate English and Scottish BBS trends (Table 1.1), farming intensification, or at least its effects on bird populations, so far appears to have been less severe in Scotland than in England. Indeed, lowland farmland in northeast Scotland is characterised by mixed arable-livestock systems in which most cereals are spring-sown and grasslands managed less intensively than in other parts of the UK. Therefore, as well as informing conservation solutions to help maintain populations locally, studying Corn Buntings in the mixed farming systems of eastern Scotland will also complement previous UK studies from different farming systems, and give additional insights into why this species has declined to extinction across large parts of the UK and Europe.

CHAPTER 2. STUDY AIMS, SPECIES AND AREA – CORN BUNTINGS IN EASTERN SCOTLAND

2.1. Background

One of the largest declines among UK and European farmland birds has been of the Corn Bunting, and consequently it is a species of high conservation concern (see Tables 1.1 & 1.2). In Scotland in 2001, it became the focus of a new conservation project by the Royal Society for the Protection of Birds (RSPB) called Farmland Bird Lifeline (FBL), which funds farm management interventions targeted at the declining Corn Bunting population on arable and mixed-farmland in eastern Scotland. Shortly afterwards in 2002, another RSPB project (led by myself) began, with the aim of assessing the effectiveness of Scotland's then new agri-environment scheme, the Rural Stewardship Scheme (RSS). Again, the focal species was the Corn Bunting, and study area was eastern Scotland.

There was already a history of work on the Corn Bunting in this region with the studies of Dr Adam Watson (AW) in Aberdeenshire and Angus, and recent annual population monitoring by Prof Chris Smout and colleagues in Fife (Watson & Rae 1997b; Elkins *et al*. 2003). It is from the two RSPB projects, and AW's studies that this PhD study was developed.

2.2. Aims of project

Building on the work undertaken by myself and colleagues for the RSPB, and using a unique long-term dataset of Corn Buntings and associated habitat variables collected by AW from 1989–2008 which had not previously been analysed, the aims of this study were to understand the causes of Corn Bunting population declines in eastern Scotland and identify management solutions. The overall aim was to develop a package of management options that will enable effective targeting of conservation in Scotland for this vulnerable species, and allow population recovery through the deployment of those options within agri-environment schemes and wider conservation management.

2.3. Outline of chapters

Chapter 1 set the context for this thesis by describing the changes that have taken place in agriculture over the past few decades, the effects these have had on bird populations throughout Europe, and the development of agri-environment schemes as the main policy response to halting and reversing species declines. Here in *Chapter 2*, I give a brief overview of Corn Bunting ecology, the size and trends of UK and European populations, and using AW's long-term dataset, determine the recent trends of populations in eastern Scotland. First, here is an outline of the studies presented in Chapters 3 – 7, including the hypotheses tested and rationale for asking this particular set of questions:

Chapter 3 – Habitat Associations with Territory Distribution and Mating System

In this chapter, I use AW's 20-year dataset from his largest study area (36 km^2, group 5 in Table 2.2) to show the habitat attributes of Corn Bunting breeding territories. For this, three predictions were tested – habitat associations would vary (1) over time as the size of the population changed; (2) seasonally according to changes in availability and quality of nesting habitat (because Corn Buntings are multi-brooded – see section 2.4); (3) according to the mating status of the male (because Corn Buntings are polygynous – see section 2.4). The aim of these analyses was to determine what makes a high-quality Corn Bunting territory in the mixed farming landscape of eastern Scotland, and to identify likely causes of population declines within this region and potential conservation solutions.

Chapter 4 – Habitat Selection by Females for Nesting

Here I determine the nesting habitat preferences of female Corn Buntings in relation to crop type and sward structure, to confirm some of the hypotheses given in Chapter 3 as explanations of seasonal habitat associations of territorial males. I use a 6-year RSPB dataset across 32 study farms in four areas of eastern Scotland to answer four questions: (1) in eastern Scotland, when and where do Corn Buntings nest? (2) How does crop use for nesting vary seasonally? (3) Which sward characteristics best explain field use by nesting Corn Buntings? (4) Can changes in sward structure explain the seasonal pattern of crop use for nesting? I compare my findings with those from other UK regions and farming systems, and discuss the implications of the types of crops grown and intensity of their management on the timing and length of the Corn Bunting breeding season and annual number of broods reared.

Chapter 5 – Nest Success and Trials of Delayed Mowing to Increase Fledging in Meadows

In this chapter, I measure the fledging success of Corn Bunting nests in each of the crop types identified as nesting habitats in Chapter 4. Data are from 21 of the study farms used in

Chapter 4 (across 2 areas and 5 years). I determine that Corn Buntings nesting in grass silage or hay fields suffer high rates of nest loss during mowing, so I also test the effectiveness of trial conservation interventions designed to increase nest success in meadows by delaying mowing until after broods had fledged. I conduct scenario tests with various mowing dates and scales of deployment, and assess the overall effect of delayed mowing on annual reproductive success at the population scale and potential population trend effects. I also discuss the viability of delayed mowing as an agri-environment scheme option from a farming perspective, and possible alternative management solutions.

Chapter 6 – Winter Habitat Use by Corn Buntings and Other Seed-eating Birds

The previous three chapters focus on Corn Buntings during the breeding season, but reduced food supply during winter is a major contributing factor to population declines of farmland birds (see Chapter 1). Here I test the effectiveness of an agri-environment scheme option ("unharvested crops") designed to provide over-winter seed food for birds by measuring use of these crops by ten seed-eating species, including Corn Bunting, relative to other seed-rich habitats. Data are from 53 RSPB study farms in eastern Scotland over three winters. To determine the best management of unharvested crops for birds, I also compare patch use between one-year and two-year old crops, and between early and late winter, and relate the pattern of bird use to changes in seed availability. I make several conservation recommendations for improving the effectiveness of unharvested crops, and discuss the farming practicality of this management option, bird use of other seed-rich habitats, and potential population effects of over-winter seed provision through agri-environment schemes.

Chapter 7 – Population Response to Agri-environment Schemes

In this final data chapter, I use population monitoring data over seven years and 71 RSPB study farms in four areas of eastern Scotland to measure the breeding population response of Corn Buntings to agri-environment management, including the options tested in Chapters 5 and 6. I compare population trends between three groups of farms: (1) scheme targeted specifically at Corn Buntings and subjected to adaptive management; (2) general scheme not targeted at Corn Buntings; (3) controls with no agri-environment management. For groups (2) and (3), population responses of Yellowhammer and Reed Bunting are also measured. I also use the observed annual growth rates of Corn Bunting populations on farms in each group to estimate the proportion of the total population that agri-environment management will need to target to halt and reverse the national decline. Discussion includes the need for effective targeting, greater uptake of schemes and within-field management options, and the importance of monitoring and adaptive management to improve existing options and introduce new ones.

Finally, Chapter 8 concludes the thesis with a general discussion of the overall findings of the study, conservation recommendations for the Corn Bunting in Scotland and elsewhere, and suggestions for improved design and implementation of agri-environment schemes to maximise their effectiveness.

2.4. The study species – Corn Bunting

2.4.1. Size, plumage and song

The Corn Bunting belongs to the *Emberizidae* family, and with a male wing-length of 96–107 mm and body-mass of 43–63 g, it is the largest species within this group (Cramp & Perrins 1994). The plumage is unremarkable, described by Snow and Perrins (1998) as "heavily streaked buff-brown", and the male and female generally look alike (Plates 1–4). However, during the breeding season, the male often shows a larger 'bib' where the breast streaks coalesce (although this may vary with the bird's posture), and paler 'bleached' upperparts (especially the tail) due to prolonged exposure to sunlight from frequent use of high perches and song-posts throughout the summer. The song is a rapid, jangling trill with a stuttering start, likened to a rattling bunch of keys (Snow & Perrins 1998). The male sings frequently and conspicuously during all daylight hours (including midday, unlike most other passerines), and song is used for both attracting females and territorial defence against rivals (Møller 1983, Olinkiewicz & Osiejuk 2003). Corn Bunting songs can vary between individuals and populations, and local 'dialects' have been the focus of several studies (e.g. McGregor 1980, 1986, McGregor & Thompson 1988, McGregor *et al*. 1997).

2.4.2. Distribution, habitat and movements

The Corn Bunting occupies open, lowland landscapes throughout much of Europe, and generally avoids woodlands, wetlands and mountainous areas. The species' range extends from North Africa in the southwest to Afghanistan and western China in the east, and occurs mostly between latitudes 35ºN and 60ºN (Snow & Perrins 1998). In central and western Europe, including the British Isles, it is a true farmland specialist, favouring arable and mixed farmland habitats. In other parts of the species' range, such as the Iberian Peninsula, it also occupies habitats such as uncultivated grasslands and scrublands. In central and northern Europe, the Corn Bunting is at least partially migratory and some populations in northeast Europe are wholly migratory, and move southwest to winter in Iberia and North Africa. However, across much of the species' range, including the British Isles, it is non-migratory, undergoing only local movements, mainly as roaming flocks during the winter with birds returning to natal areas the following summer (Wernham *et al*. 2002).

2.4.3. Diet, foraging, breeding and habitat associations

Details of Corn Bunting diet (Chapters 3 and 6), breeding habitat associations (Chapters 3 and 4), nest success (Chapter 5), and winter habitat associations (Chapter 6) are given elsewhere in this thesis, but I also give a brief account here.

Corn Bunting diet is predominantly cereal grain and seeds of grasses and arable weeds, supplemented in summer by invertebrates such as caterpillars, grasshoppers, sawfly larvae, beetles, spiders, harvestmen, moths, craneflies, hoverflies and other flies fed to chicks (Watson 1992, Hartley & Quicke 1994, Brickle & Harper 1999). All of these items are foraged mainly from the ground or in low vegetation (Plate 3). In winter, cereal stubbles and fallows are favoured, especially those with abundant weed seed, whilst winter cereals and improved grasslands are avoided (e.g. Donald & Evans 1994, Brickle & Harper 2000, Stoate *et al*. 2000, Orlowski 2006). The species also exploits cereal grains provided in feed troughs for livestock, as well as those lying on the soil surface in newly sown fields (Brickle & Harper 2000). Communal roosts, sometimes involving hundreds of Corn Buntings, form outside of the breeding season, with marked individuals recorded roosting up to 4 km from their feeding site (Harper 1995). These roosts occur in a variety of habitats, including reed beds, scrub, conifer trees, stacks of straw bales, and even on the ground within stubbles or on saltmarsh (Harper 1995).

Territorial behaviour by the male Corn Bunting often begins during winter, with song given on fine, clear days, but most territories become occupied by males from the end of March, and females from late April (Møller 1983, Harper 1995). Elevated song-posts with a clear view of the surroundings are favoured (Plates 1–2), and the male typically defends a territorial area of approximately 2–6 ha against other males (Møller 1983, Hartley *et al*. 1995, Brickle *et al*. 2000). In favoured areas, Corn Bunting breeding density can be high, with 24 territories km^{-1} recorded in a recent Danish study (Lilleør 2007) and 15 territories km^{-1} in England (Mason & Macdonald 2000a). In southern Europe, densities up to 140 birds km^{-1} occur (Diaz & Tellaria 1997).

In Britain, the onset of breeding is usually in late May or early June, with egg-laying continuing until early August (often including second clutches) where suitable nesting habitat is available (e.g. Ryves & Ryves 1934, Macdonald 1965, Hartley 1991, Yom-Tov 1992a, Brickle 1998, Brickle & Harper 2002). The nest is usually in dense vegetation on or close to the ground, within growing crops or tall grasses (Plates 7–8). Overall, most nests are in cereals, although hay fields and other grass habitats such as set-aside are also used, and on rare occasions low bushes or shrubby vegetation (e.g. Gillings & Watts 1997, Brickle & Harper 2002, Setchfield *et al*. 2012). In the Western Isles, most nests are in dune grassland, probably due to a lack of early-summer cover in spring-sown cereals (Hartley *et al*. 1995).

The female alone builds the nest, a loosely constructed cup made from coarse grass stems, lined with finer grasses and hair (Harper 1995). Eggs are laid at a rate of one per day, and clutch size is usually between three and five, rarely fewer or greater, but clutches of seven do occur (Ryves & Ryves 1934, Macdonald 1965, Hartley 1991, Harper 1995, Crick 1997, Brickle 1998). Incubation lasts for 12–14 days, beginning on the laying day of the final egg, and chicks remain in the nest for 9–13 days after hatching, often leaving before they can fly (Snow and Perrins 1998). At 15 days old, chicks are capable of flight, but the parents feed them for up to 20 days further (Harper 1995, A. Watson pers. comm.). The female will rear a second brood where suitable nesting habitat is available, but this is now rare across much of the UK (Hartley & Shepherd 1994b, Brickle & Harper 2002).

2.4.4. Some unusual traits

The Corn Bunting has a number of traits unusual amongst European passerines. First, compared with most passerines, sexual size dimorphism in the Corn Bunting is large, with the male typically 20–30% heavier and 10% longer in the wing than the female (Harper 1995). Second, unlike most passerines, the Corn Bunting has a complete post-juvenile moult (Harper 1995). In this respect, the species is more similar to sparrows and larks than to other buntings, and was one of the reasons for its former placement in a separate genus, *Miliaria*. Third, the breeding season in Britain is later than in many other related species, generally starting no earlier than May but extending into August or even September (Crick *et al*. 1994). Fourth, the mating system is complex, with some individuals displaying polygyny. Approximately a quarter of males are polygynous, typically with two to three females per male (Cramp & Perrins 1994), although the frequency and extent of polygyny varies considerably between areas. Using colour-rings to mark and identify individual birds, Harper (1995) recorded up to six females nesting simultaneously in one male territory. Paring polygynously may not be costly to females because male Corn Buntings make little contribution to feeding broods (Hartley 1991, Brickle & Harper 1999), and it may be more important for females to select territories with superior nesting or foraging habitats than males that are unmated (see Chapter 3).

2.5. Corn Bunting population trends

2.5.1. Europe

The European population estimate is between 4.8 and 12.7 million pairs, excluding the large Turkish population of 3–9 million pairs (BirdLife International 2004). Across Europe, the Corn Bunting has declined at an average of 3.5% p.a. since 1980 (PECBMS 2011), with declines reported in 22 of the 34 countries where data were available for a review (Hagemeijer & Blair 1997). Consequently, the species has an ‘unfavourable’ conservation status (concentrated in Europe with a declining population – BirdLife International 2004). Densities are highest in low-intensity mixed farmland (Donald & Aebischer 1997, Fox & Heldbjerg 2008), and the largest declines are in northern and western Europe where agriculture has intensified and specialised most (BirdLife International 2004).

2.5.2. UK

The most recent UK population estimate was 8500–12200 territories in 2000 (Baker *et al*. 2006). At the start of the twentieth century, the Corn Bunting occurred throughout the British Isles, wherever farmers grew cereals, and described as abundant though patchily distributed (Donald *et al*. 1994, Holloway 1996). Since then, large declines in the 1920s–1930s and from the 1970s (Baillie *et al*. 2010) led to extinction in Ireland (Taylor & O’Halloran 2002) and an end to regular breeding in Wales. In England and Scotland, the species’ range has also contracted, as shown by the changes in mapped distribution between the two national breeding bird atlases (Sharrock 1976, Gibbons *et al*. 1993). The number of occupied 10-km squares declined by 32%, from 1358 in 1968–72 to 921 in 1988–91, the largest range contraction of any UK FBI species (see Table 1.1). Most local extinctions in the north and west (e.g. Shetland 1978, Coll 1984, Lewis & Harris 1992, Tiree and Orkney late 1990s) were associated with an end to widespread cereal growing in those areas (Robinson *et al*.

2001, Forrester *et al*. 2007). In Scotland, the Corn Bunting is now confined almost entirely to two areas, the east coast lowlands from Fife to Inverness, and the Western Isles (Fig. 2.1).

Based on changes in 68 CBC/BBS study plots with annual monitoring from 1967 to 2008, the UK Corn Bunting population declined by an estimated 86% (lower confidence limit -94%, upper confidence limit -75%) during this period (Baillie *et al*. 2010). Between 1983 and 2008, the estimated decline was 78% (lcl -90%, ucl -62%) across 91 plots. Consequently, Corn Bunting is on the 'red list' of Birds of Conservation Concern and is a priority species in the UK Biodiversity Action Plan (Eaton *et al*. 2009). In recent years, the decline appears to have slowed, with numbers falling by 12% (lcl -28%, ucl +6%) between 1998 and 2008 across 138 1-km squares in the BBS, and in 2003–2008 increasing by 2% (lcl -16%, ucl +25%) across 140 1-km squares. However, population trend data are heavily biased towards England, because in Scotland there are too few BBS squares with Corn Buntings (n = 2–9 per year in 1994–2010) to derive an index of change in breeding numbers from national monitoring data (Risely *et al*. 2011). Local surveys suggest continuing declines. For example, in the Western Isles, repeat surveys revealed a population decline of 62% between 1995 and 2005, and 17% between 2002/3 and 2005 (Wilson, J.D. *et al*. 2007), and in Fife numbers fell by 38% between 1995 and 2002 (Elkins *et al*. 2003). Overall, the Scottish population is probably now as few as 800–900 territorial males (Table 2.1), compared with an estimated 2200 territories in 1993 (Donald & Evans 1995, Forrester *et al*. 2007).

2.5.3. Eastern Scotland

Partly because of the lack of a BBS trend for the Corn Bunting in Scotland, we[1] used AW's data to measure population change across 30 study areas (Fig. 2.2) in eastern Scotland during 1989–2007 and published the results (Watson *et al*. 2009). Descriptions of the study areas and field methods used by AW are in the paper (Appendix 1). Here, I present the methods that we used to analyse the data, along with the results followed by a short discussion.

[1] Several colleagues helped with this study. Fieldwork was designed and undertaken by Adam Watson, assisted in earlier years by Mick Marquiss, Robert Rae, Stuart Rae and Andrew Stalker, and in later years by Amanda Biggins, Alan Bull, John McMahon and Hywel Maggs. With advice from Jeremy Wilson, I designed and carried out all statistical analyses, and in collaboration with Adam Watson, the lead author of the published paper (Appendix 1), wrote the first draft and incorporated improvements suggested by the co-authors (Jeremy Wilson and Hywel Maggs), and the journal editor and an anonymous referee.

2.5.3.1. Data analysis

During the early years of the study, some Corn Bunting populations appeared to be transient, whilst other areas held birds in every year. Therefore, we divided study areas into two categories. Sixteen areas with populations in all summers during 1989–95 were called ‘groups’, and 14 areas where birds did not occur in spring 1989 or in all summers of 1989–95 were termed ‘offshoots’. Offshoots appeared in late May or the start of June, and birds then bred, not having been seen earlier in that spring or the previous winter.

To determine changes in population between years, we modelled the number of males in each of the 16 groups as a function of the *year* (fixed effect), *group* (random effect), and *log size* (km^2) of study area (offset). Effectively, therefore, we compared the density of males per group between years. For this, we used a generalised linear mixed model (GLMM) with a log-link function, and assumed a Poisson error distribution, with the standard errors adjusted for over-dispersion. First, we fitted *year* as a categorical variable (model 1), to obtain individual year means and standard errors for the density of males per group. The next step was to fit *year* as a covariate (model 2), from which the back-transformed regression coefficient gave an estimate of the annual percent rate of change in the number of males across all groups over the whole period. These analyses used the GLIMMIX procedure of SAS version 9.1. To calculate the denominator degrees of freedom for tests of fixed effects[2], we used the Kenward-Roger method (Littell *et al*. 1996).

Finally, we modelled the number of new offshoots appearing in year ^{t}, and the number of males in them, as a function of overall change in male numbers between year $^{t-1}$ and year ^{t}, using a generalised linear model (GLM) with a log-link function and Poisson errors, and correcting for over-dispersion. For this, we used the GENMOD procedure of SAS version 9.1.

2.5.3.2. Results

Between 1989 and 2007, the number of males decreased in 15 groups and increased in one (binomial test, $P < 0.001$). These declines went to extinction in 12 groups, such that only

[2] We used Wald t-tests for fixed effects in GLMMs, which test a null hypothesis of no effect by dividing the parameter estimate by its standard error (the t-value), and comparing this test statistic to zero. SAS estimates a two-tailed p-value corresponding to the t-value and associated degrees of freedom to determine whether one can reject the null hypothesis. This method applies to hypothesis testing of GLMMs throughout the thesis.

four areas still held birds in 2006 and 2007. Secondly, the annual total number of males in all areas combined ranged from 337 in 1990 to 36 in 2006. Counts were strongly correlated with the number of areas occupied across years (n = 19 years, r_s = 0.944, P < 0.0001). The maximum number of males in a group varied from five to 134, and their density from 0.8 to 8.6 per km ^{2} (Table 2.2). Densities were significantly (P < 0.0001) lower than in 1989 in each year during 1996–2007 inclusive. Overall, the number of males fell by 83% between 1989 and 2007 (t_{302} = –28.62, P < 0.0001), a mean annual rate of decline of 10.3% ± 0.33 (1 se) (Fig. 2.3).

The decline, however, was far from constant. Between 1989 and 1990, the number of areas occupied (groups and offshoots) actually increased, from 22 to 25, and in 1989–95 the annual total of males across all areas fluctuated without any trend. Falls of 49% (1995–1996), 34% (1998–1999) and 45% (2003–2004) predominate in accounting for the overall decline (Table 2.2, Fig. 2.3). In these years, declines tended to be consistent across all study areas. All 16 groups declined between 1995 and 1996, 12 out of 13 between 1998 and 1999, and seven out of eight between 2003 and 2004. Six of the 12 extinctions of groups occurred within just these three pairs of years.

The total number of known offshoots varied from year to year, owing to new ones being founded and to older ones becoming extinct. Fourteen areas held offshoots in one or more years. At two of them (areas 19 and 25), birds soon became extinct, but on each of these a second new offshoot formed in a later year. Hence, there were 16 cases of new offshoots. These occurred on land near seven groups. The distance between an offshoot and its nearest group ranged from 1–10 km, with a mean of 2.7 km and a median of 1.5 km.

The initial founding number in a new offshoot varied from one to nine males (Fig. 2.4), but all became extinct later, and in six cases an offshoot occurred for only one summer. The last new offshoot seen was in summer 1995. That summer preceded the largest annual fall in numbers in the groups, during 1995–96. Overall, the absolute change in the number of males on all groups combined between year $^{t-1}$ and year ^{t} was positively correlated with the number of new offshoots in year ^{t} (χ^2_1 = 5.14, P = 0.023), but not with the number of males in those offshoots.

2.5.3.3. Discussion

On all 30 study areas combined, the number of males declined by 83% between 1989 and 2007, and birds became extinct on all but four areas. The mean annual rate of decline was 10%, but population change varied greatly between years, and three large year-to-year declines accounted for most of the overall decline. Further analysis is needed to determine which demographic or environmental factors coincided with these three declines, and with the many other fluctuations on individual areas. One encouraging result, however was that the data on offshoots indicate that populations should have the capacity to spread rapidly in response to improvement of farmland habitats for this species through well-targeted agri-environment schemes.

Other east-Scottish studies have also revealed declines in Corn Buntings since the 1980s. Repeat surveys in 1997–99 of 10-km squares covered by the national wintering bird atlas in 1981–84 (Lack 1986) showed a 62% decline in winter counts (Hancock *et al*. 2009). The breeding distribution in Fife contracted from 23 occupied 10-km squares in 1968–72 to 14 in 1988–91 and just nine in 2000, and in Angus from 13 occupied 10-km squares in 1988–91 to just seven in 2002 (Gibbons *et al*. 1993, Elkins *et al*. 2003, RSPB unpubl. data). Data for the regional bird atlas in northeast Scotland also showed a 26% contraction in breeding distribution between 1981–84 and 2002–06, and a 34% decline in the number of occupied 10-km squares since 1968–72 (Buckland *et al*. 1990; Francis & Cook 2011). Because of this range contraction, and the extinctions of most populations in the present study, AW's four study areas that still held birds in 2007 lay far apart and were greatly isolated compared with 1989 and 1990.

The declines followed a rise of more intensive farming in northeast Scotland from the mid-1970s to the mid-1980s. Several agricultural changes occurred during this period and have continued since, and hence are associated with falling numbers of Corn Buntings. The northeast is one of the main Scottish regions for rearing beef cattle and pigs, and for growing cereals and oilseed rape, with Aberdeenshire accounting for 27% and 35% of the Scotland's total area of these crops, respectively (Cook 2008). Consequently, lowland Aberdeenshire is one of the most mixed farming areas in the UK, with an approximately equal split between grass and arable (Plates 10–13). Half of Aberdeenshire's grass area is mown, and livestock grazing densities are double the national average (Cook 2008). In meadows, earlier mowing of grass for silage has largely replaced late mowing for hay, whilst in arable crops such as cereals, changes include increased herbicide use, the removal of boundaries to make bigger fields, and more stubble fields cultivated in autumn or ploughed in early winter (Cook 2008, Francis & Cook 2011). Across all of Scotland, the area of wheat, a predominantly autumn-sown cereal, almost trebled between 1970 and 1990, from 40 000 ha to 110 000 ha, and at the same time the area of oats, a mainly spring-sown cereal, declined by three-quarters from 125 000 ha to just 30 000 ha (DEFRA 2012). The area of barley, however, remained relatively stable at around 300 000 ha, more than 80% of which is still spring-sown in Scotland, compared with 45% in England (DEFRA 2012, Scottish Government 2010c), and spring barley is likely to remain the dominant cereal in Aberdeenshire to supply an expanding malting industry (Cook 2008). Other changes include the loss of corn-ricks following the advent of combine harvesters (the last ricks on the study areas were at area 5 in winter 1988–89), and fewer farmers keeping animals, with consequent declines in over-winter stubble and fodder crops such as turnips.

In subsequent chapters, I determine the effects of land use changes on Corn Buntings in eastern Scotland, but studies elsewhere have identified most of the above as potential drivers of declines in the UK and Europe. Specifically, the switch from spring to autumn sowing is likely to have reduced annual survival rates through loss of winter seed food (Donald & Evans 1994, Mason & Macdonald 2000a, Wilson, J. *et al*. 2007), and reduced the incidence of double-broods through earlier harvesting of autumn-sown cereals (Brickle & Harper 2002). Increased pesticide use is likely to have lowered chick survival by reducing the availability of invertebrate food (Brickle *et al*. 2000) and reduced the incidence of double-broods (Setchfield *et al*. 2012). In Europe, population declines have been associated with a shift from low-intensity mixed farming with spring-sown cereals to either specialised intensive production with autumn-sown crops, or the abandonment of cereal cultivation (Donald & Aebischer 1997, Stoate *et al*. 2000, Taylor & O'Halloran 2002, Lilleør 2007, Fox & Heldbjerg 2008, Brambilla *et al*. 2009).

In the next chapter, I test several predictions to demonstrate relationships between land use and territory location, late-summer occupancy and polygyny, and to determine the likely causes of the population decline that occurred in AW's largest study area (group 5).

Table 2.1. *Estimated size of the Corn Bunting population in each Scottish region currently known occupied, and year most recently surveyed.*

Region	**Territorial males**	**Year**	**Source**
Borders	2	2011	RSPB unpubl. data
Fife	101	2011	T.C. Smout unpubl. data
Angus	47	2002	RSPB unpubl. data
Aberdeenshire & Moray	550–600	2002–2006	Francis & Cook 2011
Inverness-shire	16	2010	RSPB unpubl. data
Western Isles (Berneray–Vatersay)	117	2005	Wilson *et al*. 2007b
Total	**833–883**		

Table 2.2. *Number of territorial male Corn Buntings in each study area, June 1989–2007.* ***0*** *= extinction, first spring when no birds seen.*

a) *Groups, birds seen in and before spring 1989.*

Area	Size (km^2)	89	90	91	92	93	94	95	96	97	98	99	00	01	02	03	04	05	06	07
1[a]	19.0	21	28	31	19	37	25	40	18	25	31	21	18	31	33	17	17	16	21	30
2	14.2	19	18	18	16	33	18	32	12	19	24	19	8	12	12	9	4	3	2	2
3	6.6	4	4	5	6	7	5	11	5	7	8	4	2	3	4	1	**0**	0	0	0
4	9.0	6	6	5	7	6	5	5	1	2	2	**0**	0	0	0	0	0	0	0	0
5[b]	36.4	102	134	99	82	64	73	82	43	60	38	29	30	19	26	37	19	19	10	12
6	2.1	5	7	4	4	3	4	2	1	1	1	**0**	0	0	0	0	0	0	0	0
7	7.2	17	18	14	13	15	13	10	8	8	8	6	7	3	3	5	1	1	**0**	0
8	2.1	4	4	5	5	4	4	1	**0**	0	0	0	0	0	0	0	0	0	0	0
9	2.0	6	7	6	6	8	8	5	2	1	**0**	0	0	0	0	0	0	0	0	0
10	5.0	11	11	8	9	6	7	9	6	6	2	**0**	0	0	0	0	0	0	0	0
11	9.4	24	24	24	23	26	21	30	14	9	9	4	7	8	7	9	1	1	**0**	0
12	3.9	18	19	17	13	9	9	8	6	6	1	1	1	1	1	**0**	0	0	0	0
13	7.9	21	28	19	22	26	19	23	17	20	23	14	25	24	14	15	9	4	3	4
14	10.0	8	4	4	8	8	4	4	3	4	**0**	0	0	0	0	0	0	0	0	0
21	4.3	2	3	2	2	5	5	8	3	6	4	2	2	2	3	2	1	1	**0**	0
24	0.7	1	1	1	2	6	6	2	1	1	1	**0**	0	0	0	0	0	0	0	0
Total		269	316	262	237	263	226	272	140	175	152	100	100	103	103	95	52	45	36	48

[a] *agri-environment measures implemented on three farms since 2003*
[b] *agri-environment measures implemented on two farms since 2002 and another farm since 2006*

b) *Offshoots, birds not seen in and before spring 1989.*

Area	Size (km^2)	89	90	91	92	93	94	95	96	97	98	99	00	01	02	03	04	05	06	07
15	0.9	2	1	1	1	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0
16	0.2	0	1	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
17	0.2	0	0	0	0	1	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0
18	1.5	0	6	5	3	1	1	2	1	**0**	0	0	0	0	0	0	0	0	0	0
19	2.0	1	2	**0**	9	4	2	2	2	1	2	**0**	0	0	0	0	0	0	0	0
20	0.7	0	0	1	1	1	2	1	1	1	1	1	**0**	0	0	0	0	0	0	0
22	0.3	1	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
23	1.1	1	3	2	1	2	2	**0**	0	0	0	0	0	0	0	0	0	0	0	0
25	0.3	0	1	**0**	0	0	1	**0**	0	0	0	0	0	0	0	0	0	0	0	0
26	1.1	0	3	2	2	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0	0
27	1.0	1	1	2	2	1	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0
28	1.1	1	3	3	1	1	**0**	0	0	0	0	0	0	0	0	0	0	0	0	0
29	0.2	0	0	0	0	0	0	1	**0**	0	0	0	0	0	0	0	0	0	0	0
30	0.6	0	0	0	0	0	4	3	1	1	**0**	0	0	0	0	0	0	0	0	0
Total		7	21	16	20	11	12	9	5	3	3	1	0	0	0	0	0	0	0	0

Figure 2.1. *Map showing Corn Bunting distribution (occupied 10-km squares) across Scotland during three time-periods: pale grey = 1968–72 but not 1988–91 or 2002–06; dark grey = 1988–91 but not 2002–06; black = 2002–06. Most black and dark grey squares were also occupied during the previous time-periods. Exceptions are 17 dark grey squares apparently unoccupied in 1968–72 (10 northwest, 3 southwest and 4 eastern Scotland), and 15 black squares apparently unoccupied in 1988–91 (all in northeast Scotland, although 1988–91 survey coverage was poor in this region, and Corn Buntings were present in at least three of these 'unoccupied' squares – A. Watson pers. comm.). Sources: Gibbons et al. (1993), Francis & Cook (2011), T.C. Smout unpubl. data, RSPB unpubl. data.*

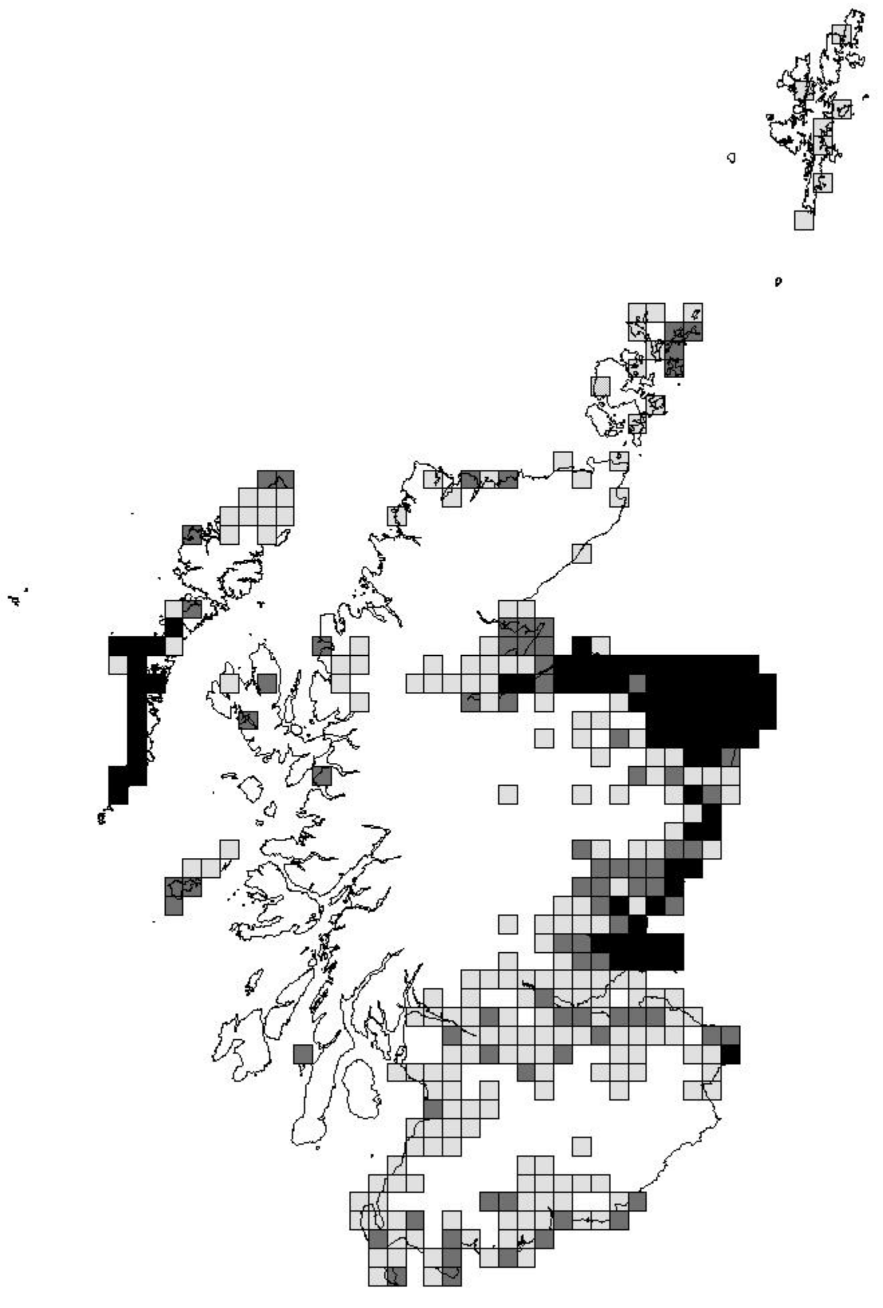


Figure 2.2. *Map showing location of AW's 30 study areas from 1989–2007 (16 groups in bold), and current Corn Bunting distribution shown as 2-km squares (shaded) with Corn Bunting records during 2002–2006 (Francis & Cook 2011, RSPB unpubl. data).*

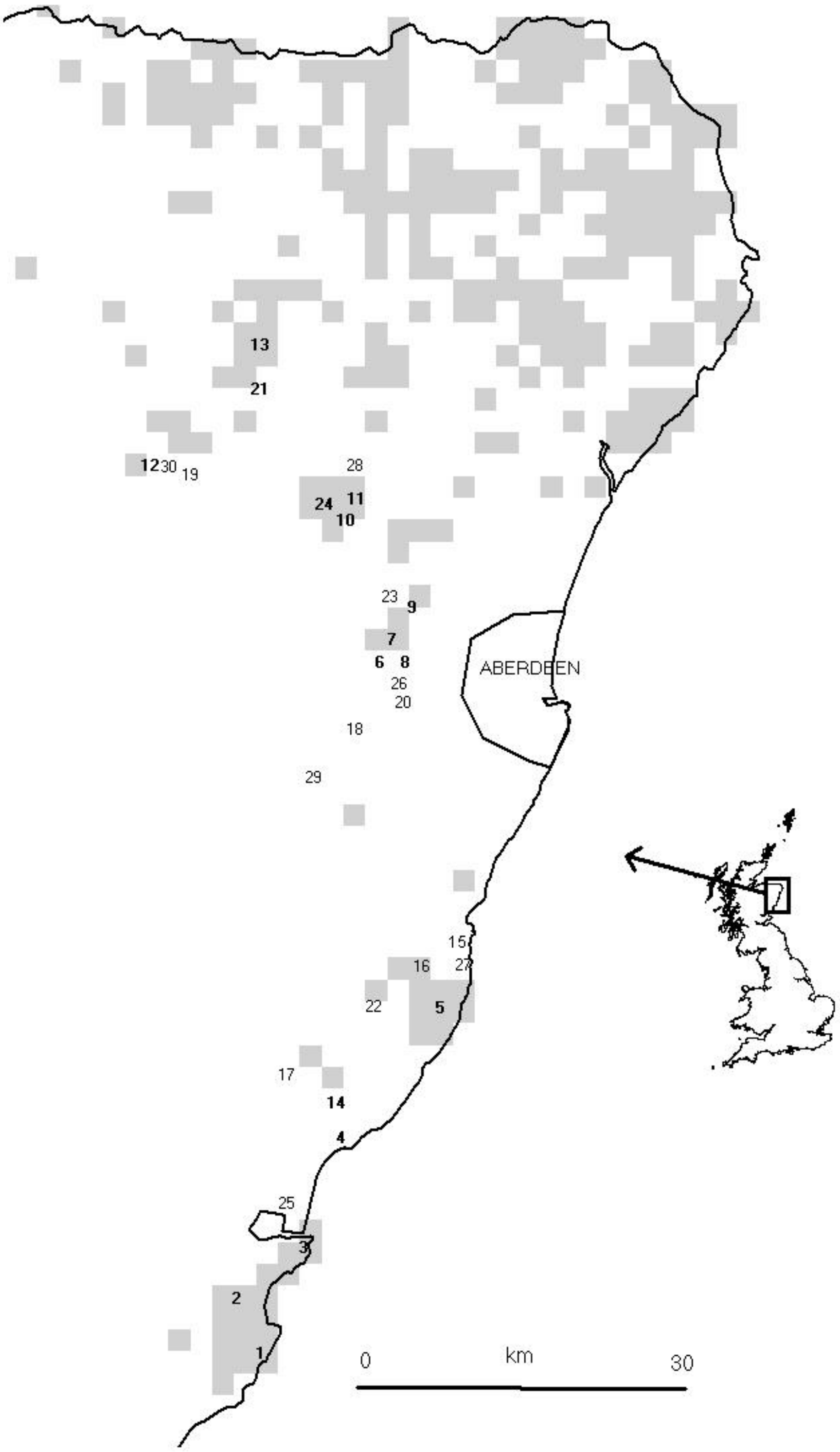


Figure 2.3. *Declines in the number of territorial male Corn Buntings, showing the model estimates for the mean density of males per group (± 1 se) for each year. The fitted line plots the average rate of decline (10.3% per year) across all groups and all years.*

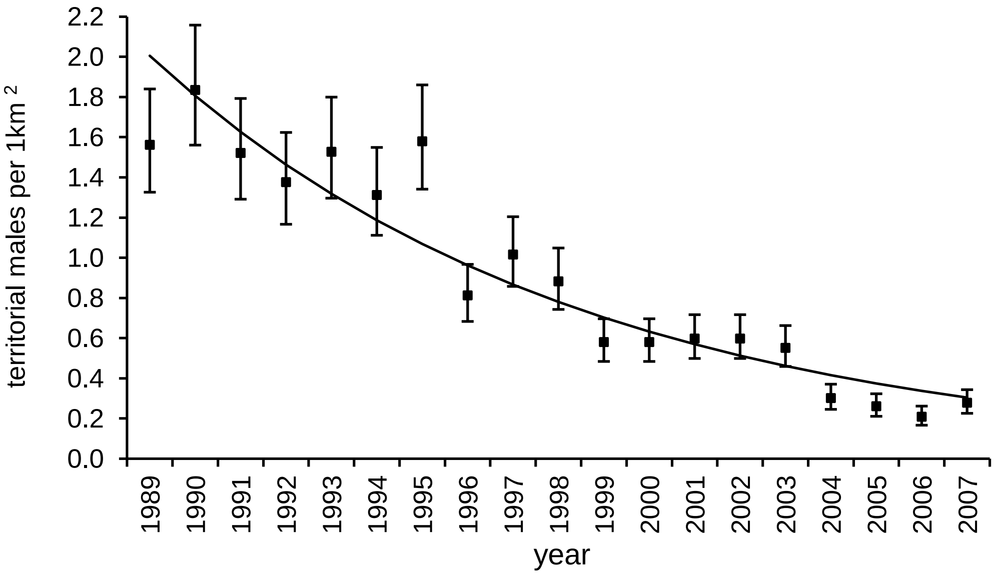
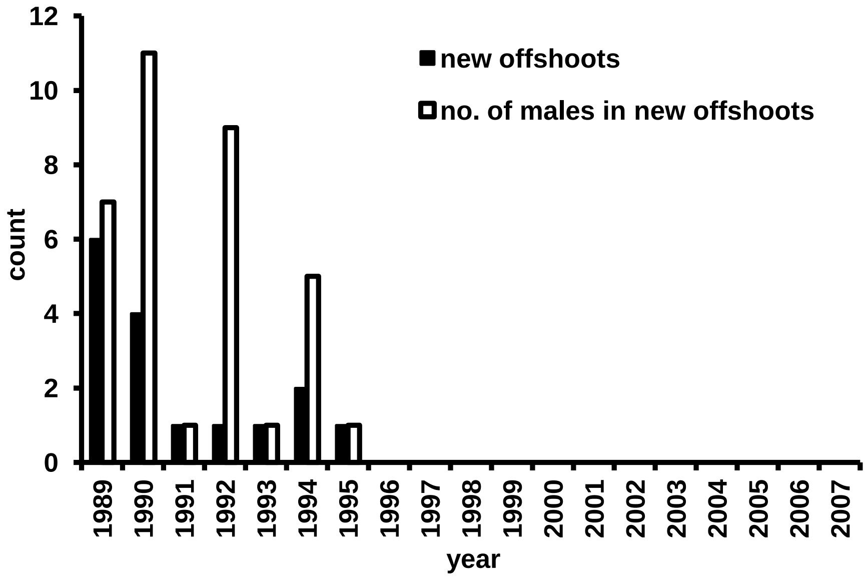


Figure 2.4. *Declines in the number of observed new Corn Bunting offshoots and in total numbers of territorial males in them.*



CHAPTER 3. HABITAT ASSOCIATIONS WITH TERRITORY DISTRIBUTION AND MATING SYSTEM

3.1. Introduction

In the previous chapter, analysis of annual population monitoring data showed that Corn Buntings had declined by 83% across 30 study areas in eastern Scotland since the late 1980s. However, population monitoring cannot easily identify the causes of declines (Clutton-Brock & Sheldon 2010) and studies at a finer scale are required. Knowledge of how best to manage habitats for threatened and declining populations depends upon understanding the species' responses to habitat variation at individual and population levels. This can be achieved by studying species occurrence in relation to spatial variation in habitat (Sergio & Newton 2003, Brotons *et al*. 2004b), and habitat of individual home ranges or territories compared with unoccupied locations (Wiens 1989, Johnson 2007).

Animals defend territories primarily to secure food, shelter from predators, and during the breeding season, a mate and nest site (Newton 1998). Some hold territories for only part of their lifecycle. For example, many species of birds defend breeding territories, but flock together in winter or migrate to milder regions, often in response to seasonal changes in food availability. One such species is the Corn Bunting, where although most populations are non-migratory, birds tend to vacate breeding territories during the winter when they form roaming flocks (Wernham *et al*. 2002).

The ecology of Corn Buntings is generally well known, giving a good indication of the types of habitats that should constitute a high-quality breeding territory. They nest in dense vegetation on or close to the ground, usually within growing crops or tall grasses. Where suitable nesting habitat is available, they rear two broods during a breeding season of over three months (Thompson & Gribbin 1986, Hartley & Shepherd 1994b, Brickle & Harper 2002). The breeding season, in the UK starting in May or June, is later than in many other related species (Crick *et al*. 1994), and the mating system varies between individuals. Approximately a quarter of males are polygynous, typically with two to three females per male (Cramp & Perrins 1994), although the frequency and extent of polygyny varies considerably between areas (Harper 1995). The diet is predominantly cereal grain and seeds of grasses and arable weeds, supplemented in summer by invertebrates fed to chicks (Watson 1992, Hartley & Quicke 1994, Brickle & Harper 1999). In intensively managed farmland, Corn Buntings therefore face increased nest loss from earlier harvesting operations, reductions in invertebrate and weed seed-food because of pesticide use and loss of semi-natural habitats, reductions in cereal grain due to cleaner harvesting, and elimination of winter fallow periods from arable crop rotations (Wilson *et al*. 2007a).

Several previous studies have investigated habitat associations of territorial Corn Buntings during the breeding season (e.g. Møller 1983, Thompson & Gribbin 1986, Hartley *et al*. 1995, Gillings & Watts 1997, Brickle *et al*. 2000, Mason & Macdonald 2000, Golawski & Dombrowski 2002, Lilleør 2007, Brambilla *et al*. 2009). However, as with the majority of studies at the territory level in other farmland bird species (with the notable exception of Grey Partridge – e.g. Potts & Aebsicher 1995) none of the studies listed were long-term (their duration varied from 1–6 years). One problem with studies undertaken over a small number of breeding seasons is that critical insights may be missed because studies are too brief (Wiens 1984). In short studies, one cannot test how habitat associations change over the years in response to changes in agricultural management as populations decline. Moreover, the attributes of territories occupied persistently cannot be distinguished from those readily abandoned (Sergio & Newton 2003). Long-term data allow these questions to be tackled, and offer useful insights into causes of decline and possible conservation interventions (e.g. Sergio *et al*. 2004, Pinto *et al*. 2005). Also, despite the dynamic nature of agricultural habitats due to rapid crop growth and farming operations, I am not aware of any study on Corn Buntings that specifically measured changes in habitat associations during the course of a single breeding season.

Here, we[3] use AW's 20-year (1989–2008) study of Corn Buntings in his largest study area (area/group 5 in Chapter 2) to show habitat attributes of breeding territories, with the aim of determining what makes a high-quality territory for this species in the mixed farming landscape of eastern Scotland, and to identify likely causes of population declines within this region. We did this by testing the following predictions. First, animals tend to occupy more habitats when population density is high, and vacate the poorest ones first as the density falls, leaving the best habitats occupied for longest (Newton 1998). Given that the size of our study population changed, we predicted variation in habitat associations with territories over time. In particular, we predicted strong, positive associations with attributes that consistently

[3] Several colleagues helped with this study. Almost all fieldwork was undertaken by Adam Watson, assisted in earlier years by Mick Marquiss, Robert Rae, Stuart Rae and Des Thompson, and in later years by Amanda Biggins, Alan Bull, Steven Coyne, John McMahon and Hywel Maggs. With advice from Jeremy Wilson, I designed and carried out all data preparation and statistical analyses, and as lead author of the paper (Appendix 2), wrote the first draft and incorporated improvements suggested by the co-authors (Adam Watson, Jeremy Wilson, Hywel Maggs), my supervisor John Deag, and referees/journal editors Nick Brickle, Stephen Browne, Dan Chamberlain, Paul Donald and an anonymous referee.

offered high-quality breeding habitat throughout the study, and ‘new’ habitat associations (positive and negative) in later years as the population declined and contracted into the 'best' remaining territories (those enabling maximium reproductive success), and as agriculture intensified. Second, because multi-brooded farmland species, especially those nesting in crops such as Corn Bunting, may shift locations within the same season (e.g. Gilroy *et al*. 2010), we predicted that territories providing early-summer nesting habitat and late-harvested crops were those most likely to be occupied throughout the breeding season. Third, as Corn Buntings frequently show polygyny (e.g. Hartley *et al*. 1995), we predicted that males with territories offering better food and nest-sites were more likely to be polygynous (Searcy & Yasukawa 1989).

3.2. Methods

3.2.1. Study area

Monitoring covered 3645 ha of coastal farmland in eastern Scotland (Fig. 3.1). During the study, farmers grew autumn-sown or spring-sown cereals, oilseed rape, vegetables, and grass mown for silage or hay, or grazed by cattle, sheep, horses and pigs, plus minor crops (Table 3.1). They left some fields uncultivated (fallow) as rough grass or set-aside. Field boundaries were mostly fences, stone walls and ditches, creating an open landscape interspersed with patches of scrub, small conifer plantations, and trees around houses (Plate 10). Since 2002, agri-environment measures targeted at Corn Buntings to improve breeding success and winter survival were applied on two farms, and extended to a third in 2006, covering 35–50 ha p.a. (1% of the study area). Measures involved delayed mowing or spraying of grass and set-aside until Corn Buntings had finished nesting, extensively managed (fewer pesticides applied) spring cereals followed by over-winter stubbles, and plots of unharvested crops to provide overwinter seed food.

3.2.2. Field methods

To locate territorial male Corn Buntings, repeated visits (by AW) were made to all parts of the study area during May and early June until successive counts showed no increase. Because they sing frequently and conspicuously, two visits were usually sufficient for each sub-area (Perkins *et al*. 2011 and see Chapter 7). Most visits were during early mornings or evenings, in calm weather, when observers can hear songs up to 500 m away. In such periods, all males usually sang and spent much time alert on high lookouts, and females often joined them. This was supplemented by daytime checks in good weather. The most frequently used song-post of each male was recorded on a 1:25 000 map. Females were also located during these and additional observations, especially in early – mid May before nesting had begun, when they showed themselves readily and closely associated with the males, allowing the number paired with each male to be determined. Following the main fledging period of first broods, counts over the whole study area were repeated in early to mid-July, continuing into early August in years with late nesting. All fields (contiguous patches of the same crop type) were mapped each year, and crop type and weed score recorded. For all fields and crop types, non-crop plant cover including under-sown plants in cereal crops (henceforth 'weed score') was estimated visually in the field (0 to 7 scale for 0, 1–2, 2–5, 5–25, 25–50, 50–75, 75–99, and 100% cover), always by the same observer (AW), initially by checking weed scores against many colour photographs. The weed score was averaged across each field prior to analysis. Weed scores were recorded once in early summer at the main period for nestlings in first broods, and again in late summer for nestlings in repeat or second broods. In winter 2006–07, I mapped all telephone and power lines, because Corn Buntings use overhead wires as song-posts. Although we cannot be certain that all wires were present throughout the study, there were no large changes observed.

3.2.3. Characterising Corn Bunting territories

We measured habitat associations of territories by comparing land use centred on male song-posts in early summer with that centred on points selected randomly from the intersections of a 100 m grid covering the study area. The number of grid points selected varied between years but was always within ± 50% of the number of territories. This ensured random variation in the ratio of territories to these null sites between years. This was necessary to allow modelling of year as a random effect to control for repeated measures of the same birds in subsequent years, given that Corn Buntings can live up to seven years (Robinson 2005) and show strong site-fidelity between breeding seasons (Shepherd *et al*. 1997). To represent a nominal territory area, we used a circle of 150 m radius drawn around the mapped position of each male and each randomly selected point (area = 7.02 ha). Workers who mapped boundaries defended by territorial male Corn Buntings reported a mean territory size of 2–6 ha, but females frequently forage outside the male territory when provisioning chicks (Møller 1983, Hartley *et al*. 1995, Brickle *et al*. 2000). We chose a 150 m radius because 95% of nests found in associated studies across eastern Scotland lay within 150 m of the male's main song-post (see Chapter 4). Where two or more circles of the same type (territory or null site) overlapped, we redrew the boundary along the midpoint of the area of overlap. Each year's circle and habitat map was overlaid to determine habitat composition (areas of fields and lengths of linear features) on each territory and null site. Circles with > 10% areas of unknown habitat (e.g. those at the edge of the study area) were excluded (n = 23 circles across all years). All digital mapping used MapInfo Professional version 6.

3.2.4. Statistical modelling

Initially, the dataset contained 45 habitat variables, of which 40 described crop type. To reduce the number of variables in the modelling procedure (see below), we grouped crop types with similar vegetation structure and management (e.g. spring-sown barley and oats). Exceptionally, crop types that appeared to be similar (e.g. autumn-sown wheat and barley) were retained as separate predictors because their means and standard errors differed significantly between territories and null sites (Wilcoxon signed rank test). We retained other habitat variables (e.g. length of overhead wires) based on existing knowledge of Corn Buntings. This process reduced the dataset to nine habitat variables that we considered *a priori* to be important predictors of Corn Bunting territory locations, but whose influence relative to one another was unknown. Each predictor was rescaled to values between 0–1 so that their parameter estimates (and magnitude of effect) could be directly compared (Table 3.2).

The probability of circle status (binary response variable) was modelled as a function of habitat variables in a GLMM (using the SAS GLIMMIX procedure) with a logit-link function and binomial error distribution, with *year* as a random (covariate) effect specified with a first-order autoregressive covariance structure. We modelled three binary response variables, one using all circles and two with just the territory circles: (a) Territory location – circle was a Corn Bunting territory (1) versus null site (0); (b) Late-summer occupancy – territory occupied by male all summer (1) versus early summer only (0); and, (c) Polygyny – territory occupied in early summer by a polygynous male (1) versus a monogamous male (0).

Habitat associations with territory occupancy and mating status, and their change over time, were assessed using information-theoretic methods and model averaging (Burnham & Anderson 2002) in a three stage process. In the first stage, the set of models over which model averaging was carried out was identified using Akaike's information criterion adjusted for small samples (AIC_C). Akaike weights (w_i) were calculated to identify the smallest set of models containing the best approximating model with 95% confidence.

In the second stage, we used the confidence sets derived from the above procedure to determine habitat associations of territory occupancy and mating status, and their change over time. We did this by adding the year|habitat interaction term (whose 'year' component was fitted as a covariate) for each habitat variable in a model to all GLMMs in the confidence set. In these first two stages, we fitted all GLMMs using a maximum likelihood framework (using the Laplace approximation), as recommended when using information-theoretic model selection procedures with mixed models (Bolker *et al*. 2009).

Thirdly, to obtain unbiased parameter estimates and standard errors, all GLMMs in each confidence set were re-fitted using a restricted maximum likelihood framework (REML), allowing use of the Kenward-Roger method to calculate degrees of freedom and standard errors (Bolker *et al*. 2009). We then inspected the model-averaged parameter estimates and standard errors (weighted by the w_i of each model – Burnham & Anderson 2002) for each main effect and interaction term to assess the magnitude of their effects.

Finally, we repeated the modelling of territory location separately for each of the 3-year periods 1989–1991, 1998–2000 and 2005–2007. This was to investigate how the results would have differed had data been available only from short-term (3-year) studies undertaken at the beginning, middle or end of the 20-year period. For this, we followed the same model selection method as in the preliminary modelling stage.

3.2.5. Predictors and sets of models

All possible combinations of the nine predictors gave 511 candidate models. However, predictor variables differed slightly amongst the three sets of models (Table 3.2). Habitat features that stayed constant during the breeding season (*wires* and *boundary*) were not considered in models of late-summer occupancy (b). Instead, we included the male's mating status in early summer - *mated early* - and subdivided *other crops* into *winter rape* and *spring non-cereals* because these crop types are harvested at different times (mid-summer and late summer, respectively).

Correlation matrices for predictor variables are given in Table 3.3. Inter-correlation between individual variables was generally low, the strongest being between *wires* and *boundary* (r_s =

0.361–0.408). Several others showed moderate correlation (0.20 < r_s < 0.30). However, predictor variables that described proportion area of crop type summed to one in 9% of circles, and > 0.9 in 51% of circles. Consequently, individual crop-area predictors were negatively correlated with the combined value of all other crop-area predictors (r_s = 0.382–0.755), so these predictors were not strictly independent of one another. Despite this, modelling results were unlikely to have been greatly affected given the information-theoretic and multi-model inference approach adopted.

3.3. Results

3.3.1. Trends in Corn Buntings and habitats

Territorial male Corn Buntings declined from a peak of 134 in 1990 to nine in 2008, and the range from 41 occupied 1-km squares in 1991 to just four in 2007 and six in 2008. Population size and 1-km square occupancy were strongly correlated across years (n = 20 years, r_s = 0.971, P < 0.0001; Fig. 3.2). The proportion of territories that remained occupied by males throughout summer varied annually from 5% to 68%, but decreased overall from 52% in 1989–91 to 23% in 2006–08 (Fig. 3.3a). The proportions of males apparently mated polygynously (2–6 females) in early summer ranged from 5% to 32%, unmated (0 females) from 0% to 47%, and monogamously (1 female) from 32% to 91% (Fig. 3.3b). These proportions were correlated with population size (polygynous r_s = -0.605, P = 0.005; unmated males r_s = -0.497, P = 0.026; monogamous r_s = 0.660, P = 0.002) (Fig. 3.4a). Therefore, the frequency of unmated males and polygynous males (3% and 8% respectively in 1989–91) increased as the population declined, to 15% each in 2006–08 (Fig. 3.3b). However, the mean number of females observed in polygynous territories during these years fell from 2.62 to 2.00 (Fig. 3.3c). The sex ratio recorded across all breeding territories varied between years from 0.95 to 1.26 females per male (mean = 1.10), but was not correlated with population size (r_s = -0.221, P = 0.348) (Fig. 3.4b).

No large changes in cropping occurred during the 20-year study period, although some crop types showed short-term fluctuations (Table 3.1). Weed scores declined, with mean scores across all fields correlated with year in early summer (r_s = -0.459, P = 0.042) and late summer (r_s = -0.929, P < 0.0001). These declines were particularly large in cereal fields (Fig. 3.5), with mean scores in spring cereals falling by approximately 50% between 1989–91 and 2006–08. Finally, the total length of field boundary across the whole study area was negatively correlated with year (r_s = -0.582, P = 0.007), due mainly to an 18% increase in the mean size of arable fields, from 7.14 ha in 1989–91 to 8.45 ha in 2006–08.

3.3.2. Habitat associations of territory locations, late-summer occupancy and polygyny

3.3.2.1. Preliminary model selection

In preliminary modelling, habitat composition was compared amongst 964 territory/years and 1004 null site/years, and 12 GLMMs formed the 95% confidence set (Table 3.4a). All 12 models included *weeds early*, *wires*, *winter barley*, *spring cereals* and *forage grasses*. Each of these predictors had a selection probability (the probability of it being in the best approximating model) of almost one, indicating very strong support. *Boundary* had a selection probability of 0.803, indicating moderate support, whilst selection probabilities for the other three predictors ranged from 0.469 to 0.712, indicating weak support.

Of the 964 territory/years, 480 were occupied throughout summer and 484 occupied only in early summer. A 95% confidence set of 23 GLMMs (Table 3.4b) described habitat differences between these two groups of territories. Six predictors had strong support with selection probabilities > 0.88. These were *weeds late*, *mated early*, *fallow* (included in all 23 models in the confidence set), *winter barley* (17 GLMMs), *winter rape* (16 GLMMs) and *spring cereals* (19 GLMMs).

Comparison of 118 territory/years with polygynous males versus 772 territory/years with monogamous males yielded a 95% confidence set of 63 GLMMs (Table 3.4c). Predictors with strong support across the 95% confidence set (selection probabilities > 0.84) were *weeds early*, *winter wheat*, *boundary* and *winter barley*. The first of these predictors was in all 63 models in the 95% confidence set, and the other three were in 55, 49 and 50 of these GLMMs, respectively.

For all GLMMs within confidence sets, Cohen’s kappa values ranged from 0.323 to 0.551 (Table 3.4), indicating moderate to good model fit.

3.3.2.2. Habitat associations and year effects – territory location

After incorporating year*habitat interaction effects, inspection of the parameter estimates (Table 3.5a) showed that the effect of *weeds early* (model-averaged parameter estimate =

6.191 ± 1.359 se; Fig. 3.6a) was almost three times greater than that of any of the other habitat variables. *Winter barley* and *spring cereals* had similar magnitudes of effect (parameter estimates = 2.157 ± 0.605 se and 1.859 ± 0.531 se, respectively), and that of *forage grasses* was 1.415 ± 0.588 se and *boundary* was 1.480 ± 0.775 se. All of these were positively associated with the probability of a circle being a Corn Bunting territory. There were two strong year|habitat interactions associated with territory locations (Table 3.5a). These were *year|wires* (0.197 ± 0.057 se) and *year|fallow* (0.224 ± 0.106 se), indicating that a positive association between territories and *wires* was stronger in the later years of the study (Fig. 3.6b), and that territories only became positively associated with *fallow* in later years (Fig. 3.6c). An effect was also apparent for *year|weeds* (0.235 ± 0.171 se), whose positive association with territories was stronger in later years of the study (Table 3.5a; Fig. 3.6a).

3.3.2.3. Habitat associations and year effects – late-summer occupancy

Late-summer occupancy showed strong positive associations with a male's success in attracting a mate in early summer (*mated early*), weed abundance in late summer (*weeds late*) and *spring cereals*, and negative associations with *winter rape* and *winter barley* (Table 3.5b). Inspection of the parameter estimates (Table 3.5b) showed that the effect of *weeds late* (6.982 ± 1.540 se) was more than 2.5 times that of *mated early* (2.787 ± 0.581 se) and almost five times that of *spring cereals* (1.401 ± 0.816 se), and *winter rape* (-1.473 ± 0.941 se), these being the four effects of greatest magnitude. Late-summer occupancy was negatively associated with *fallow*, and a weak *year|fallow* effect (-0.166 ± 0.118 se) suggested that although in later years male Corn Buntings had become increasingly attracted to set-aside and rough grass in early summer (see above), it was increasingly likely in later years that territories with these habitats would be abandoned in mid-summer.

3.3.2.4. Habitat associations and year effects – polygyny

Polygyny was most strongly associated with weed abundance (*weeds early*), and was also positively associated with *boundary*, but negatively associated with *winter wheat* (Table 3.5c). Inspection of the parameter estimates (Table 3.5c) shows that the effect of *weeds early* (13.157 ± 1.905 se) was more than nine times that of *winter wheat* (-1.386 ± 3.057 se), whose effect size was in turn four times that of *boundary* (0.295 ± 1.843 se). Two year|habitat interactions were of note. One was *year|weeds early* (-0.289 ± 0.193 se), suggesting that the positive association of weeds with polygyny was weaker in later years.

The second was *year|winter barley* (-0.479 ± 0.197 se), indicating an increasingly negative association of winter barley with polygyny over time. In addition, although not a variable with a very high selection probability in the initial GLMM without year|habitat effects (0.56), there was also a strong *year|forage grasses* effect (0.350 ± 0.151 se) suggesting the emergence of a positive association between polygynous males and forage grasses in later years.

3.3.3. Comparison between 20-year and 3-year analyses

Habitat composition between territories and null sites was compared separately for each of the 3-year periods 1989–1991 (n = 321 territory/years and 367 null site/years), 1998–2000 (n = 96 territory/years and 108 null site/years) and 2005–2007 (n = 41 territory/years and 37 null site/years). Notably, the 95% confidence sets of models generated from each of these 3-year 'studies' were considerably larger than that generated from models across the full 20-years, and were inversely proportional to sample size (Table 3.6). Nevertheless, in all three short-term 'studies', *weeds early* and *wires* had high selection probabilities (> 0.8), indicating strong and consistent support for a positive association with the probability of a circle being a Corn Bunting territory. Of the other habitat predictors, *spring cereals*, *winter barley*, *forage grasses* and *boundary* (all positive) had selection probabilities > 0.8 in 1989–1991 (Table 3.6a), but in 1998–2000 (Table 3.6b) only *forage grasses* (positive) had strong support, and no other variables had strong support in 2005–2007 (Table 3.6c). Whilst the support shown for *weeds early* and *wires* in all three short-term 'studies' was consistent with results derived from modelling the complete 20-year dataset, two predictors (*spring cereals* and *winter barley*) that had very strong support in the long-term analysis (Table 3.6d) received weak support (selection probabilities < 0.5) in two of the three short-term 'studies'. Further, despite *fallow* having a large positive parameter estimate (model-averaged parameter estimate = 3.298 ± 1.774 se) in the 2005–2007 'study' (Table 3.6c), there was insufficient power for it to receive strong support (selection probability = 0.67), whereas the long-term analysis did reveal a positive association between Corn Bunting territories and *fallow* in later years.

3.4. Discussion

Corn Buntings declined by 91% over 20 years on the study site, part of a wider decline in eastern Scotland (Watson *et al*. 2009, and see Chapter 2). The frequency of late-summer occupancy also declined, from just over half of territories in early years to less than a quarter by the end of the study. As with most Corn Bunting populations, some males were polygynous, with annual rates of 5–32% of males in early summer, and 22–64% in late summer. The frequency of polygyny and of unmated males increased as the population declined, with a corresponding reduction in monogamy (the sex ratio did not change over time). Although lower monitoring duration of each individual territory when the population was high may have led to some females being missed in those years, the mating status of males was determined before nesting had begun, when females were conspicuous. Therefore, it is unlikely that any such bias was sufficient to have greatly influenced our results, and a lack of correlation between the observed sex ratio and population size supports this conclusion. The explanation may be that in later years, only those males holding territories with high-quality habitats were able to attract females, and that such habitats were becoming rarer and more patchily distributed as the years passed. However, cropping remained largely unchanged. The main recorded indications of agricultural intensification were declines in weed score, especially in cereal crops, and an increase in the size of arable fields due to removal of boundaries.

3.4.1. Habitat associations of territory locations

Corn Bunting territories in early summer were strongly associated with weedy fields, winter barley, spring cereals, forage grasses, and wires, and to a lesser extent with field-boundary features. By far the strongest predictor of territory location was weed abundance. Weeds provide dense ground cover within crops that helps to conceal nests and flightless young from predators (Hartley & Shepherd 1994b, Hartley *et al*. 1995), and also host a wide range of invertebrates that are chick-food for many farmland bird species (Wilson *et al*. 1999). Brickle *et al*. (2000) showed that abundance of the main groups fed to Corn Bunting nestlings (*Opiliones*, *Lepidoptera* larvae, *Symphyta* larvae and *Orthoptera*) was positively correlated with chick condition. Further, they found that these invertebrates were more abundant in field areas subjected to fewer pesticide applications, whilst other studies have shown that reduced pesticide use can improve the breeding productivity of Grey Partridge and Yellowhammer (Rands 1985, Hart *et al*. 2006).

Favoured crops included winter barley and forage grasses, which both offer tall dense swards attractive to nesting Corn Buntings during early summer. However, as with other meadow-nesting species such as Corncrake and Whinchat (Green *et al*. 1997, Müller *et al*. 2005), nest losses in forage grasses are high when fields are cut for silage or hay (Wilson *et al*. 2007a, Perkins *et al*. 2011, and see Chapter 5). Winter barley also provides insect-rich foraging habitat in early summer (Douglas *et al*. 2010), and seed food. It is the first cereal crop to ripen, with part-ripe grains typically available from mid-June in eastern Scotland, 3–4 weeks earlier than winter wheat (Watson 1992, and see Chapter 4). The availability of part-ripe cereal grains has been linked to the earlier onset of breeding in Corn Buntings (Brickle & Harper 2002). Spring cereals also offer good pre-breeding foraging habitats, as Corn Buntings frequently eat newly drilled grain in spring (Brickle & Harper 2000). Further, the open sward during early stages of crop growth may provide passerines with easy foraging access to insects and seeds lying on the ground (Morris *et al*. 2002, Menz *et al*. 2009a,b, Gilroy *et al*. 2010). Other attractions of spring cereals are their tendency for greater weed burdens than autumn-sown cereals (Hald 1999, and see Fig.3.5a,b), and provision of nest-sites once crops mature. Finally, overhead wires offer elevated song perches, whose positive association with Corn Bunting territories is well known (Lilleør 2007), whilst field-boundary features such as fences, ditches and farm tracks also provide song-posts and insect-rich foraging habitat (Brickle *et al*. 2000, Mason & Macdonald 2000a).

3.4.2. Seasonal effects

One of our predictions was that late-summer territory occupancy would be associated with the availability of late-harvested crops for nesting. Accordingly, Corn Bunting territories with weedier fields and more spring cereals were the ones most likely to be occupied throughout summer. The growth and maturation of spring cereals as the season progresses makes them increasingly attractive to Corn Buntings for nesting and foraging until their harvest from late August (see Chapter 4). By contrast, territories with winter barley, winter rape, and fallow land such as set-aside were more likely to be abandoned in mid-summer. Set-aside fields are frequently mown or sprayed with herbicides in July to control weeds, thus reducing invertebrate abundance and destroying nesting habitat (Watson & Rae 1997a). Similarly, winter barley and rape fields rapidly deteriorate as nesting or foraging habitats following spraying and harvesting in July and August. Intensive monitoring on our other study sites (see Chapters 4 and 5), including observations of female Corn Buntings nest building whilst still feeding fledglings, confirmed that individual birds switched habitats between nesting attempts. This, rather than subsets of the population breeding at different times in different habitats, was the most likely explanation for the seasonal shift in habitat associations of territorial males. Change in habitat use by a given pair during a single breeding season has been reported in other farmland species (e.g. Wilson *et al*. 1997, Gilroy *et al*. 2010), but too many studies ignore this possibility (Brambilla & Rubolini 2009). To understand fully the conservation requirements of multiple-brooded species, especially those occupying rapidly changing environments such as farmland, one must study habitat associations for the entire breeding season.

3.4.3. Polygyny

Male Corn Buntings occupy breeding territories up to two months earlier than females (e.g. Møller 1983), and selection of a territory or male by a female should aim to maximise her reproductive success. Her choice may be determined by perceived parental quality of the male (from his song, display, size, plumage or dominance over other males), or by the quality of resources within his territory, such as food and nest sites (Wimberger 1988; Johnson & Searcy 1993). In reality, both are often true, as the 'best' males tend to occupy the 'best' territories (Petit 1991). Polygyny has been recorded in 39% of 122 European passerines studied (Møller 1986), but is regular in far fewer (Bennett & Owens 2002). It is often argued that polygamous females suffer greater costs than monogamous females, because they are competing for care by a single male (Slagsvold & Lifjeld 1994; Ranta & Kaitala 1999). However, according to one model (the Polygyny Threshold Model) it may benefit a female to settle as a secondary female on a territory of high quality than as a monogamous female on a poor territory (Verner 1964; Searcy & Yasukawa 1989 and studies therein). Therefore, assuming there is variation in quality between territories, those of polygynous males should contain more food or better nesting habitat than those of monogamous or unmated males.

We predicted that habitat composition of territories would differ between polygynous and monogamous males, with polygynous males occupying habitats with more food and better nest-sites. The only detailed study to date found no such difference, and the authors concluded that polygyny might simply arise through random female settlement within the nesting habitat (Hartley & Shepherd 1995). However, they did not measure fine-scale variation in habitat quality, such as weed abundance, although their study was on low-intensity crofting land and machair where all territories may have been of similar quality and capable of supporting polygyny. In contrast, our study did detect habitat differences. Territories occupied by multiple females had fields with higher weed scores, a greater length of field boundary per unit area, and less winter cereal than territories with just one female. One possibility is that winter cereals can support monogamous pairs of Corn Buntings (hence the early summer territory association with winter barley), but less often provide enough invertebrate food to support polygyny, due to low weed abundance in these crops (Fig. 3.5a,b).

3.4.4. Temporal trends in habitat associations

Our final prediction was that the combined effects of population decline and intensification of land use would lead to changes in habitat associations over time. This was indeed the case. A stronger association with wires in later years perhaps reflected fewer territories that had combined the best song-posts (wires) with high-quality nesting or foraging habitat in the early years when the population was large. Territory associations with weeds also became stronger over time, and in later years, a new territory association with fallow land emerged. Although this was partly due to targeting of set-aside management at breeding Corn Buntings since 2002, these agri-environment measures affected just 1% of the study area. Therefore, it seems likely that an overall decline in quality of other habitats also contributed, such as the increasing scarcity of weedy cereals. By 2004–2008, just 1% of spring barley fields had > 5% weed cover in late summer, ten times fewer fields than in 1989–1993 (Fig. 3.5c). Declining weed scores in cereals may also explain changes in habitat associations with polygyny (weaker weed and stronger crop associations in later years). For example, in later years, polygyny was positively associated with forage grasses, and showed a stronger negative association with winter barley. This may relate to females switching nesting habitat preference from cereals to grass fields in later years, due to grass swards increasingly providing denser nest cover than cereals. Such changes, however, could have exacerbated the population decline because grass fields are not always the 'best' habitats. High rates of nest loss from mowing in grass silage fields make them an ecological trap (Battin 2004, and see Chapter 5).

3.4.5. Conservation implications

Corn Bunting declines have previously been linked to changes in cropping, notably localised reductions in the area of cereals grown (Donald *et al*. 1994) and to the increasing trend for autumn-sowing of cereals (Brickle & Harper 2002). Because winter cereals are harvested up to one month earlier than spring cereals, late-summer nesting habitats are often scarce in the modern farming landscape, restricting female Corn Buntings to just one brood (Brickle & Harper 2002). Autumn-sowing also removes the opportunity for overwinter stubbles, which are important foraging habitats for Corn Buntings outside the breeding season (e.g. Perkins *et al*. 2008a and see Chapter 6). In the present study, there were no clear trends in cropping across the 20 years and many crops (c. 40% of cereals) were spring-sown, yet the Corn Bunting population declined to near-extinction. Lower weed scores in later years, however, suggested that crop management had become more intensive. National statistics on herbicide use support this conclusion, as the annual active-substance-treated area of cereals increased by 53% across Scotland from 1990 to 2008 (FERA 2011). The practice of under-sowing cereals with grass or clover to create rotational leys in traditional crop rotations has also declined, and although we do not have trend data, the area of grass under-sown to crops in northeast Scotland in 2008 was just 7106 ha, approximately 3.8% of arable land (http://www.scotland.gov.uk/Topics/Statistics/Browse/Agriculture-Fisheries/Pub EconomicReport. Accessed 28 March 2011). By contrast, in the 1960s, one in ten arable fields was under-sown (Shrubb 2003).

Several studies have shown that intensity of crop management is important for this species (Donald & Aebischer 1997, Brickle *et al*. 2000, Fox & Heldbjerg 2008, Setchfield *et al*. 2012), and the strong influence of weed abundance on all measured aspects of Corn Bunting territory occupancy in the present study is consistent with those findings. It is therefore likely that intensification of crop management rather than changes in cropping areas was the main cause of the population decline in our eastern Scottish study area. Plausible mechanisms for this are suppressed weed and invertebrate abundance reducing breeding productivity through poorer nest concealment and chick diet, and females increasingly attracted to nest in the denser swards of forage grasses and set-aside where nest loss rates to mowing are high.

Conservation recommendations arising from the present study include the need to provide early-summer and late-summer nesting habitats close to one another. This gives female Corn Buntings the opportunity to rear two broods in what can be a prolonged breeding season (late May to early September in eastern Scotland). The findings suggest that winter barley or late-cut hay grown alongside weed-rich or under-sown spring cereals would be a good combination. Set-aside or similar agri-environment crop types also attract breeding Corn Buntings and should remain uncut and unsprayed throughout the breeding season. Their placement next to high song-posts such as wires will increase the likelihood of occupancy.

3.4.6. Value of long-term studies

Long-term datasets allow workers to detect associations or changes that are episodic, cumulative, or acting slowly over many years (Silvertown *et al*. 2010). Here, we can demonstrate the value of long-term study by comparing findings from the full 20-year analysis with three short-term 'studies' using 3-year subsets of the data. Although some relationships were consistent across all four analyses (weeds and wires), others were apparent in only one or two of the short-term studies. Had data just been available for the middle (1998–2000) or late years (2005–2007), for example, we would not have detected territory associations with winter barley and spring cereals. Instead, results would have emphasised associations with forage grasses and fallow land such as set-aside. Without knowledge that Corn Buntings have high rates of nest loss in these last two habitats, such findings could have led to risky conservation recommendations. Furthermore, the 20-year analysis shows that the association with fallow changed over time and was weak or even negative in earlier years, when other habitats such as cereal fields were weedier and more attractive to Corn Buntings. Clearly, short-term studies cannot detect temporal changes such as these, and insights into possible causes of population declines are lost.

Finally, long-term studies often give a better indication of the true baseline of a study population and the environment in which it occurs (Silvertown *et al*. 2010). When the present study began in 1989, the Corn Bunting population was three times and eight times higher, respectively, than during 1998–2000 and 2005–2007, with mean weed scores in spring cereals approximately 50% and 150% higher. This information helps us to appreciate what has been lost from agricultural landscapes over the past two decades, and how rapidly populations can collapse. It also shows what might be achievable in the future through concerted conservation actions to reverse population declines.

Table 3.1. *Area (ha) of crop types across the study site (total = 3645 ha).*

Year	SB	SO	WW	WB	OSR	SRLIN	ROOT	FG	PAS	RES	ROU	LEG	DAFF
1989	797	48	546	305	478	0	244	501	555	19	117	0	17
1990	618	34	536	364	584	24	198	344	753	0	137	0	27
1991	584	17	593	361	609	2	235	395	610	10	157	18	33
1992	483	46	624	384	445	77	228	459	641	0	176	28	29
1993	468	95	404	262	336	209	202	715	481	0	278	134	38
1994	584	31	440	285	211	308	180	544	556	42	269	129	29
1995	571	37	576	353	173	175	218	530	629	19	243	65	30
1996	675	60	471	341	162	72	289	570	630	40	205	43	47
1997	810	38	469	366	181	191	117	629	559	7	141	60	48
1998	633	106	489	438	272	187	194	452	540	6	193	58	46
1999	914	56	374	327	200	100	221	537	528	37	247	21	59
2000	669	48	519	524	133	91	114	717	472	16	199	43	63
2001	818	66	319	514	172	33	148	663	521	0	228	93	50
2002	734	35	310	552	167	44	128	609	555	0	284	157	55
2003	680	41	349	451	215	82	142	471	621	20	291	175	64
2004	663	29	513	381	247	41	195	479	614	12	240	111	73
2005	821	39	510	351	228	13	209	520	548	8	246	27	74
2006	662	0	479	504	201	0	150	498	649	26	286	89	50
2007	806	21	548	519	234	0	147	451	537	3	224	79	42
2008	777	39	411	512	344	0	183	372	640	36	151	84	58

SB = spring-sown barley Hordeum vulgare; SO = spring-sown oats Avena sativa; WW = autumn-sown wheat Triticum aestivum; WB = autumn-sown barley; OSR = autumn-sown oilseed rape Brassica napus; SRLIN = spring-sown oilseed rape and linseed Linum usitatissimum; ROOT = root vegetables (mainly potatoes Solanum tuberosum and turnips Brassica rapa (> 75% in most years), broccoli B. oleracea, and a small area of carrots Daucus carota and cabbages (mostly < 5%); FG = forage grass mown for silage or hay; PAS = grazed pasture (sheep, cattle, pigs and horses); RES = newly-sown grass (excludes grasses under-sown to crops); ROU = rough grass and set-aside (rotational or non-rotational), and includes agri-environment measures since 2002 (3–21 ha p.a. of unharvested crops and 10–31 ha p.a. of late-cut or late-sprayed set-aside); LEG = legumes (peas Pisum sativum, beans Vicia faba); DAFF = daffodils Narcissus pseudonarcissus.

Table 3.2. *Predictors used in modelling habitat associations with (a) Territory location (response variable = territory or null site); (b) Late-summer occupancy (territory occupied by male all summer or early summer only); (c) Polygyny (territory occupied in early summer by a polygynous male or a monogamous male).*

Predictor	Description	Analysis
wires ^{l*}	Overhead wires of telephone and power lines (excluding high-voltage lines mounted on pylons)	(a)(c)
boundary ^{l*}	Field boundaries (either side of roads/tracks measured separately)	(a)(c)
spring cereals ^{p}	Spring-sown cereals, first-year unharvested crops and newly-sown grass	(a)(b)(c)
forage grasses ^{p}	Grass mown for silage or hay	(a)(b)(c)
fallow ^{p}	Rotational and non-rotational set-aside, rough grass and unharvested crops	(a)(b)(c)
winter barley ^{p}	Autumn-sown barley	(a)(b)(c)
winter wheat ^{p}	Autumn-sown wheat	(a)(b)(c)
other crops ^{p}	All non-cereal crops (vegetables, rape, linseed, legumes and daffodils)	(a)(c)
weeds early *	Highest weed score of any field within a circle during early summer	(a)(c)
weeds late *	Highest weed score of any field within a circle during late summer	(b)
winter rape ^{p}	Autumn-sown oilseed rape	(b)
spring non-cereals ^{p}	Spring-sown non-cereal crops (vegetables, rape, linseed, legumes and daffodils)	(b)
mated early	Paired status of male in early summer (0 = no female, unmated; 1 = at least one female, mated)	(b)

^{p} *= proportion area of circle;* ^{l} *= length (m) per 1 ha of circle area;* * *= converted to 0–1 scale by dividing by the maximum observed value (286 for boundary and 131 for wires), or by 100 for weeds early and weeds late.*

Table 3.3. *Correlation matrices of predictors used in models, showing Spearman's coefficients (r_s values).* ***Bold*** *= statistically significant ($P < 0.05$) with a moderate degree of correlation ($r_s > 0.200$).* [a] *summed value of all predictors measuring crop area, excluding the column predictor;* [b] *combined crop cover excludes spring non-cereals and winter rape;* [c] *combined crop cover excludes other crops.*

a) *All circles (n = 1968), early summer.*

	boundary	*spring cereals*	*forage grasses*	*fallow*	*winter wheat*	*winter barley*	*other crops*	*weeds early*	*spring non-cereals*	*winter rape*
wires	**0.408**	0.074	0.035	0.007	-0.002	0.076	-0.048	0.155	0.002	-0.023
boundary		0.122	0.143	**0.207**	-0.018	0.035	0.028	**0.256**	0.139	-0.024
spring cereals			-0.087	-0.003	-0.197	**-0.212**	**-0.206**	**0.239**	-0.036	**-0.204**
forage grasses				0.006	-0.169	-0.157	**-0.226**	0.083	-0.118	-0.164
fallow					-0.142	-0.068	-0.098	**0.275**	0.018	-0.116
winter wheat						-0.005	-0.025	**-0.238**	-0.054	0.019
winter barley							-0.095	-0.098	**-0.215**	0.010
other crops								-0.033		
weeds early									0.109	-0.148
spring non-cereals										-0.107
Combined crop cover [a]		[b] **-0.650**	[b] **-0.555**	[b] **-0.382**	[b] **-0.480**	[b] **-0.513**	[b] **-0.572**		[c] **-0.451**	[c] **-0.470**

Table 3.3 cont.

b) *Territory circles only (n = 964), early summer.*

	boundary	*spring cereals*	*forage grasses*	*fallow*	*winter wheat*	*winter barley*	*other crops*	*weeds early*	*spring non-cereals*	*winter rape*
wires	**0.361**	0.011	0.007	0.001	0.025	0.034	-0.063	-0.055	-0.006	-0.043
boundary		0.058	0.137	**0.203**	-0.007	-0.039	0.065	**0.215**	0.173	-0.029
spring cereals			-0.137	-0.028	**-0.208**	**-0.296**	**-0.275**	0.135	-0.064	**-0.278**
forage grasses				-0.051	-0.167	**-0.209**	-0.174	0.038	-0.067	-0.151
fallow					-0.123	-0.046	-0.089	**0.293**	0.026	-0.124
winter wheat						-0.021	0.011	**-0.235**	-0.079	0.106
winter barley							-0.129	-0.184	-0.191	0.017
other crops								0.016		
weeds early									0.188	-0.188
spring non-cereals										-0.085
Combined crop cover [a]		[b] **-0.755**	[b] **-0.639**	[b] **-0.435**	[b] **-0.525**	[b] **-0.618**	[b] **-0.643**		[c] **-0.501**	[c] **-0.503**

Table 3.4. *Subset of GLMMs (prior to adding year*habitat interaction terms) for which there is 95% confidence that the set contains the best approximating model (Akaike weights w_i sum to 0.95), presented in rank order where model 1 is the best fitting model (smallest AIC_C value). For analysis (c) with a 95% confidence set > 30 models, only those GLMMs whose AIC_C value was within 2 of the best fitting model (Δ $AIC_C \leq 2$) are presented. Cohen's kappa values indicate model fit on a 0–1 scale (1 = perfect fit), based on the proportion of circles that the model correctly predicts to be in each group when applied to the raw data (Manel et al. 2001). Y = predictor included in the model. For each predictor: S Prob = selection probability (the probability of a predictor being in the best approximating model, calculated by summing the Akaike weights w_i of all candidate models containing that predictor); Param and SE = model-averaged parameter estimate and standard error across the 95% confidence set of models (derived from Akaike weights w_i); N = number of models in 95% confidence set that included the predictor.*

a) *Territory location (n = 964 Corn Bunting territory/years and 1004 null site/years; AIC_C of model 1 = 2284.78).*

Model rank	*weeds early*	*wires*	*winter barley*	*spring cereals*	*forage grasses*	*boundary*	*fallow*	*winter wheat*	*other crops*	Δ AIC_C	w_i	Cohen's kappa
1	Y	Y	Y	Y	Y	Y	Y	Y	Y	0	0.246	0.400
2	Y	Y	Y	Y	Y	Y	Y	Y	-	1.07	0.144	0.400
3	Y	Y	Y	Y	Y	Y	Y	-	-	1.55	0.113	0.400
4	Y	Y	Y	Y	Y	Y	-	-	-	2.19	0.082	0.402
5	Y	Y	Y	Y	Y	Y	-	Y	-	2.42	0.073	0.411
6	Y	Y	Y	Y	Y	Y	Y	-	Y	2.76	0.062	0.399
7	Y	Y	Y	Y	Y	Y	-	Y	Y	3.28	0.048	0.402
8	Y	Y	Y	Y	Y	-	Y	Y	Y	3.40	0.045	0.402
9	Y	Y	Y	Y	Y	-	Y	-	-	3.59	0.041	0.398
10	Y	Y	Y	Y	Y	-	Y	Y	-	3.59	0.041	0.400
11	Y	Y	Y	Y	Y	Y	-	-	Y	3.98	0.034	0.396
12	Y	Y	Y	Y	Y	-	-	-	-	4.92	0.021	0.401
S Prob	> 0.999	> 0.999	> 0.999	> 0.999	> 0.999	0.803	0.712	0.619	0.469			
Param	7.936	2.727	1.881	1.449	1.439	0.781	0.908	0.534	0.370			
SE	0.770	0.256	0.276	0.230	0.250	0.350	0.429	0.291	0.259			

Table 3.4 cont.

b) *Late-summer occupancy (n = 480 territory/years occupied by male all summer and 484 early summer only; AIC$_C$ of model 1 = 1133.50).*

Model rank	*weeds late*	*mated early*	*fallow*	*winter rape*	*winter barley*	*spring cereals*	*winter wheat*	*spring non-cereals*	*forage grasses*	Δ AIC$_C$	w_i	Cohen's kappa
1	Y	Y	Y	Y	Y	Y	Y	-	-	0	0.213	0.337
2	Y	Y	Y	Y	Y	Y	Y	Y	-	1.07	0.125	0.323
3	Y	Y	Y	Y	Y	Y	-	-	-	1.22	0.116	0.347
4	Y	Y	Y	Y	Y	Y	-	Y	-	1.55	0.098	0.352
5	Y	Y	Y	Y	Y	Y	Y	-	Y	2.07	0.076	0.335
6	Y	Y	Y	Y	Y	Y	-	Y	Y	2.92	0.050	0.347
7	Y	Y	Y	Y	Y	Y	-	-	Y	3.17	0.044	0.352
8	Y	Y	Y	Y	Y	Y	Y	Y	Y	3.20	0.043	0.325
9	Y	Y	Y	Y	Y	-	Y	-	Y	3.27	0.042	0.337
10	Y	Y	Y	Y	Y	-	Y	-	-	3.44	0.038	0.347
11	Y	Y	Y	Y	-	Y	-	Y	Y	4.79	0.019	0.354
12	Y	Y	Y	Y	Y	-	Y	Y	Y	5.31	0.015	0.335
13	Y	Y	Y	Y	Y	-	Y	Y	-	5.34	0.015	0.339
14	Y	Y	Y	-	-	Y	-	Y	Y	5.83	0.012	0.345
15	Y	Y	Y	-	Y	Y	-	Y	Y	6.15	0.010	0.345
16	Y	Y	Y	Y	-	Y	Y	Y	Y	6.84	0.007	0.356
17	Y	Y	Y	Y	-	Y	-	Y	-	7.06	0.006	0.372
18	Y	Y	Y	-	Y	Y	-	Y	-	7.35	0.005	0.376
19	Y	Y	Y	Y	-	Y	-	-	Y	7.63	0.005	0.356
20	Y	Y	Y	-	-	Y	Y	Y	Y	7.88	0.004	0.345
21	Y	Y	Y	-	Y	Y	Y	Y	Y	8.04	0.004	0.345
22	Y	Y	Y	-	Y	Y	Y	Y	-	8.24	0.003	0.366
23	Y	Y	Y	-	Y	Y	-	-	-	8.25	0.003	0.389
S Prob	> 0.999	> 0.999	0.982	0.940	0.923	0.883	0.603	0.438	0.355			
Param	5.650	2.734	-2.011	-1.494	-1.145	0.922	-0.866	0.681	0.147			
SE	0.769	0.560	0.602	0.471	0.380	0.330	0.456	0.550	0.374			

Table 3.4 cont.

c) *Polygyny (n = 118 polygynous and 772 monogamous male territory/years in early summer; AIC_C of model 1 = 417.62)*

Model rank	*weeds early*	*winter wheat*	*boundary*	*winter barley*	*wires*	*forage grasses*	*spring cereals*	*other crops*	*fallow*	Δ AIC_C	w_i	Cohen's kappa
1	Y	Y	Y	Y	Y	-	-	-	-	0	0.066	0.497
2	Y	Y	Y	Y	Y	-	-	-	Y	0.49	0.051	0.479
3	Y	Y	Y	Y	Y	Y	-	-	-	0.57	0.049	0.497
4	Y	Y	Y	Y	-	-	-	-	-	0.65	0.047	0.551
5	Y	Y	Y	Y	Y	Y	Y	Y	-	0.83	0.043	0.492
6	Y	Y	Y	Y	Y	Y	Y	-	-	1.03	0.039	0.517
7	Y	Y	Y	-	Y	Y	Y	Y	-	1.04	0.039	0.497
8	Y	Y	Y	Y	-	Y	-	-	-	1.30	0.034	0.540
9	Y	Y	Y	Y	Y	-	Y	-	-	1.64	0.029	0.477
10	Y	Y	Y	Y	-	-	-	-	Y	1.67	0.029	0.528
11	Y	Y	Y	Y	Y	Y	-	-	Y	1.73	0.028	0.483
12	Y	Y	Y	Y	-	Y	Y	Y	-	1.75	0.027	0.504
13	Y	Y	Y	Y	Y	-	-	Y	-	1.89	0.026	0.509
14	Y	Y	Y	-	-	Y	Y	Y	-	2.00	0.024	0.497
N=63	63	55	49	50	28	39	36	32	29			
S Prob	> 0.999	0.960	0.924	0.842	0.581	0.562	0.487	0.414	0.348			
Param	9.604	-5.070	2.481	-2.541	-1.292	1.153	1.017	1.126	-0.870			
SE	0.910	2.111	0.942	1.013	0.757	0.647	0.595	0.718	0.911			

Table 3.5. *Habitat associations and their variation over time, following the addition of year|habitat interaction effects to each model in the 95% confidence sets (see Table 3.4 for further details of these sets of models). All models included year as a random covariate effect (specified with a first-order autoregressive covariance structure), whilst the 'year' component of an interaction term was also fitted as a covariate. Model-averaged parameter estimates (Par) and standard errors (se) across each set of models (N), weighted by the Akaike weight (w_i) of each model, are shown. S Prob = selection probability of main habitat terms in the preliminary model selection stage. This is the probability of a predictor being in the best approximating model, calculated by summing the w_i of all candidate models containing that predictor.* ***Bold*** *= predictor with high (> 0.8) selection probability. N = number of models across which Par and se were averaged (GLMMs in the 95% confidence set that included the main habitat term).* [a] *weeds early used in (a) and (c), and weeds late in (b).*

	(a) Territory location (95% set = 12 GLMMs) *n* = 964 territories and 1004 null sites				**(b)** Late-summer occupancy (95% set = 23 GLMMs) *n* = 480 all summer and 484 early summer				**(c)** Polygyny (95% set = 63 GLMMs) *n* = 118 polygynous and 772 monogamous			
	Par	se	S Prob	N	Par	se	S Prob	N	Par	se	S Prob	N
Intercept	-2.134	0.594	-	12	-3.123	0.835	-	23	-4.218	0.886	-	63
Mated early					**2.787**	**0.581**	**> 0.999**	**23**				
Weeds [a]	**6.191**	**1.359**	**> 0.999**	**12**	**6.982**	**1.540**	**> 0.999**	**23**	**13.157**	**1.905**	**> 0.999**	**63**
Year|weeds [a]	0.235	0.171	-	12	-0.127	0.164	-	23	-0.289	0.193	-	63
Wires	**1.447**	**0.441**	**> 0.999**	**12**					-1.426	1.529	0.581	28
Year|wires	0.197	0.057	-	12					-0.031	0.162	-	28
Winter barley	**2.157**	**0.605**	**> 0.999**	**12**	**-0.648**	**0.839**	**0.923**	**17**	**1.231**	**1.342**	**0.842**	**50**
Year|winter barley	-0.038	0.084	-	12	-0.051	0.107	-	17	-0.479	0.197	-	50
Spring cereals	**1.895**	**0.531**	**> 0.999**	**12**	**1.401**	**0.816**	**0.883**	**19**	-0.682	1.094	0.487	36
Year|spring cereals	-0.058	0.074	-	12	-0.055	0.109	-	19	0.173	0.110	-	36
Forage grasses	**1.415**	**0.588**	**> 0.999**	**12**	0.486	0.831	0.355	13	-2.358	1.496	0.562	39
Year|forage grasses	0.007	0.083	-	12	-0.016	0.115	-	13	0.350	0.151	-	39
Boundary	**1.480**	**0.775**	**0.803**	**8**					**0.295**	**1.843**	**0.924**	**49**
Year|boundary	-0.097	0.107	-	8					0.207	0.177	-	49
Fallow	-0.747	0.856	0.712	7	**-0.349**	**1.141**	**0.982**	**23**	-2.759	2.207	0.348	29
Year|fallow	0.224	0.106	-	7	-0.166	0.118	-	23	0.174	0.172	-	29
Winter wheat	1.061	0.649	0.619	6	-0.567	0.940	0.603	12	**-1.386**	**3.057**	**0.960**	**55**
Year|winter wheat	-0.084	0.099	-	6	-0.027	0.144	-	12	-0.739	0.503	-	55
Other crops	1.023	0.645	0.469	5					0.410	1.410	0.414	32
Year|other crops	-0.107	0.099	-	5					0.034	0.174	-	32
Winter rape					**-1.473**	**0.941**	**0.940**	**16**				
Year|winter rape					0.045	0.148	-	16				
Spring non-cereals					0.857	1.103	0.438	15				
Year|spring non-cereals					-0.015	0.149	-	15				

Table 3.6. *Model-averaged parameter estimates (Par), standard errors (se) and selection probabilities (S Prob) of habitat variables across each set of models (N) in separate analyses modelling territory location (territories and null sites) for the 3-year subsets of the data (a) 1989–91, (b) 1998–2000, and (c) 2005–07. This shows how the results would have varied between short-term (3-year) studies undertaken at the beginning, middle and end of the 20-year period. Also shown are (d) results from the complete 20-year dataset, to allow direct comparison between all four periods. All models included year as a random covariate effect (specified with a first-order autoregressive covariance structure). N = number of models across which Par and se were averaged (GLMMs in the 95% confidence set that included the habitat term).* ***Bold*** *= predictor with high (> 0.8) selection probability.*

	(a) Early years 1989–1991 *n* = 321 territory/years and 367 null site/years (95% set = 25 GLMMs)				**(b)** Middle years 1998–2000 *n* = 96 territory/years and 108 null site/years (95% set = 70 GLMMs)				**(c)** Late years 2005–2007 *n* = 41 territory/years and 37 null site/years (95% set = 158 GLMMs)				**(d)** All years 1989–2008 *n* = 964 territory/years and 1004 null site/years (95% set = 12 GLMMs)			
	Par	se	S Prob	N	Par	se	S Prob	N	Par	se	S Prob	N	Par	Se	S Prob	N
Intercept	-2.847	0.399	-	25	-2.512	5.567	-	70	7.800	8.852	-	158	-2.010	0.203	-	12
Weeds early	**6.593**	**1.337**	**> 0.999**	**25**	**4.425**	**1.385**	**0.993**	**70**	**12.904**	**4.387**	**0.993**	**158**	**7.936**	**0.770**	**> 0.999**	**12**
Wires	**1.393**	**0.432**	**0.980**	**24**	**6.749**	**1.009**	**> 0.999**	**70**	**3.465**	**1.502**	**0.878**	**113**	**2.727**	**0.256**	**> 0.999**	**12**
Winter barley	**1.474**	**0.457**	**0.971**	**23**	0.392	0.845	0.283	30	-1.658	1.239	0.428	70	**1.881**	**0.276**	**> 0.999**	**12**
Spring cereals	**1.790**	**0.376**	**> 0.999**	**25**	0.899	0.733	0.408	32	1.158	1.307	0.247	60	**1.449**	**0.230**	**> 0.999**	**12**
Forage grasses	**1.230**	**0.448**	**0.912**	**18**	**2.196**	**0.819**	**0.921**	**60**	0.457	1.762	0.266	65	**1.439**	**0.250**	**> 0.999**	**12**
Boundary	**1.472**	**0.613**	**0.924**	**18**	-1.448	1.470	0.354	30	2.428	1.825	0.370	78	**0.781**	**0.350**	**0.803**	**8**
Fallow	-0.811	0.952	0.359	11	0.850	1.318	0.293	30	3.298	1.774	0.669	89	0.908	0.429	0.712	7
Winter wheat	0.673	0.454	0.465	11	-0.511	1.045	0.291	33	-2.473	1.829	0.412	69	0.534	0.291	0.619	6
Other crops	0.677	0.387	0.612	12	-1.462	1.024	0.543	40	0.703	1.300	0.259	60	0.370	0.259	0.469	5

Figure 3.1. *Map of study area, showing field boundaries and distribution of Corn Bunting territories (black dots) in 1990, the year with highest population (134 territories). Approximate centre of study area = 56º54´ N, 2º15´ W.*

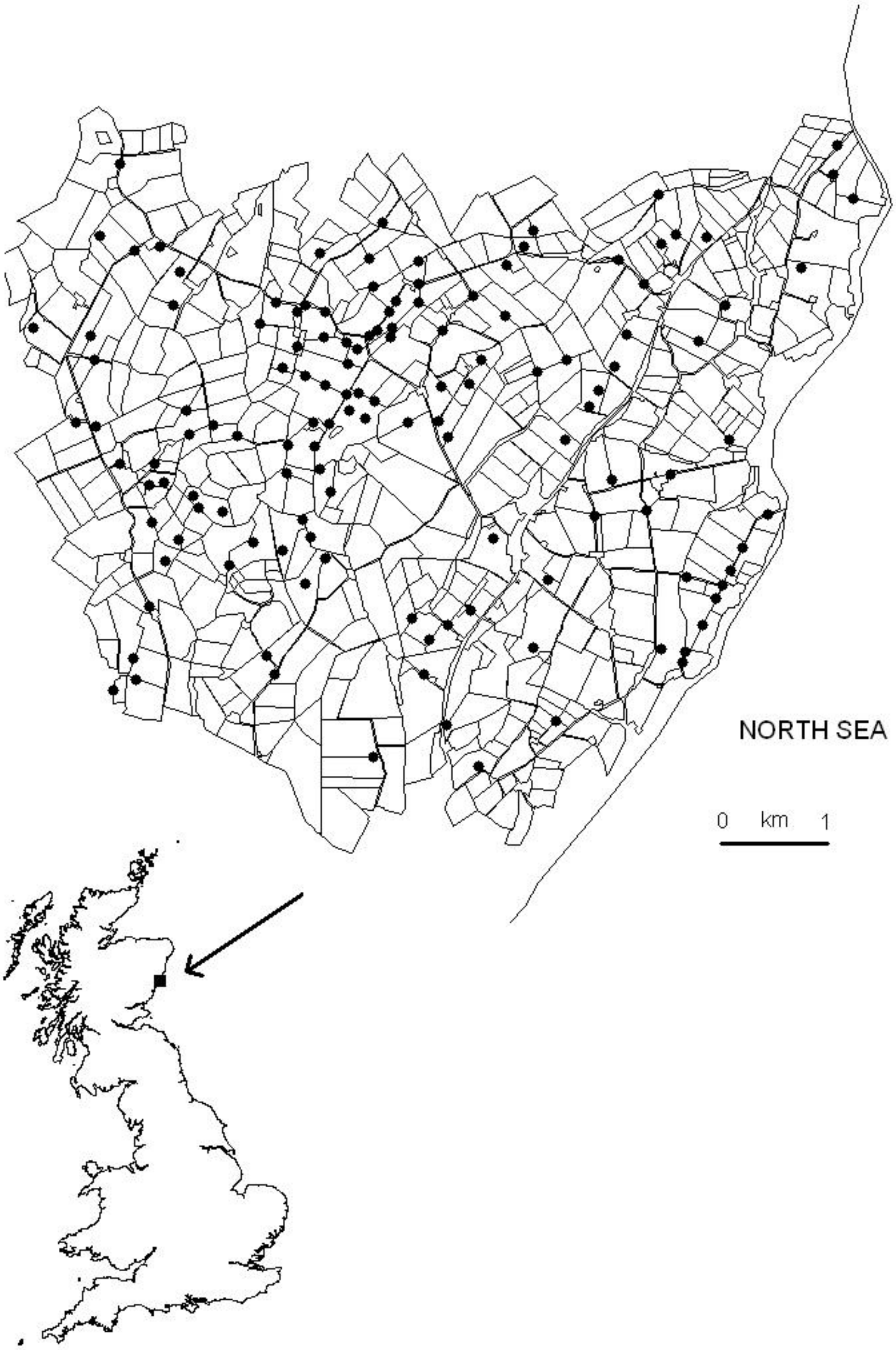


Figure 3.2. *Annual population size (territorial males) and distribution (occupied 1-km squares), 1989 – 2008.*

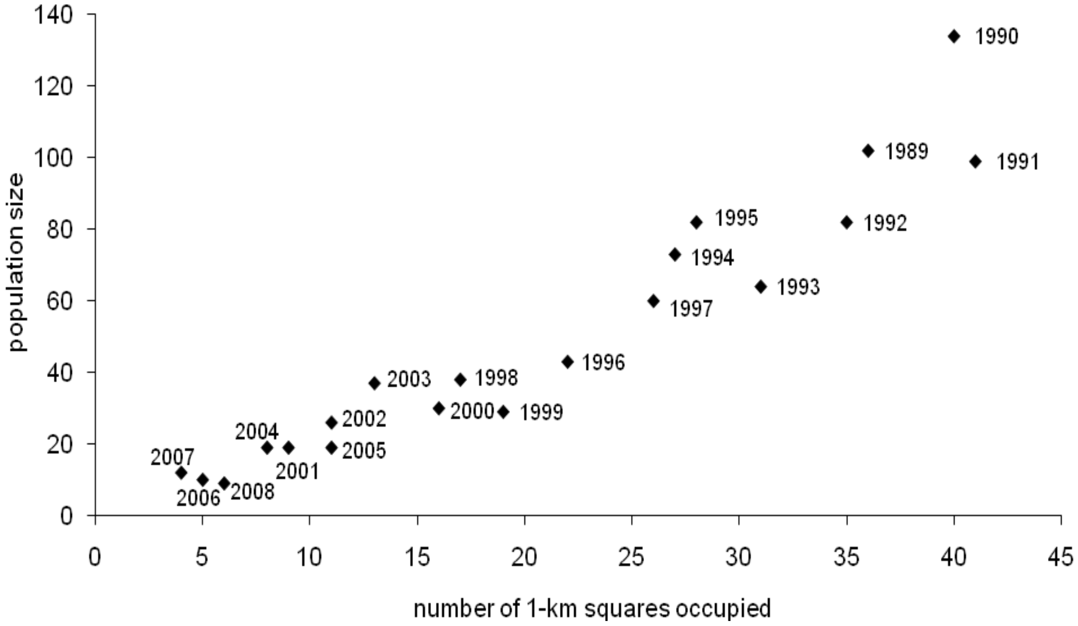
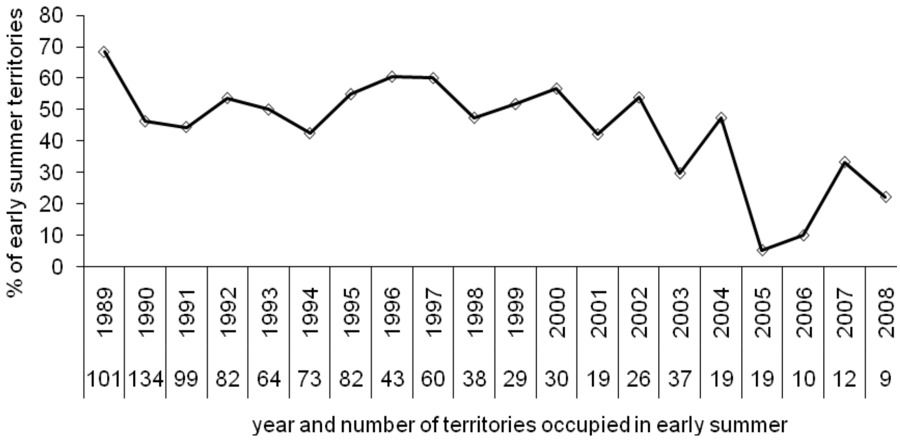
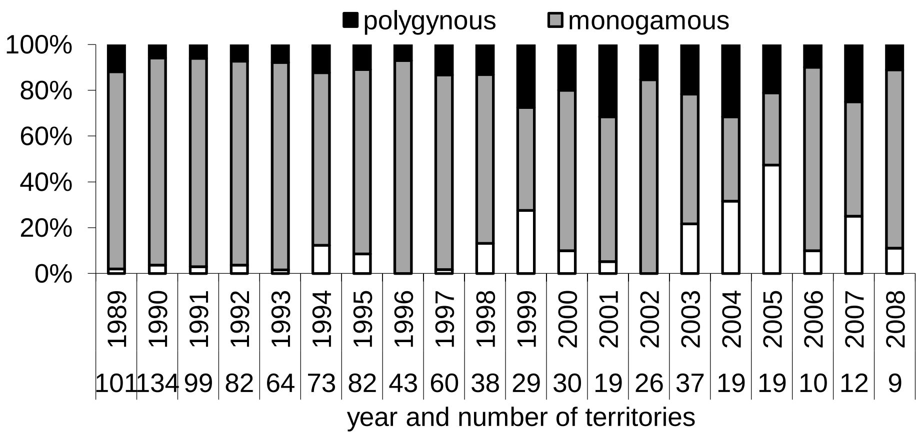


Figure 3.3. *Change in patterns of territory occupancy, 1989 – 2008.*

a) *Proportion of males holding territories into late summer*



b) *Proportions of males unmated, monogamous and polygynous in early summer*



c) *Distribution of females amongst polygynous male territories in early summer.*

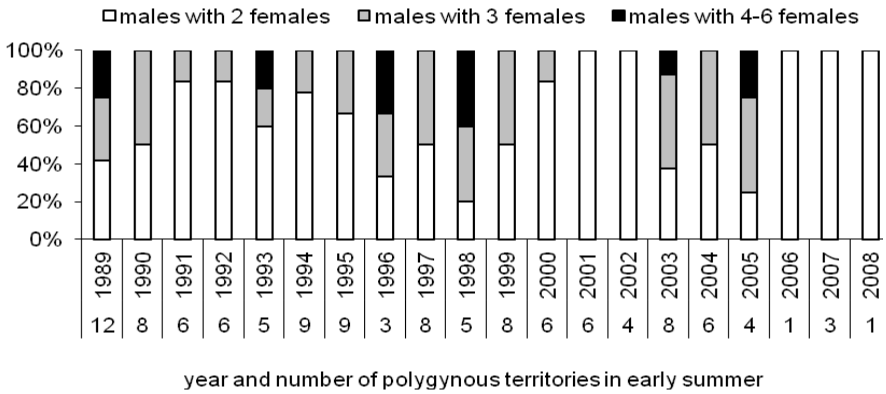
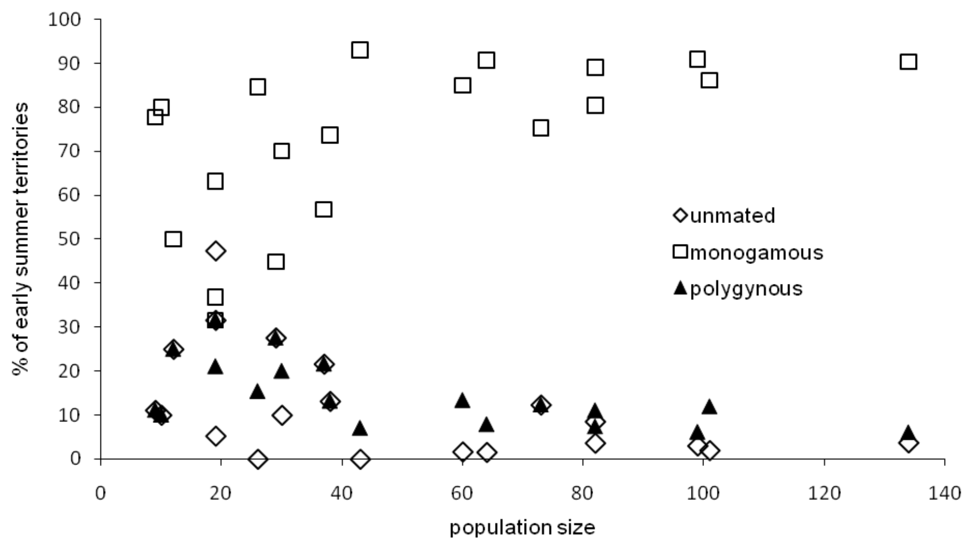


Figure 3.4. *Relationship between population size (territorial males) and:*

a) *Early-summer proportions of males mated polygynously, monogamously, and unmated.*



b) Sex ratio (number of females per male).

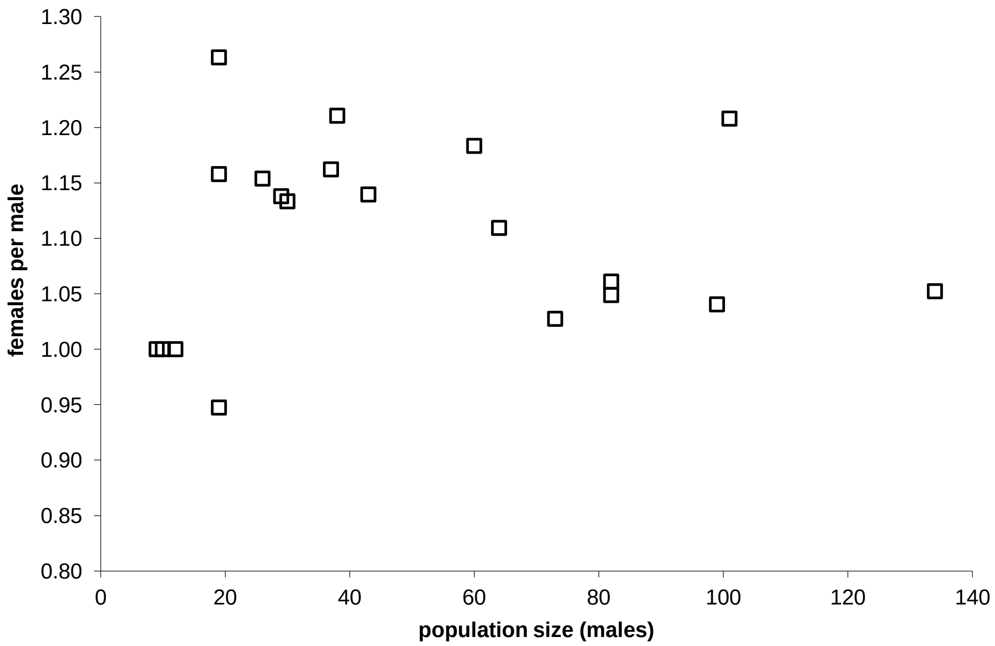
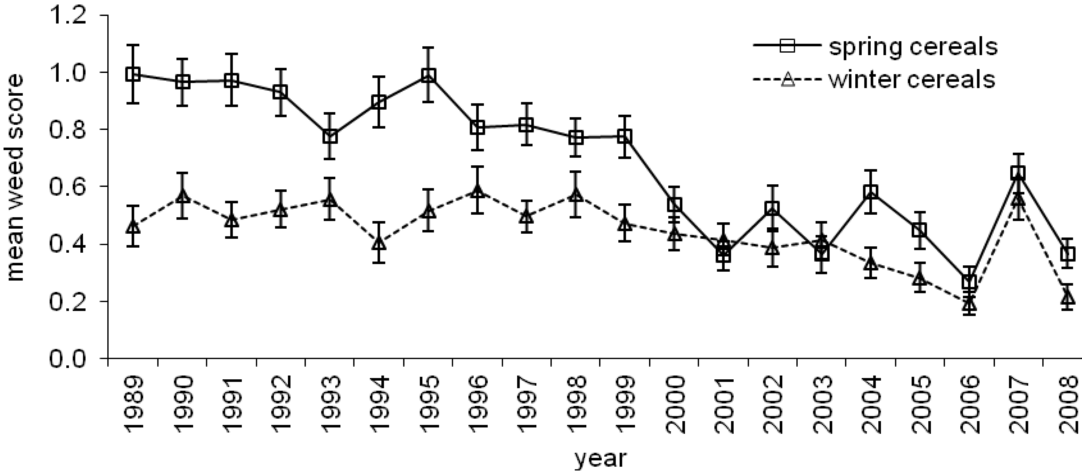


Figure 3.5. *Weed abundance in cereal fields, and differences between spring-sown and autumn-sown (winter) cereals, 1989–2008. Weed abundance in each field was estimated visually and scored from 0–7 for 0, 1–2, 2–5, 5–25, 25–50, 50–75, 75–99, and 100% cover, respectively, of ground surface covered by weeds or under-sown plants.*

a) *Spring and winter cereal fields in early summer (mean score ± se per year across 2254 spring cereal field/years and 2052 winter cereal field/years).*



b) *Spring and winter cereal fields in late summer (mean score ± se per year across 2254 spring cereal field/years and 2052 winter cereal field/years).*

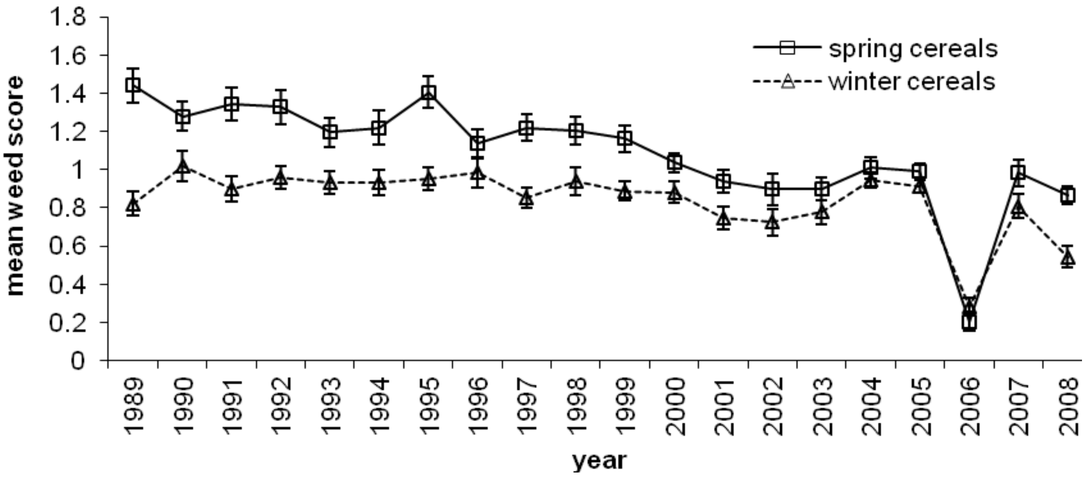


Fig. 3.5 cont.

c) *Frequency distribution of late-summer weed score categories among spring barley fields by year (% fields in each category per year across 2088 field/years).*

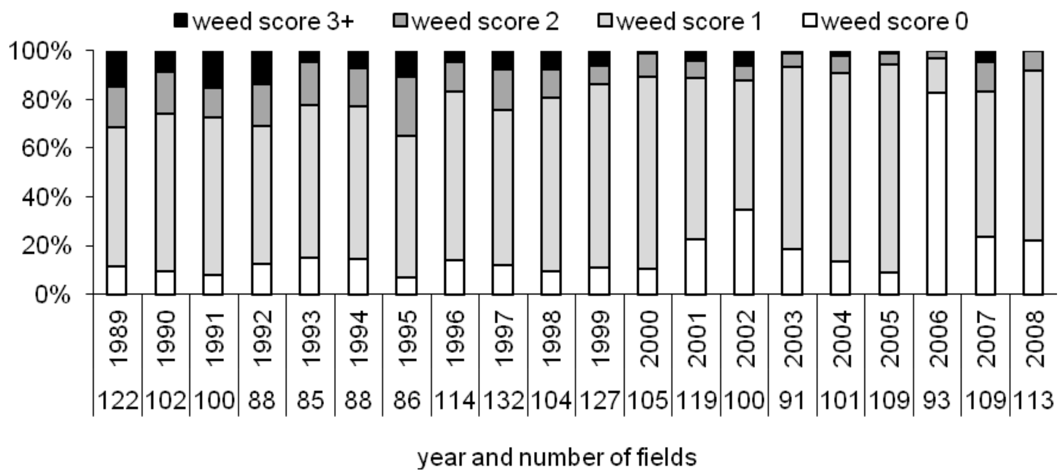


Figure 3.6. *Fitted relationship between habitat predictors and territory location (probability of a circle being a territory). Model-averaged parameter estimates from the final set of GLMMs that included year effects (see Table 3.5a) are used. For each plot, examples of real territories (one 'high quality' and one 'low quality', as defined by the other predictors) are used, whereby in (a) and (b) the predictor shown is increased from zero (its real value) whilst all other predictors are held constant at their actual values. In (c) all predictors are held constant at their actual values whilst year is increased from 1 to 20.*

a) *Weeds early. Origin points are a randomly selected 'high quality' territory whose habitat composition is 51% winter barley, 48% spring cereals, 1% other crops, with 0.40 boundary (113 m/ha), 0 wires (0 m/ha), and 0 weeds early (0% cover in weediest field), and a 'low quality' territory whose habitat composition is 93% winter wheat, with 0.09 boundary (26 m/ha), 0 wires (0 m/ha), and 0 weeds early (0% cover in weediest field). Year is set to 1 (1989) or 20 (2008) to show temporal variation in the weeds effect.*

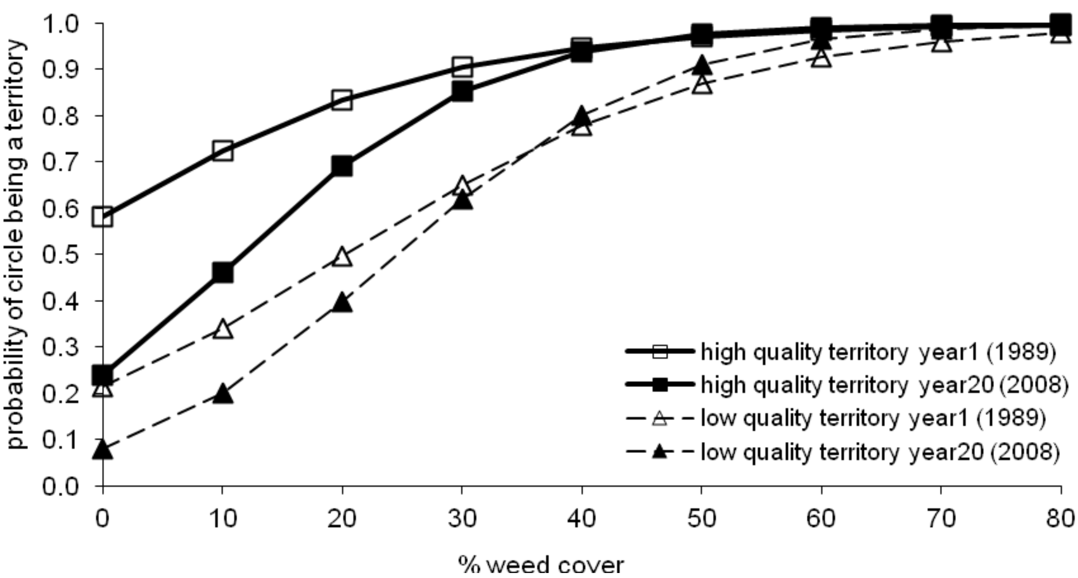


Fig. 3.6 cont.

b) *Wires. 'High quality' territory = 57% spring cereals, 31% winter barley, 12% fallow, 0.035 weeds early (3.5% cover in weediest field), 0.32 boundary (92 m/ha), and 0 wires (0 m/ha); 'low quality' territory = 83% other crops, 17% winter wheat, 0.015 weeds early (1.5% cover in weediest field), 0.26 boundary (74 m/ha), and 0 wires (0 m/ha). Year is set to 1 (1989) or 20 (2008) to show temporal variation in wires effect.*

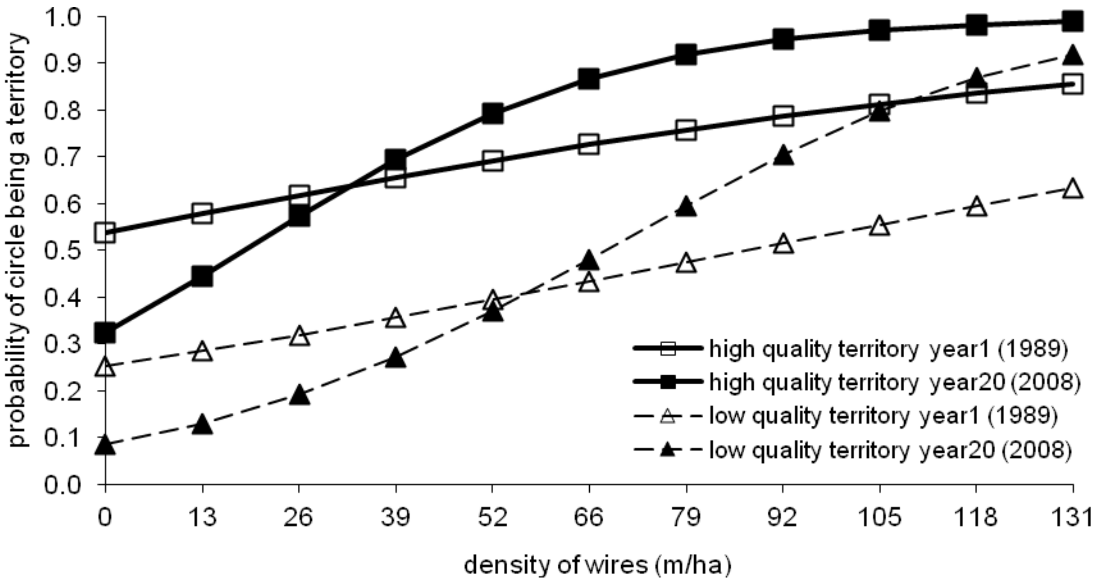
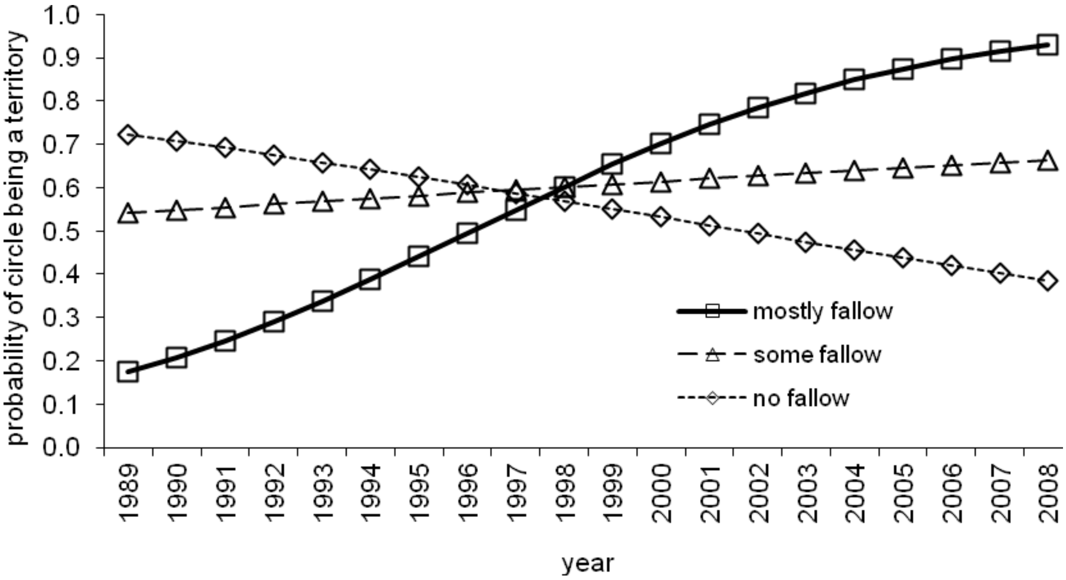


Fig. 3.6 cont.

c) *Fallow. Mostly fallow = 99% fallow, 1% spring cereals, 0.15 weeds early (15% cover in weediest field), 0.22 boundary (63 m/ha), and 0 wires (0 m/ha); some fallow = 37% fallow, 63% spring cereals, 0.15 weeds early (15% cover in weediest field), 0.32 boundary (92 m/ha), and 0 wires (0 m/ha); no fallow = 93% spring cereals, 0.15 weeds early (15% cover in weediest field), 0.32 boundary (92 m/ha), and 0 wires (0 m/ha).*



CHAPTER 4. HABITAT SELECTION BY FEMALES FOR NESTING

4.1. Introduction

Selecting a nest site is one of the most important decisions a bird has to make. During laying and incubation, both adult and eggs are vulnerable to predators, whilst chicks that remain in the nest are also at risk prior to fledging. Ground-nesting birds are especially vulnerable because they and their nests are accessible to a wider range of predators than arboreal nesters, including mammals such as foxes and ground-dwelling mustelids and rodents (e.g. Weidinger 2002). Anti-predator adaptations include colonial nesting, where defence in numbers can drive away predators, avoidance by selecting inaccessible nest sites such as islands, rock ledges or holes, and nest camouflage or concealment within vegetation. The nest protection tactic used largely determines the preferred nesting habitat of a species, and nest site selection is often a trade-off between concealment and keeping potential predators in view (Gotmark *et al*. 1995). Some nest on bare ground or in short vegetation because this allows early visual detection of an approaching predator, enabling the incubating bird to leave the nest in good time, and to then lure the predator away using distraction. Examples of UK farmland species that use this strategy include Lapwing, Stone Curlew and Skylark (Wilson *et al*. 1997, Green *et al*. 2000, Sheldon *et al*. 2005). Others such as Grey Partridge and Corncrake rely on plumage crypsis, sitting tight to avoid detection as a predator passes, so they nest in dense vegetation that offers good concealment (Rands 1986, Green & Stowe 1993). Incubating grouse are even known to lower their heart and breathing rates when a predator approaches, giving off less smell, noise and movement (Watson & Moss 2008).

Like most passerines, Corn Buntings are territorial during the breeding season, and rely on a combination of camouflage and concealment as the best form of nest protection. Females, which do all of the incubation, tend to sit tight, flushing at the last moment when approached by humans (e.g. Macdonald 1965, Brickle 1998). Therefore, they may prefer tall, dense swards that give greater visual concealment from predators (Evans 2004).

In the previous chapter, I showed that territorial Corn Buntings were strongly associated with particular crop types, and that these changed as the breeding season progressed. I hypothesised that selection of nesting habitat by females could be responsible for these associations and seasonal changes. In particular, I suggested that strong associations with weeds, cereals and forage grasses were due to their attractiveness to nesting females because they give tall, dense swards with good nest concealment.

In this chapter, I support these statements by presenting data on nest site selection by female Corn Buntings from field studies across several farms and years. The main aims were to answer the following questions: (1) In eastern Scotland, when and where do Corn Buntings nest? (2) How does crop use for nesting vary seasonally? (3) Which sward characteristics best explain field use by nesting Corn Buntings? (4) Can changes in sward structure explain the seasonal pattern of crop use for nesting?

4.2. Methods

4.2.1. Study sites

We[4] monitored Corn Bunting nesting activity on 32 farms during 2004–2009, varying annually from eight in 2004 to 23 in 2007, in four areas of eastern Scotland (Aberdeenshire, Angus, Fife and Inverness-shire – Fig. 4.1). All fields (contiguous patches of the same crop type) were mapped in each year on all study farms and their surroundings (250 m buffer). Land use was mainly a combination of autumn- and spring-sown cereals, oilseed rape, vegetables (root crops and legumes), forage grasses mown for silage or hay, pastures grazed by beef cattle and sheep, rough grass and set-aside (Table 4.1). Some spring-sown cereals (1.1–4.1% pa) were left unharvested as agri-environment scheme measures to provide over-winter seed for birds, and others (0–2.9% pa) were sown as a cereal-legume mixture for arable silage.

4.2.2. Nest monitoring

Each farm was visited at least once per week from May to early September, and repeated observations of territorial males led to all females on mosts farms being located. Corn Buntings are often polygynous, so even in territories where a female and nest had already been located, repeated checks were made for additional females. On most farms, nesting activity in all territories was monitored. Where high territory densities made this impossible (three farms), monitoring focused on territories that included forage grasses, as part of an experimental study to monitor the effect of delayed mowing on nesting success (see Chapter

[4] Several colleagues helped with this study. I designed all aspects of monitoring and fieldwork methods, but data collection was shared with Hywel Maggs, Amanda Biggins, Ken Bruce, Alan Bull, Steven Coyne, Richard Firmin, Clive McKay, John McMahon and Adam Watson. I personally undertook monitoring on ten of the 32 farms. I also designed and carried out all statistical analyses and the write-up of this chapter, with advice from my supervisors Jeremy Wilson and John Deag.

5). Despite this restrictive monitoring applying to farms that held up to 30% of territories in our overall study population, such territories included other available nesting habitats, so it was unlikely to have biased our results. Prolonged observations of female behaviour (nest building, entering or leaving a nest, or feeding chicks) was used to locate nests, with checks made subsequently every three to seven days until the nest failed or chicks fledged. Nests themselves were not visited (or only once to ring the chicks), to reduce the risk of causing predations or desertions. Instead, behavioural observations of the nesting 'pair' were used to determine the status of the nest (eggs, chicks, fledged or failed). Nests were easiest to locate at the building or chick stage, because females were very conspicuous when carrying straw or dried grass during nest building, and food when provisioning chicks. When incubating, a female may leave the nest only once per hour, so nests at the egg stage were more difficult to detect. We detected such nests by close observation of the male, who would join the female when she left the nest to feed. The exact nest location could be found when the female went back to resume incubation. Inevitably, some nests failed or fledged before they could be found, but we consider it likely that approximately 70–80% of nesting attempts within monitored territories were located.

For each nest, I used observation dates of females incubating clutches or feeding broods to estimate the first egg date (FED), based on the mid-point of possible date ranges. I assumed an incubation period of 12 days beginning on the laying day of the final egg, a laying period of one egg per day, and clutch size of four. Clutch size was based on a mean clutch/brood size of 3.31 from 86 nests visited to ring chicks, rounded up to four in the analyses to account for likely partial clutch/brood reductions prior to nest visits (see Table 5.3 in Chapter 5 for comparison with clutch and brood sizes recorded in other UK studies). I used a brood period (chicks in the nest) of 9–13 days (Snow and Perrins 1998), and assumed chicks were capable of flight at 15 days old. On occasions when nests were visited to ring broods (n = 83), and an estimate of chick age allowed back-calculation of FED. Thus, for a female observed nest building on 1 June, incubating on 9 June, and feeding chicks in the nest on 20 June:

Using nest building and incubation observations, FED range = 1–6 June
Back-calculating from chick feeding observation, FED range = 23 May–5 June
Combining the two, FED range = 1–5 June. Estimated FED = mid-point = 3 June

4.2.3. Vegetation sward measurements

To determine the fine-scale habitat requirements of nesting Corn Buntings, in 2004–08 vegetation sward characteristics of crops were measured. Crop swards were measured 1–3 times (165 once, 215 twice and 72 three times) during May – August (n = 266 May to mid-June, 284 mid-June to mid-July and 264 mid-July to August) in a sample of forage grass and cereal fields on each farm (FG = 89 fields, SC = 238, AC = 85 – see Table 4.1 for definitions of abbreviations). A small sample of fields with other crop types were also measured (VEG = 7 fields, ROU = 18, PAS = 12, RES = 3). In most cases, all sward measurements across a farm were taken on the same day. Although the timing of sward measurements did not always coincide with field selection for nesting by females (i.e. nest building), a plot of measurement date against FED of nests within territories intersecting the measured field indicated that the sampling regime gave a good general approximation of sward conditions during nest building (Fig. 4.2).

In 2004–05, ten sampling points per field were used, but to reduce the time taken to measure each sward, and allow a larger sample of fields, this was reduced to five in later years. At each sampling point, three measurements of sward height (cm) and one each of weed score and sward density at ground level (an index of nest concealment afforded by stems, leaves, and flowering heads of the crop itself, and by weeds) were recorded, using a sward stick (1 m cane marked at 10 cm intervals with coloured tape). Dicotyledonous weeds (i.e. all non-crop plants, excluding grasses) were recorded on a scale of zero to ten, according to the number of 10 cm sections of the sward stick that had a weed plant within 5 cm when the stick was laid flat on the ground. Sward density was scored on a scale of zero to ten, based on the number of coloured tape markers that were partially or wholly visible from above the crop, with the sward stick laid flat on the ground.

For example, the score was zero if none of the markers were visible (representing maximum nest concealment), and ten if all of the markers were visible (minimum nest concealment). To aid later interpretation of results, the inverse of this score was used as the sward density score in analyses. For cereals, the average stage of crop development within a field was also recorded on each visit. Scores were based on the Zadoks system, which uses a detailed two-digit code from 00–99 that can be applied to any cereal type (Simmons *et al*. 1995). Here, I used the first digit only (0–9 scale, with 0.5 divisions for crops that transcended two stages) to give the principal growth stage of the crop, where: 0 = germination; 1 = seedling development; 2 = tillering; 3 = stem elongation; 4 = boot (growing head enclosed by flag leaf sheath); 5 = head emergence; 6 = flowering; 7 = milk development in kernel; 8 = dough development in kernel; 9 = ripening kernel.

4.2.4. Measuring nesting habitat availability

Habitat availability for nesting females was determined by land use within the territory of the male to which she was paired. Although many territories were in approximately the same place in subsequent years, boundaries often moved and new territories became occupied, partly as a response to between-year changes in field use. Territory abandonment also occurred due to presumed mortality of philopatric adults. Therefore, a new territory map was drawn in each year. The centre of each male's territory was taken to be the most frequently used song-post, assessed from his mapped position on each farm visit. Each male's main song-post was plotted onto a 1:25 000 digital map, together with the location of each nest. Because 95% of nests lay within 150 m of the male's main song-post (Fig. 4.3a), a circle of 150 m radius drawn around the main song-post of each male was used to represent a nominal territory (area = 7.02 ha). Where two or more territory circles overlapped, their boundary was redrawn along the midpoint of the area of overlap. Annual territory and land use maps were overlain to measure the area of each field within a territory. The area of each crop type within a territory (from hereon referred to as a *habitat patch*) was then calculated by summing these field areas. All digital mapping and measurements of areas and distances used MapInfo Professional version 6.

4.2.5. Data analysis

4.2.5.1. Crop selection for nesting

To determine the seasonal pattern of crop selection by nesting female Corn Buntings, I modelled the probability of habitat patch selection (response variable *habitat patch* = selected, 1 or not selected, 0) as a function of *crop type* (fixed effect categorical variable with 8 levels – see Table 4.1) and the interaction term *crop type|FED* of the nest (covariate). For this, I used a GLMM (generalized linear mixed model, using the SAS GLIMMIX procedure) with a logit-link function and binomial error distribution, and specified *ln size* (ha) of the habitat patch as an 'offset' variable. To control for the repeated measures structure of the data (>1 habitat patch per nest, >1 nest per territory, and multiple territories per farm), I fitted *patchID* (a unique identifier for each habitat patch, n = 1054) as a random categorical effect, nested within *year* to control for temporal auto-correlation and annual weather effects on crop growth and timing of nesting. Denominator degrees of freedom for tests of fixed effects (see Chapter 2) were calculated using the Kenward-Roger method (Littell *et al*. 1996).

4.2.5.2. Sward characteristics selected for nesting

Because of a high degree of inter-correlation between sward variables (Table 4.2), for three categories of field type (all fields, cereals, and grass), I used a PCA (principle component analysis) to reduce the original set of variables to two independent linear combinations of variables (*PCA 1* and *PCA 2*; Table 4.3). Across all fields, *PCA 1* represented a transition from short swards with low density but high weed scores to tall dense swards with low weed scores, and *PCA 2* a gradient from sparse swards with little weed cover to dense weedy swards. Similarly, in cereals, *PCA 1* represented a gradient from short sparse swards with low Zadok's scores, to tall dense mature crops, and *PCA 2* represented a transition from sparse swards with little weed cover to dense weedy swards. In grass fields, *PCA 1* represented a transition from short sparse to tall dense swards, and *PCA 2* reflected weed score.

Using a GLMM with a logit-link function and binomial error distribution, I then modelled the probability of field use for nesting (response variable = number of nests within a field divided by the number of nesting attempts recorded within territories intersecting that field) as a function of the sward variables *PCA 1* and *PCA 2*. Sward measurements were within the same date range as the FED of these nests (Fig. 4.2). To control for repeated measurements from the same field within each year, and for between-year auto-correlation, I fitted *period* (categorical 3-level effect – early, mid, and late season, corresponding to date of sward measurement) and *fieldID* (a unique identifier for each field, n = 213) as random effects, both nested within *year* (categorical 5-level effect). First, I applied this modelling approach across all fields, before analysing the two main sward types, cereals and grasses, separately.

4.2.5.3. Sward characteristics of crop types

To show how sward characteristics varied during the growing season, and how this may have influenced the seasonal pattern of crop use by nesting Corn Buntings, scatter plots of *height*, *density* and *Zadok's score* (cereals only) against date were constructed for each of the main crop types measured (AC, SC, FG, ROU). This was done using the SAS GPLOT procedure, specifying a regression analysis with 95% confidence limits. For cereals and FG, I also plotted *height* against *density*, to show how this relationship varied between cereal and grass crops. Finally, for each of the three main cereal types grown in eastern Scotland (spring barley, winter barley and winter wheat), I plotted *Zadok's score* against date.

4.3. Results

Across all years, 580 nests were located on the 32 farms (Table 4.1). Most nesting attempts (37%) were discovered at the building stage, with 26% found at the egg stage and 32% at the chick stage. The remaining 5% were found after young had recently left the nest, but were still unable to fly. The mean distance from a nest to the male's main song-post was 68 m (sd = 42 m, range = 6 – 265 m), and 95% of nests were within 150 m, 43% within 50 m, and one-fifth >100 m from the main song-post (Fig. 4.3a). The mean distance from a nest to the nearest field boundary was 33 m (sd = 28 m, range = 0 – 230 m). Most nests (66%) were within 30 m, but only 17% were <10 m, leaving 49% between 10 m and 30 m of a boundary (Fig. 4.3b). Just 14% of nests were >50 m from a field boundary.

Clutches were laid between 17 May and 16 August, peaking in the second and third weeks of June (Fig. 4.4a). The date of the earliest and latest clutch varied between years as follows: 19 May and 12 August 2004; 24 May and 4 August 2005; 4 June and 29 July 2006; 17 May and 10 August 2007; 22 May and 16 August 2008; 1 June and 2 August 2009. The latest recorded active nest, with 13-day old chicks, was on 9 September 2004. Nesting began up to ten days later and finished up to 15 days earlier in Fife/Angus than in Aberdeenshire/Inverness-shire (Fig. 4.4b). At least 31 second clutches followed successful fledging of a first brood (Table 4.4), where the identity of females were known from colour-rings or observations of nest building whilst still feeding fledglings. The period between fledging of the first brood and initiation of the next clutch was typically less than two weeks (mean = 12 days, sd = 8 days). No birds were recorded laying clutches following two successful, fledged broods.

4.3.1. Crop selection for nesting

Most nests were in spring-sown cereals (288 nests; 49.7%), forage grasses (129; 22.2%), autumn-sown cereals (67; 11.6%), and other types of grass such as non-rotational set-aside, field margins and newly sown grass (66; 11.4%). However, crop types used for nesting varied seasonally (Fig. 4.5). Also, due to land use differences between regions (less grass and more vegetables in Fife/Angus – see Table 4.1), a much higher proportion of the nests in

Fife/Angus were in cereals and vegetables, with, by contrast to Aberdeenshire/Inverness-shire, very few nests in grass (Fig. 4.5).

Overall, nests were distributed non-randomly with respect to the availability of each crop type (F_{1513} = 9.44, *P* <0.0001). There were significant *crop type|FED* interactions for five crops, with probability of a female initiating a nest in FG and PAS declining as the season progressed, and of nesting in RES, SC and VEG increasing as the season progressed (Table 4.5, Fig. 4.6). In early season (pre- 10 June), the probability of field use for nesting was higher in FG than all other crops, whilst from mid-June onwards, SC had the highest probability of use, except for the rare crop type RES. By mid-July, amongst the crop types widespread and most used for nesting overall, AC and FG had a very low probability of use, less than half of the probability of use in ROU, and below a third of that in SC.

4.3.2. Sward characteristics selected for nesting

Across all field types, probability of field use for nesting by Corn Buntings was positively associated with both of the principle component axes describing sward characteristics (Table 4.6), but more strongly so with *PCA 2* (sward density and weed score) than *PCA 1* (sward density and height). In cereals, field use was strongly positively associated with *PCA 2* (sward density and weeds), but the positive relationship with *PCA 1* (sward height and crop maturity) was non-significant. In grass fields, use for nesting was positively associated with *PCA 1* (sward height and density), but not with *PCA 2* (weeds).

Figure 4.7 shows the modelled relationship between sward characteristics and probability of field use for nesting. In cereal fields, with an increase in the weed score from 1 to 9.5 and sward density score from 2.5 to 8.5 (an increase in the value of *PCA 2* from -1 to 2), there was a fourfold increase in the probability of field use for nesting for any given crop height and stage of maturity (Fig. 4.7a). In grass fields, the probability of field use doubled with an increase in sward height from 20 cm to 60 cm, and an associated increase in sward density from 2 to 9 (an increase in the value of *PCA 1* from -1.5 to 1; Fig 4.7b).

4.3.3. Sward characteristics of crop types

Scatter plots showing the seasonal variation in *height* and *density* of FG, AC, SC and ROU are in Fig. 4.8. During May to early June, FG (20–70 cm) and AC (50–100 cm) were the tallest crops, when most SC swards were less than 30 cm tall. Throughout June to mid July, SC sward heights increased rapidly, reaching 70–100 cm before showing a slight decline from early August. AC swards remained at approximately 70–100 cm throughout, and FG swards (40–100 cm) declined from mid–late June due to harvesting of some crops, and swards partially collapsing in others. ROU sward heights varied considerably (10–90 cm), but tended to increase slowly throughout the season.

For all crop types, *density* values varied tremendously between individual fields. However, few SC crops had a sward density value greater than 4 during May to early June, in contrast to FG and AC (Fig. 4.8). Seasonal changes in sward density generally reflected changes in sward height, with a rapid increase in SC from May to early–mid July (0–1 to 5–6) followed, overall, by a slight decline. FG sward densities increased rapidly during May to mid June, reaching average values of 7–8, the highest of any crop type, before declining due to crop harvesting. AC average sward density values declined from a peak of 5–7 in early July, despite crop heights remaining unchanged. ROU values showed tremendous variation throughout the season, but tended to increase until mid–late July. Sward density values increased more rapidly with sward height in FG than cereals, such that an increase from 20 cm to 40 cm to 60 cm would result in sward density increasing from 2 to 6 to 8 in FG, and from 1.5 to 4 to 4.5 in cereals (Fig. 4.9). Sward density values in cereals tended to level off at 4–5 for heights of 40–70 cm, with further increases as heights exceeded 70 cm.

Figure 4.10 shows the seasonal pattern of *Zadok's score* for the three main cereal types grown in eastern Scotland. Autumn-sown barley matured earliest, with average values of 5–6 during May to early June, compared with 2–5 for autumn-sown wheat and 1–3 for spring-sown barley. A Zadok's score of 7 indicates the development of green, part-ripe grains, and this stage was typically reached earliest in autumn barley (mid June – early July), followed by autumn wheat (late June – July) and spring barley (mid–late July).

4.4. Discussion

Corn Buntings laid clutches from mid May to mid August, the great majority (89%) in June and July. Cereals and grasses accounted for 94% of nests, but there was seasonal variation in crop use. In May and early June, almost two-thirds of nests were in forage grasses and autumn-sown cereals, despite these fields covering just 30% of the study area. As the season progressed, selection of forage grasses declined (probability of field use halved between late May and late June) whilst that of spring cereals, vegetables and re-seeded grass increased. The likelihood of a Corn Bunting nesting in a spring cereal field doubled between late May and mid-July, and these fields (one-third of the study area) held two-thirds of nests initiated in mid July – August. Use of set-aside and rough grass did not change during the season, and these habitats held 8% of nests in each period despite covering just 4% of the area surveyed.

Sward structure, and changes due to growth in some crop types and harvesting of others, largely explained seasonal patterns of crop use for nesting. Corn Buntings preferred to nest in fields with dense swards, and in cereals, nesting was more strongly associated with high scores of sward density and weed abundance than with sward height and the stage of crop development. In grasses, however, field use for nesting was positively associated with sward height and density, but there was no association with weeds.

4.4.1. Nest location in relation to field boundary and male song-post

Eighty percent of nests were within 100 m of the male's main song-post, and two-thirds within 30 m of a field boundary. This information is useful to conservation practitioners because it helps to predict where nests are likely to be, aiding effective targeting of management prescriptions. Although field-boundary features themselves were rarely used (just seven nests), 17% of nests were within 10 m of a field boundary. UK agri-environment schemes include several field-boundary management options, including the provision of grass margins around arable fields. This type of management could potentially provide nest sites for ground-nesting birds safe from within-field agricultural operations such as crop harvesting. Our data suggest that grass margins would need to be 10–20 m wide to attract nesting Corn Buntings with any regularity, but in the current and recent Scottish national agri-environment schemes, the maximum width of the main option ("Management of grass margin or beetle-bank in arable fields") is 6 m (SEERAD 2003, Scottish Government 2010a). In our study, only 5% of nests were within 6 m of a field boundary, although in southern England, Brickle & Harper (2000) recorded 8% of 120 Corn Bunting nests in grass margins mostly under 3 m wide. However, one recent study (Morris & Gilroy 2008) showed that nest predation rates of ground-nesting birds (Skylarks and Yellow Wagtails) by mammals were higher when nests were closer to field boundaries (although Skylark nest survival within grass margins was relatively high), so further research is necessary before recommending wide grass margins as a conservation management option for Corn Buntings.

4.4.2. Seasonal crop use and influence of sward structure

In early summer, Corn Buntings selected forage grasses and autumn-sown cereals because these crops offered the tallest and densest swards. By contrast, spring cereals were short and sparse at this time, offering little nest cover. Among grass fields, the probability of use of a sward 50–60 cm tall (typical of forage grasses in early June) was more than twice that of a sward less than 20 cm tall (typical of grazed pastures). However, farmland habitats can quickly change with crop growth and operations such as harvesting and spraying with pesticides. The rapid growth of spring cereals during June – mid July explained their increased use by nesting Corn Buntings, with mean density scores and sward heights doubling (from 3 to 6, and 30 to 70 cm, respectively) during this period as Zadok's scores increased from 3 (stem elongation) to 6 (flowering). At the same time, harvesting reduced the availability of forage grass crops. This involves mowing the grass to a height of *c.* 5 cm, effectively making fields unsuitable for nesting Corn Buntings and destroying any nests still active (see Chapter 5). By late season (mid July – mid August), few forage grass crops remained uncut, and harvesting of barley had also begun, thus reducing the availability of autumn cereals. Almost all nests initiated during this period were in spring-sown crops, mostly cereals but also vegetables and re-seeded grass, the latter being a strongly selected but scarce field type with high rates of nest losses to cutting to control weeds.

As far as I am aware, no other recent UK study has reported widespread nesting by Corn Buntings in forage grasses (Gillings & Watts 1997, Murphy 2000, Brickle & Harper 2002, Setchfield *et al*. 2012), perhaps because first silage cuts in more southern areas are usually completed in May, before birds have started to nest (Vickery *et al*. 2001). Overall, most nests are in cereals, although hay fields and other grass habitats such as set-aside are also used, and on rare occasions low bushes or shrubby vegetation. In eastern Scotland, a lack of early-summer cover in spring-sown cereals is partly responsible for Corn Buntings nesting in forage grasses, whilst in the Western Isles they mainly nest beneath hogweed *Heracleum sphondylium* plants in dune grassland, whose large leaves provide greater nest concealment than cereals sown in late spring and at low plant densities (Hartley *et al*. 1995).

The seasonal pattern of crop use for nesting reflected within-summer changes in habitat associations (from forage grasses and winter barley in early summer to spring cereals in late summer) by territorial males presented in Chapter 3. Mid-season shifts in breeding habitat association or territory locations in response to changes in vegetation sward structure have been reported in other species too, notably Skylark, Woodlark *Lullula arborea* and Yellow

Wagtail (Wilson *et al*. 1997, Chamberlain *et al*. 1999, Brambilla & Rubolini 2009, Gilroy *et al*. 2010). Like the Corn Bunting, all three species nest on the ground (two in crops) and rear multiple broods over several weeks or months, and such habitat switching may be commonplace in other species with similar breeding ecology. For example, at a finer scale, Yellowhammers switched from nesting on the ground amongst herbaceous vegetation to nesting in hedgerows as the season progressed and shrubby vegetation cover increased (Bradbury *et al*. 2000).

4.4.3. Influence of weeds on field selection for nesting

In the previous chapter, I determined that weed abundance was a strong predictor of male territory locations and polygyny. Here, I show that cereal field use for nesting was also associated with high weed scores. The probability of use was three to four times higher in a field with weed score 9.5 and density 8.5 than in a field with weed score 1 and density 2.5 (Fig. 4.7a). Weed plants host invertebrates that provide food for Corn Bunting chicks (Wilson *et al*. 1999), and in one study the main invertebrate groups taken (*Opiliones*, *Lepidoptera* larvae, *Symphyta* larvae and *Orthoptera*) were more abundant in crops with fewer pesticide applications (Brickle *et al*. 2000). Furthermore, Brickle *et al*. (2000) showed that when invertebrate food was abundant close to the nest, provisioning trips by parents were shorter, and chick condition and survival was higher. Fledglings can also remain within the crop close to the nest if food is readily available, potentially reducing their exposure to avian predators such as Sparrowhawks *Accipiter nisus*.

Weeds also provide nest concealment by forming a dense ground layer within crops, but only if the weed plants are allowed to mature. This was clear to see on one of our study farms, where in the absence of herbicide use, spring cereal fields with very little bare ground visible because of the dense weed cover attracted many nesting Corn Buntings. Although nest concealment does not always prevent predation, particularly if the main predator uses scent to locate nests (e.g. Davis 2005, Colombelli-Negrel & Kleindorfer 2009, Schüttler *et al*. 2009), experimental studies by Weidinger (2002) showed that concealment and passive defence (against rodents) through nest attentiveness independently reduced nest predation in Yellowhammers. As well as providing visual concealment, it is possible that dense weeds could inhibit the movements of predators through a crop, and potentially conceal scent given off by the incubating female or chicks from olfactory predators.

4.4.4. Sward and crop maturity effects on timing of nesting

Of all UK farmland birds, Corn Buntings are one of the last to begin nesting. In eastern Scotland over six years, egg-laying in the earliest clutch began on 17 May – 4 June, and the latest clutch 29 July – 16 August. Although the main nesting period is similar to that reported in other UK studies (Table 4.7), the range of first egg dates is greater than for most. Interestingly, the egg-laying period in Aberdeenshire/Inverness-shire was 3–4 weeks longer than in Fife/Angus (Fig. 4.3). This may have been due land use differences between the two. Greater availability of forage grasses in Aberdeenshire/Inverness-shire may have allowed an earlier onset of nesting (by up to 10 days), whilst a larger proportion of cereals spring-sown in these regions, or perhaps managed less intensively, may have encouraged later nesting (by up to 15 days) than in Fife/Angus (Table 4.1). However, it is not clear why Corn Buntings did not nest earlier in autumn-sown cereals in Fife/Angus, given their widespread availability.

One possibility may be a lack of ground cover provided by weeds in autumn-sown cereals. For example, Field *et al*. (2007) showed that cover provided by residue on the soil surface from the previous crop allowed Skylarks to nest 25 days earlier in autumn-sown wheat fields established by minimum tillage than in those prepared with conventional ploughing. In our study, weed plants recorded within autumn-sown cereals were often too small (less than 2 cm tall) to provide a dense ground layer, and their small size was likely due to suppression of growth through size-asymmetric competition for resources such as sunlight by the larger crop plants (Schwinning & Weiner 1998), combined with the effects of herbicide use. It is likely, therefore, that sward height and crop architecture had a much greater influence than weed cover on sward density scores recorded for autumn cereals. This probably explains the negative correlation in our data between sward density and weed score in cereals, particularly as my weed scoring method did not include a measure of the size of weed plants, or percentage ground cover (unlike the method used in Chapter 3).

If nest concealment in autumn cereals does depend largely on the architecture of the crop itself, cereal type may have an effect. In Fife/Angus, the main autumn-sown cereal was wheat (72% of AC across all years), whereas in Aberdeenshire/Inverness-shire it was barley (58% of AC). Wheat matures more slowly than barley (Fig. 4.10), and tends to have a more open sward structure. Barley seed heads droop as they mature (from Zadok's score 6), forming a canopy, whereas wheat heads do not. Therefore, sufficient cover to attract nesting

Corn Buntings into autumn wheat may come at a later crop development stage than in autumn barley.

However, factors other than vegetation cover may dictate the onset of nesting. These include food availability for females gaining body condition prior to egg-laying, or for provisioning chicks later on. Brickle & Harper (2002) suggested the former, with Corn Buntings nesting once unripe grain became available (Zadok's score 7), but could not determine whether birds were responding to availability of the grain itself, or some correlate of it such as invertebrate food. Dietary studies showed that female Corn Buntings did eat unripe grain prior to egg-laying, but also took other seeds and invertebrates, including large numbers of click beetles which were rare in the species' diet at other times of year on the same site (Brickle & Harper 2002). In our study, unripe grains were typically available from mid-June in autumn barley, late June in autumn wheat and mid-July in spring barley. If the main cue for Corn Buntings to begin nesting is availability of unripe grain, this should not have occurred until the second half of June, but the earliest nests were one month prior to this.

Whilst it is possible that females required other specific food items, such as invertebrates, to attain breeding condition, there is another possible link between the timing of nesting and stage of cereal crop development. Many Corn Buntings initiated nesting attempts in spring cereals 2–3 weeks before these crops were bearing grains. This perhaps supports the idea that females time the onset of nesting so that chick hatching coincides with maximum availability of chick-food, including part-ripe grain. Unripe grain can be an important food for bunting chicks and fledglings (Watson 1992, Stoate *et al*. 1998, Brickle & Harper 1999), and during chick-provisioning watches, I observed female Corn Buntings flying up to 1 km to collect grain from the nearest autumn barley field. Corn Buntings on the Western Isles, however, rarely fed seeds of any kind to their chicks, perhaps because there was little need to in a non-intensive farming system where invertebrate abundance was high (Hartley & Quicke 1994). Indeed, one recent study in northeast Scotland showed grain to be an inferior food for Yellowhammer nestlings relative to protein-rich invertebrates (Douglas *et al*. in press). However, it may form a vital food source on intensively managed farms with few chick-food invertebrates, or during cold wet periods when insect activity and availability is restricted.

Ensuring that the chick period coincides with peak food availability may be especially important in polygynous species such as Corn Bunting, where males offer little parental care (Hartley & Shepherd 1994a), and Yom-Tov (1992b) reported a tendency for later nesting among polygynous passerines than in related monogamous species. Of 29 nests watched in

2007, I observed the male provisioning chicks at only 11 of these (38%), and at nests with male help, his contribution was just 32% of total feeds (17% across all nests, including those with no male help). In the Western Isles, Hartley & Shepherd (1994a) also recorded low provisioning rates amongst male Corn Buntings, although this increased as chicks became older, from one-fifth of males provisioning broods less than three days of age, and contributing 3% of total feeds, to approximately 80% of males feeding broods more than eight days old, providing 22% of total feeds.

4.4.5. Implications for breeding productivity and population trends

For multiple-brooded species, the availability of suitable nesting habitat over the entire breeding season is critical. A reduction in the number of broods reared annually per pair may have contributed to the population declines of several farmland bird species within the UK (Siriwardena *et al*. 2000a). These include Skylark, whose national population decline is partly due to increased height, density and homogeneity of crop vegetation with a switch to autumn-sown cereals, preventing second nesting attempts or encouraging birds to nest in sub-optimal sites with high predation risk (Wilson *et al*. 1997, Donald *et al*. 2002). Similar effects may be driving population declines in Yellow Wagtail on arable farmland (Morris & Gilroy 2008, Gilroy *et al.* 2011), whilst examples of reduced vegetation sward heterogeneity limiting nesting opportunities for species using other habitats are given in Wilson *et al*. (2005).

For Corn Bunting, this study and others have shown that where early and late-season nesting habitats are available, the breeding season can last from May–August, easily long enough to rear two broods. However, late-summer nesting is now rare in much of the UK, and very few Corn Buntings attempt to rear a second brood, let alone succeed in doing so. In the present study, double brooding did occur, although it was less frequent in Fife/Angus where the breeding season was shorter. Some females successfully raised two broods in the same crop type, often re-nesting within the same field, but a diversity of crop types usually offers greater opportunities for double brooding.

The timing of sowing of cereals and their subsequent harvest therefore has important implications for nesting Corn Buntings. This is because in the many areas where autumn-sown cereals predominate, very little suitable nesting habitat remains following harvesting of the cereal crops in early July. In their Sussex study, Brickle & Harper (2002) found that following harvest of all cereal crops and cutting of set-aside fields by mid August, only 3%

of the study site remained suitable (tall grassy habitat) for nesting Corn Buntings, compared with 56% in early July. Today, 80–90% of cereals in England are autumn-sown, compared with 70–80% spring-sown in England and Wales in the 1960s (Wilson *et al*. 2009). This latter figure is typical of the present-day situation in eastern Scotland, where almost two-thirds of cereals on our study farms were spring-sown, rising to 74% across just the Aberdeenshire/Inverness-shire farms. Thus, lack of late-season nesting habitat does not appear to be a problem in eastern Scotland, although the quality of that habitat (weed abundance) may influence the likelihood of late nesting (Setchfield *et al*. 2012).

At the other end of the breeding season, onset of nesting may be constrained by lack of availability of dense swards with sufficient nest cover, particularly where cereal fields have few weeds. In Aberdeenshire/Inverness-shire, the widespread availability of tall, dense uncut forage grass crops in mid-May to mid-June allows Corn Buntings to nest earlier than would otherwise be possible, especially on farms where all of the cereals are spring-sown. However, whilst dense crop swards may give good nest concealment from predators, nesting within agricultural crops brings other threats. Farming operations such as ploughing, sowing, spraying fertilisers or pesticides, and harvesting often cause nest failure, either through direct destruction, disturbance leading to desertion, or by increasing their exposure to predators (Newton 2004). In a sample of 496 nest records from across the UK during 1948–1991, Crick *et al*. (1994) showed that the proportion of Corn Bunting nest failures due to agricultural operations increased from 10% pre- 1970 to 43% post- 1970. By contrast, in Yellowhammer and Reed Bunting that nest mainly in non-cropped habitats, there was no such change, with agricultural operations accounting for 17–19% of failures in both periods.

In eastern Scotland, few nesting attempts in forage grasses are successful because most fields are mown during June or July before broods reach fledging age. Only those nests initiated early, or located in fields cut late, survive to fledging. Nests in set-aside and re-seeded grass are also vulnerable to destruction by mowing (Watson & Rae 1997a,b). I return to this subject in detail in Chapter 5, where I analyse nest success rates and assess the effectiveness of an agri-environment measure to improve breeding success in forage grass fields.

Table 4.1. *Area (ha) surveyed and land use composition (%) by year and region, and percentage of nests (across all years per region) by crop type. All crop types except for OTH were used in a GLMM to determine crop selection for nesting (see section 4.2.5.1).*

Region	Year	Area (ha)	SC	AC	VEG	OSR	FG	PAS	ROU	RES	OTH	No. of nests
Aberdeenshire	2004	1674	32.7	15.5	2.5	3.9	15.5	17.6	4.6	1.0	6.8	67
	2005	2688	36.5	14.0	2.8	1.9	14.1	20.0	5.1	1.2	4.0	113
	2006	1081	38.0	3.1	4.0	0.1	11.9	31.3	9.6	0.9	1.0	31
	2007	2443	34.5	9.8	2.7	1.8	15.3	24.2	5.6	1.5	4.6	140
	2008	1142	40.9	10.0	5.1	0.5	13.7	19.9	3.0	0.0	2.3	40
	2009	1414	38.3	8.2	6.7	0.1	12.1	23.3	6.1	1.1	2.8	27
% nests			**48.6**	**6.9**	**2.9**	**0.2**	**25.4**	**2.9**	**10.0**	**2.4**	**0.7**	**418**
Inverness-shire	2006	493	18.0	18.2	7.5	10.9	17.8	21.2	1.1	1.4	5.0	10
	2007	455	17.5	26.1	5.8	7.6	16.4	19.1	2.2	2.6	2.6	14
	2008	455	25.1	22.8	6.2	8.4	13.7	14.4	3.5	0.0	3.1	13
% nests			**29.7**	**29.7**	**2.7**	**0.0**	**37.8**	**0.0**	**0.0**	**0.0**	**0.0**	**37**
Angus	2006	335	27.2	25.3	34.9	6.4	4.6	0.1	1.3	0.0	0.3	10
	2007	467	40.0	22.9	17.6	4.5	8.7	3.4	2.8	0.0	0.2	9
	2008	480	47.4	15.6	23.6	2.5	4.3	6.4	0.0	0.0	0.2	11
	2009	754	31.2	32.3	20.3	6.0	2.8	4.2	2.0	0.0	1.2	15
% nests			**68.9**	**15.6**	**0.0**	**0.0**	**15.6**	**0.0**	**0.0**	**na**	**0.0**	**40**
Fife	2006	1079	21.3	34.2	19.6	5.5	1.3	8.4	6.1	0.9	2.6	24
	2007	1012	19.4	37.9	17.4	5.2	1.6	11.8	3.3	0.6	2.9	13
	2008	1011	23.4	34.3	19.0	4.9	2.4	11.0	2.2	0.0	2.8	15
	2009	739	30.9	32.3	21.4	0.9	1.6	8.2	0.2	0.0	2.7	28
% nests			**53.8**	**25.0**	**13.8**	**2.5**	**2.5**	**0.0**	**1.3**	**1.3**	**0.0**	**80**

SC = spring-sown cereals (barley Hordeum vulgare and oats Avena sativa), legume/cereal mixture mown for arable silage, and unharvested crops (spring-sown cereal/brassica mixture left unharvested for one or two years); AC = autumn-sown cereals (wheat Triticum aestivum, barley and oats); VEG = root vegetables (potatoes Solanum tuberosum, carrots Daucus carota, turnips Brassica rapa, and cabbages B. oleracea) and legumes (peas Pisum sativum, beans Vicia faba); OSR = oilseed rape B. napus; FG = forage grass mown for silage or hay; PAS = grazed pasture (sheep, cattle, pigs and horses); ROU = rough grass and set-aside (rotational or non-rotational); RES = newly-sown grass; OTH = other habitats (e.g. soft fruit, daffodils Narcissus pseudonarcissus, scrub, wetland, woodland, farm buildings)

Table 4.2. *Correlation matrix of sward variables, showing Spearman correlation coefficients (r_s values). n = total number of field measurements, summed across all years.* ***Bold*** *= statistically significant (P < 0.05) with a moderate degree of correlation (r_s > 0.200).*

		Density	*Weeds*	*Zadoks score*
All fields	*Height*	**0.451**	**-0.315**	
n=775	*Density*		-0.080	
Cereal fields	*Height*	**0.516**	**-0.415**	**0.670**
n=538	*Density*		**-0.213**	**0.285**
	Weeds			**-0.201**
Grass fields	*Height*	**0.620**	0.025	
n=191	*Density*		0.065	

Table 4.3. *Eigenvector coefficients from the principle component analyses on sward variables for* ***(a)*** *all fields,* ***(b)*** *cereals, and* ***(c)*** *grass fields. In each case, only the first two principal component axes are shown (and used in GLMMs), because combined they explained more than 75% of the variability in the data. Eigenvalues for each axis and the proportion of variance explained (R^2) are also given.*

Sward variable	**(a)** All fields		**(b)** Cereal fields		**(c)** Grass fields	
	PCA 1	PCA 2	PCA 1	PCA 2	PCA 1	PCA 2
Height	0.70	0.02	0.63	-0.01	0.70	-0.09
Density	0.57	0.59	0.40	0.67	0.70	-0.07
Weeds	-0.44	0.81	-0.38	0.74	0.11	0.99
Zadok's score			0.55	0.05		
Eigenvalue	1.59	0.96	2.23	0.90	1.62	0.99
R^2	0.53	0.32	0.56	0.23	0.54	0.33

Table 4.4. *Habitat combinations used for first and second nests by females known to have re-nested following a successful first brood***.**

First nest	Second nest	Frequency	Successful second broods
AC	AC	1	1
AC	ROU	1	1
SC	SC	12	5
SC	FG	1	1
SC	ROU	1	0
FG	SC	3	1–2
FG	FG	4	1
ROU	SC	2	1–2
ROU	ROU	3	2
VEG	VEG	3	0
TOTAL		31	13–15

Table 4.5. *Parameter estimates and standard errors of predictors fitted in the GLMM, Model 1; Response variable habitat patch (selected for nesting, 1 or not selected, 0) = crop type + crop type|FED, with patchID nested within year as a random effect, and ln size of habitat patch as an offset. N = 577 nesting selection events (i.e. incidences of a habitat patch being selected) and 952 non-events (i.e. incidences of a habitat patch not being selected). The parameter estimates shown are relative to the intercept, which incorporates a reference crop (PAS). Year(patchID) parameter estimate = 0.4116, SE = 0.1698. Model fit* χ^2/df = *0.67.*

Effect	Parameter estimate	SE	df	t	*P*
Intercept (PAS)	0.2966	1.0642	1513	0.28	0.7805
AC	-0.7065	1.2097	1513	-0.58	0.5593
VEG	-4.9910	1.4064	1513	-3.55	0.0004
FG	2.1558	1.2014	1513	1.79	0.0729
SC	-1.8368	1.1288	1513	-1.63	0.1039
ROU	-1.5218	1.2506	1513	-1.22	0.2238
RES	-9.9128	3.4029	1513	-2.91	0.0036
OSR	-3.7801	3.1475	1513	-1.20	0.2299
PAS	0	-	-	-	-
FED|AC	-0.0159	0.0113	1239	-1.41	0.1584
FED|VEG	0.0477	0.0136	1513	3.52	0.0004
FED|FG	-0.0564	0.0101	1513	-5.56	<0.0001
FED|SC	0.0239	0.0062	1513	3.83	0.0001
FED|ROU	0.0070	0.0109	1513	0.64	0.5201
FED|RES	0.1292	0.0457	1513	2.83	0.0047
FED|OSR	0.0196	0.0546	1513	0.36	0.7192
FED|PAS	-0.0671	0.0232	1513	-2.89	0.0040

Table 4.6. *Parameter estimates and standard errors of predictors fitted in GLMMs assessing the relative influence of sward height, density, weeds and Zadok's score (cereals only), expressed as PCA axes, on Corn Bunting use for nesting across **(a)** all fields, **(b)** cereals only and **(c)** grass fields only. Response variable number of nests within a field (event) divided by number of nesting attempts recorded within territories intersecting that field (trial) = PCA 1 + PCA 2, with period, fieldID and year as random effects. Nesting selection events are incidences of a habitat patch being selected, and non-events are incidences of a habitat patch not being selected.*

a) *All fields (n = 206 nesting selection events and 959 trials). Year(period|fieldID) parameter estimate = 1.2244, SE = 0.2659. Model fit χ^2/df = 0.83.*

Effect	Parameter estimate	SE	df	t	*P*
Intercept	-1.5104	0.1132	410	-13.34	<0.0001
PCA 1 (*height, density*)	0.2992	0.0954	410	3.14	0.0018
PCA 2 (*density, weeds*)	0.4705	0.1113	376.8	4.23	<0.0001

b) *Cereal fields (n = 114 nesting selection events and 676 trials). Year(period|fieldID) parameter estimate = 1.1488, SE = 0.3321. Model fit χ^2/df = 0.78.*

Effect	Parameter estimate	SE	df	t	*P*
Intercept	-1.8719	0.1486	268	-12.60	<0.0001
PCA 1 (*height, Zadok's score*)	0.1567	0.1089	268	1.44	0.1511
PCA 2 (*density, weeds*)	0.5806	0.1450	268	4.00	<0.0001

c) *Grass fields (n = 78 nesting selection events and 224 trials). Year(period|fieldID) parameter estimate = 1.1076, SE = 0.4903. Model fit χ^2/df = 0.99.*

Effect	Parameter estimate	SE	Df	t	*P*
Intercept	-0.7374	0.1948	87.74	-3.79	0.0003
PCA 1 (*height, density*)	0.3653	0.1609	88.17	2.27	0.0256
PCA 2 (*weeds*)	-0.1298	0.1879	78.32	-0.69	0.4918

Table 4.7. *Timing of nesting of Corn Buntings reported in other UK studies.*

Study area	Years	Nests	FED range	Reference
Cornwall ^{E}	1933–34	54	Early June–early August	Ryves & Ryves 1934
Cornwall ^{E}	2006–08	200	29 May–3 August	Setchfield 2012
England ^{E}	1948–89	c.400	2 May–7 August	Yom-Tov 1992a
Lancashire ^{E}	1999	18	Early June–early July	Murphy 2000
Lincolnshire ^{E}	1994	32	5 June–13 July	Gillings & Watts 1997
Sussex ^{E}	1995–97	120	21 May–29 July	Brickle 1998
Sutherland ^{S}	1957–64	35	16 May–early August	Macdonald 1965
Western Isles ^{S}	1989–90	73	30 May–9 August	Hartley 1991

^{E} = *England;* ^{S} = *Scotland.*

Figure 4.1. *Map showing location of the 32 study farms, and the four sub-regions within eastern Scotland. Nest success studies presented in Chapter 5 relate to all farms in Inverness-shire and Aberdeenshire (excluding two denoted by open circles).*

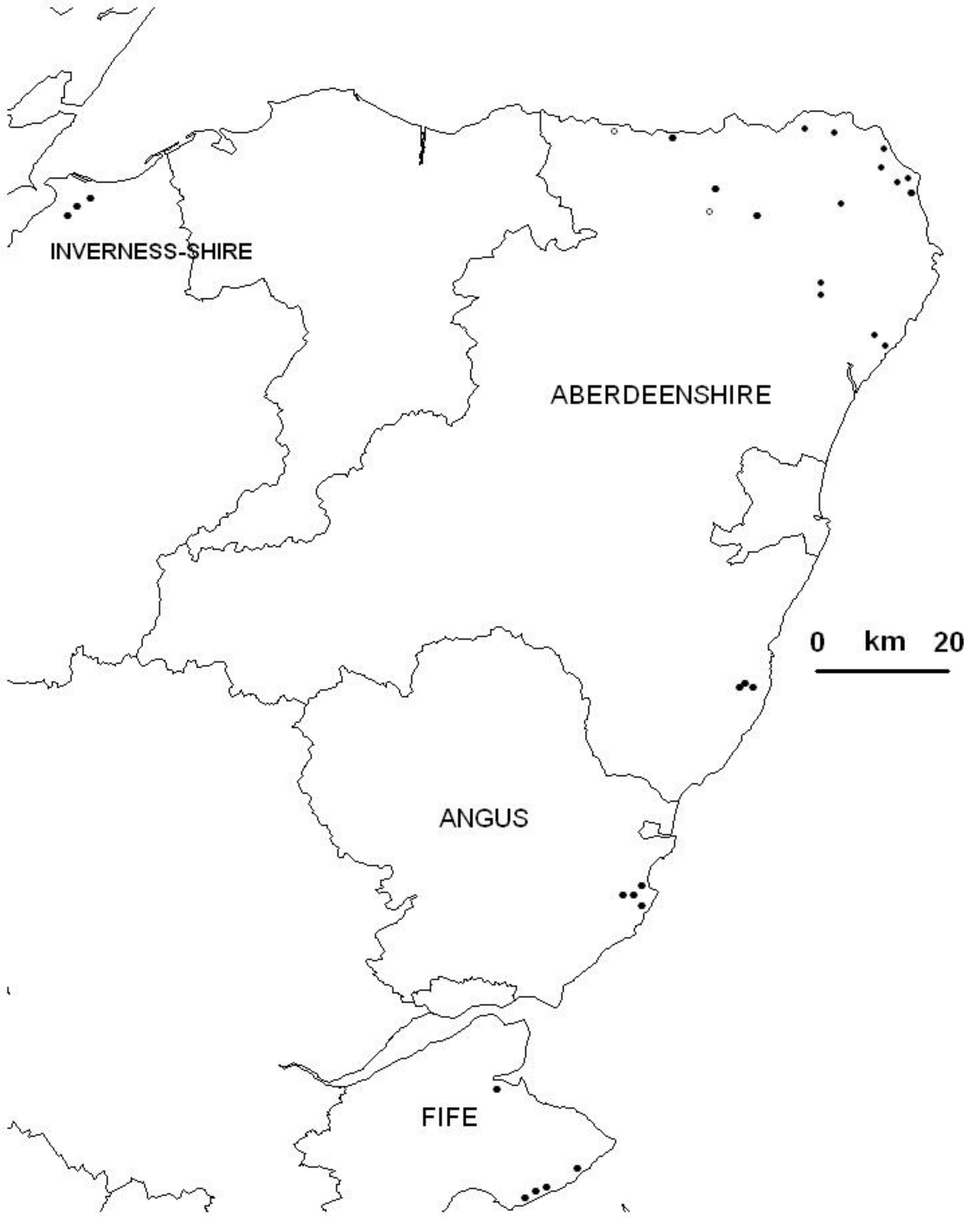


Figure 4.2. *Date of sward measurement plotted against first egg date (FED) of nests used when modelling probability of field use as a function of sward characteristics. Although there is some scatter, the plot indicates that measurements gave a good approximation of sward conditions during nest building, when field selection by females took place.*

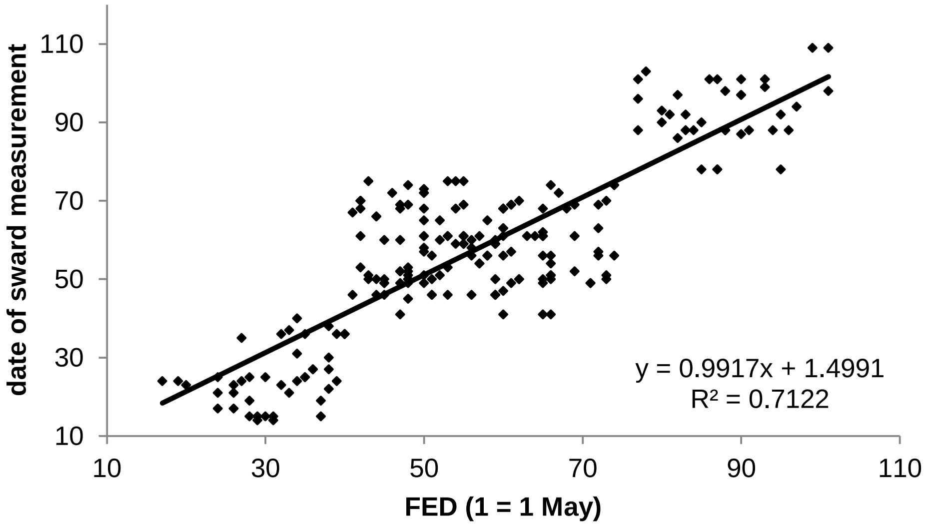
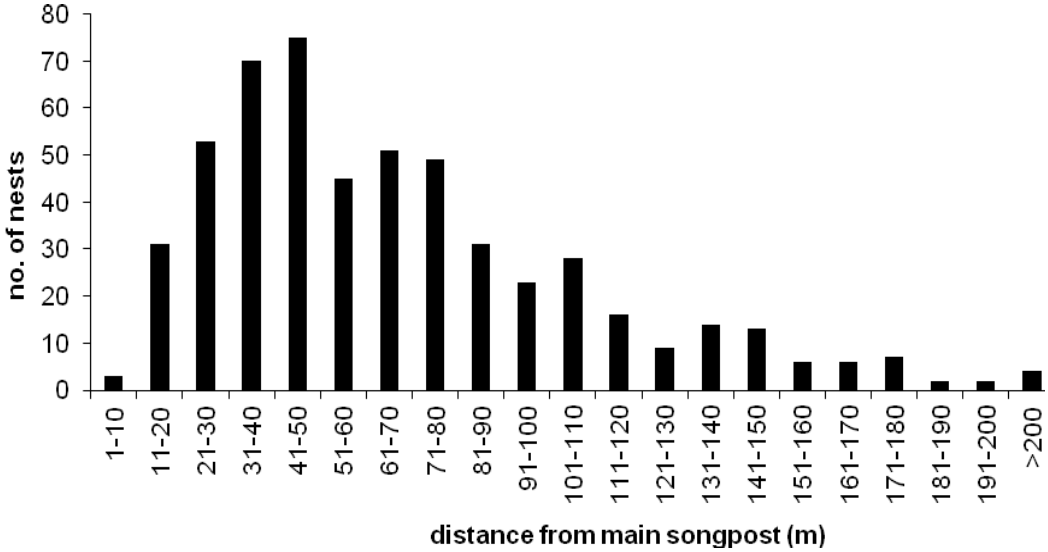


Figure 4.3. *Frequency distribution of nest distances from:*

a) *Main song-post of the male (n = 538 nests).*



b) *Nearest field boundary (n = 542 nests).*

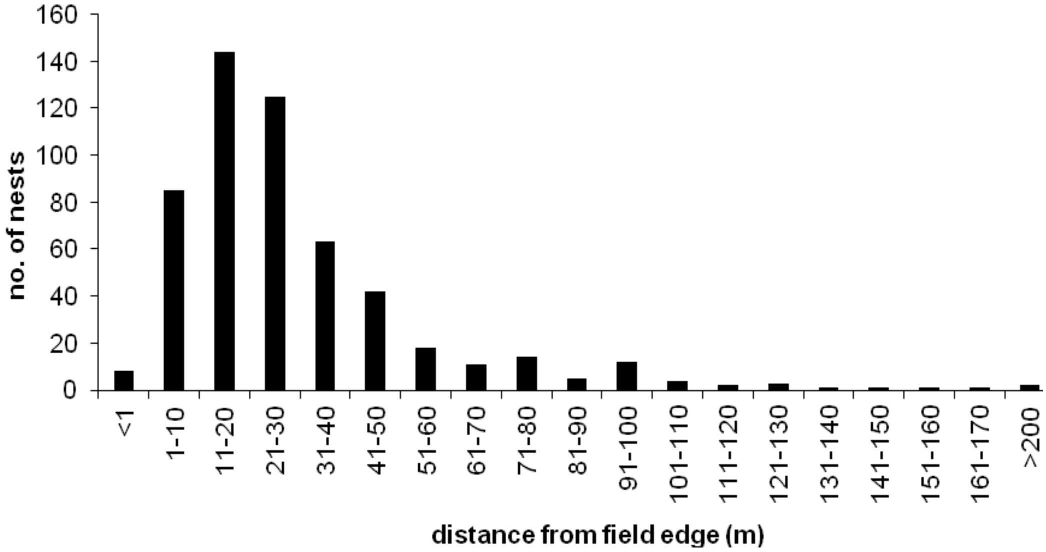
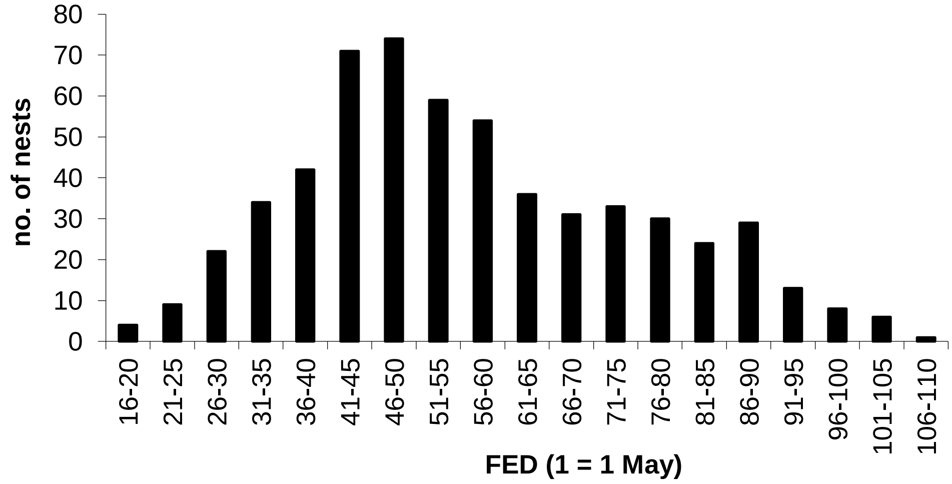


Figure 4.4. *Frequency distribution of first egg dates (FED) of nests for:*

a) *All regions combined.*



b) *Aberdeenshire/Inverness-shire versus Fife/Angus.*

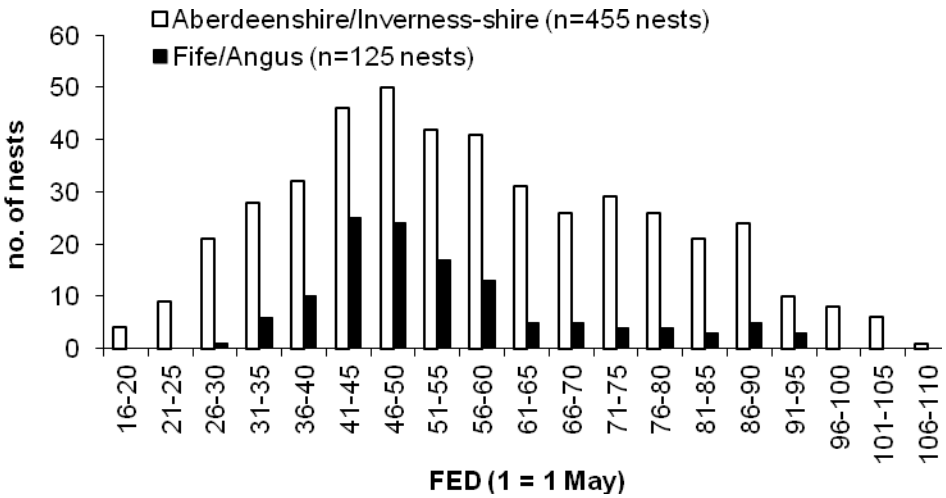
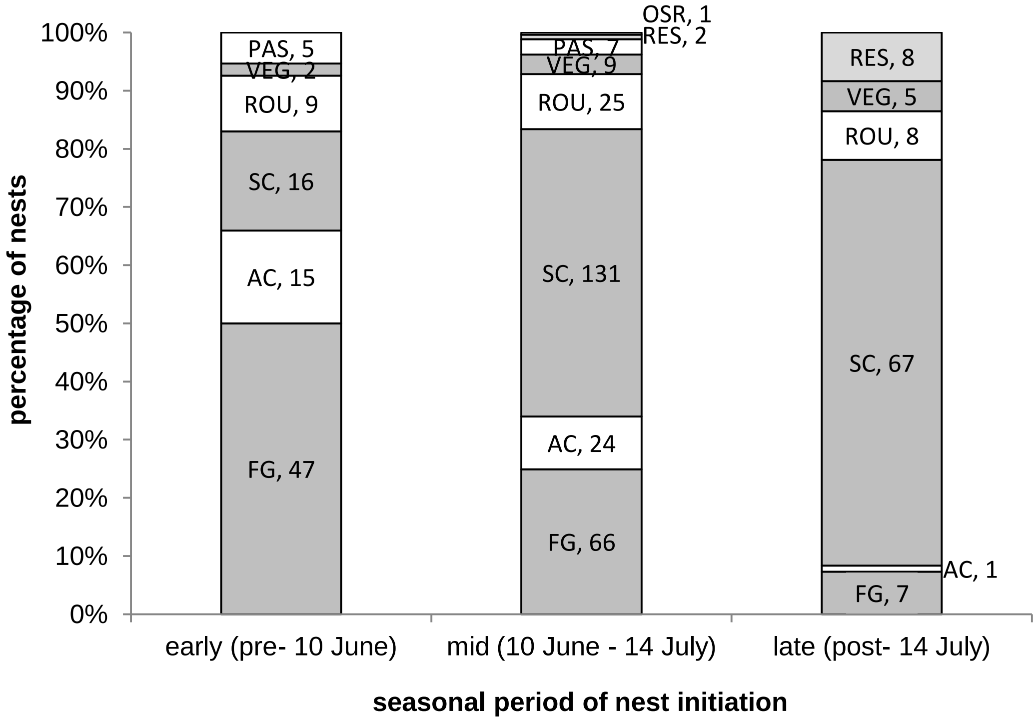


Figure 4.5. *Distribution of nests amongst crop types, sub-divided into early, mid and late season by date of nest initiation (first egg date). 'Early' and 'late' approximate to the earliest and latest 20% of nests, respectively. The number of nests is given after each crop type label.*

a) *Aberdeenshire & Inverness-shire (n = 455 nests during 2004–09).*



b) *Fife & Angus (n = 125 nests during 2006–09).*

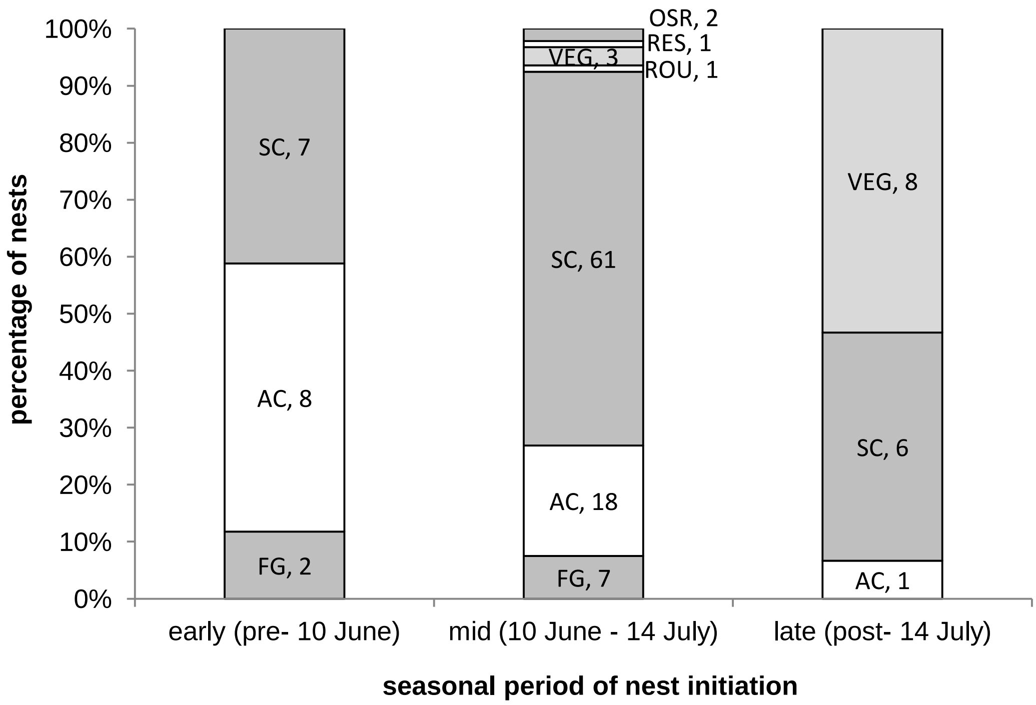
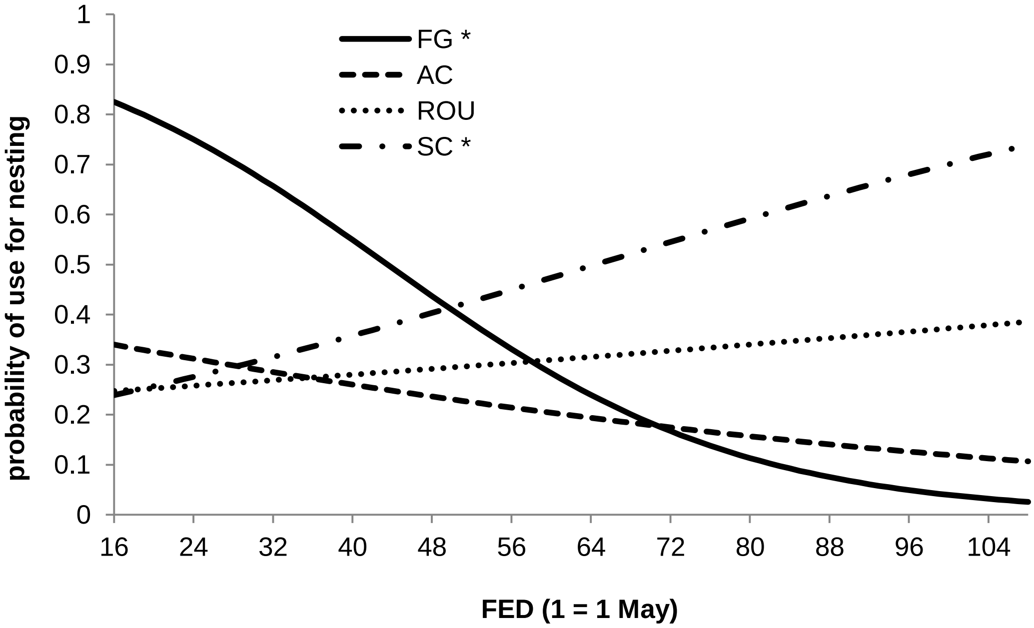


Figure 4.6. *Seasonal variation in probability of crop use for nesting, fitted using GLMM Model 1 (see Table 4.5). * = significant seasonal trend (P < 0.05 for crop type|FED effect in Table 4.4).*

a) *Crops widespread and often used for nesting.*



b) *Crops scarce or used infrequently for nesting.*

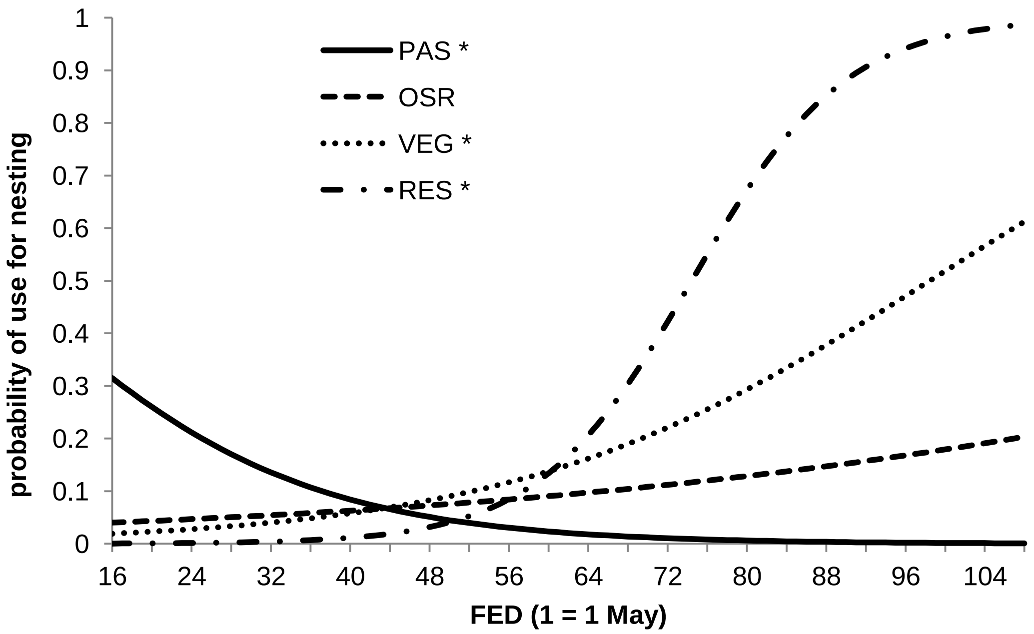
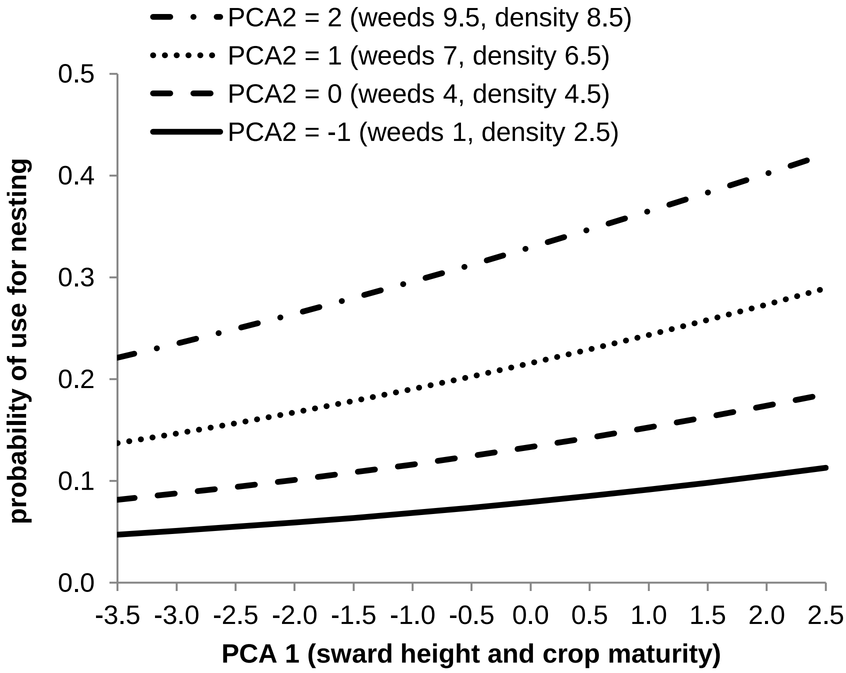


Figure 4.7. *Fitted relationship between crop sward characteristics expressed as PCA axes (using the model parameter estimates presented in Table 4.5) and the probability of field use for nesting. For each line plotted, PCA 1 is increased from zero whilst PCA 2 is held constant. Each line represents a different value for PCA 2.*

a) *Cereal fields. Each incremental increase of one along the x-axis (PCA 1) represents a 15 cm increase in sward height and a 1.2 increase in Zadok's score. Where PCA 1 = 0, sward height = 64 cm and Zadok's score = 5.6.*



b) *Grass fields. Each incremental increase of one along the x-axis (PCA 1) represents a 15 cm increase in sward height and a 2.8 increase in sward density score. Where PCA 1 = 0, sward height = 45 cm and density score = 6.1.*

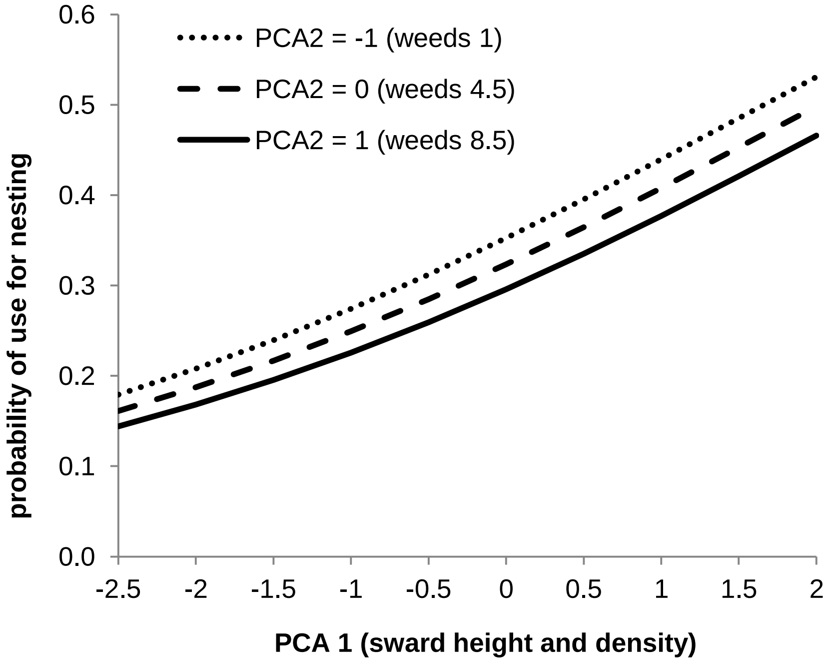
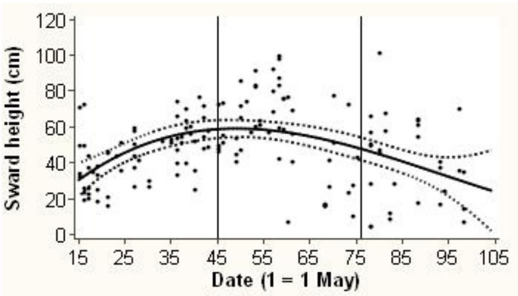
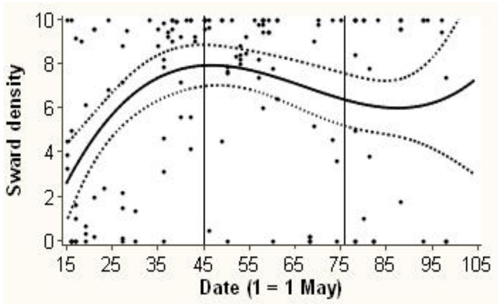


Figure 4.8. *Scatter plots showing seasonal variation in sward height and density of grass fields (FG and ROU) and cereal fields (AC and SC). The solid line plots the mean predicted value for each date, and dotted lines the 95% confidence limits (using the SAS GPLOT procedure, specifying a cubic regression equation). Vertical lines sub-divide the plots by period (early, mid and late season).*

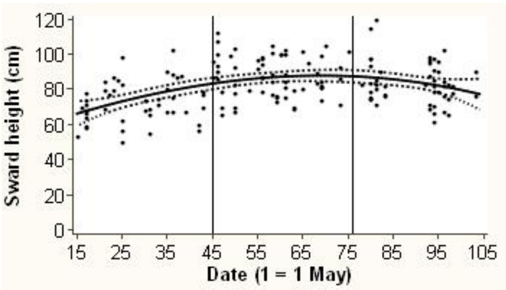
a) *FG height.*



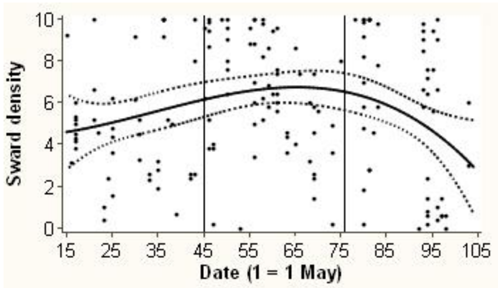
b) *FG density.*



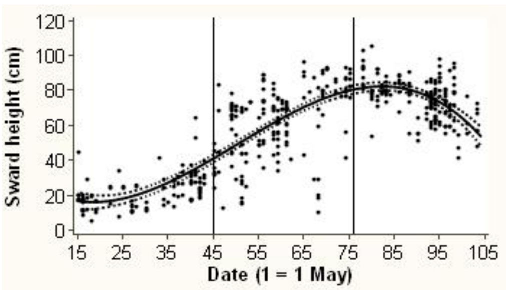
c) *AC height.*



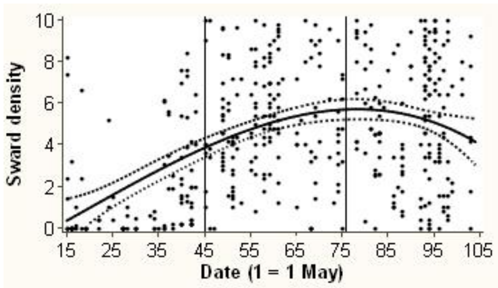
d) *AC density.*



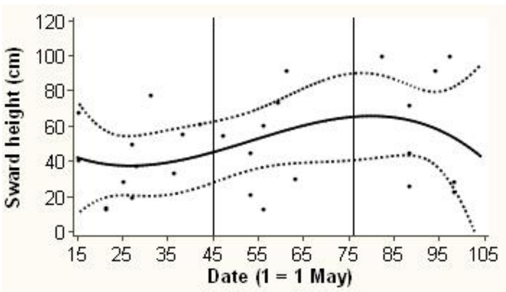
e) *SC height.*



f) *SC density.*



g) *ROU height.*



h) *ROU density.*

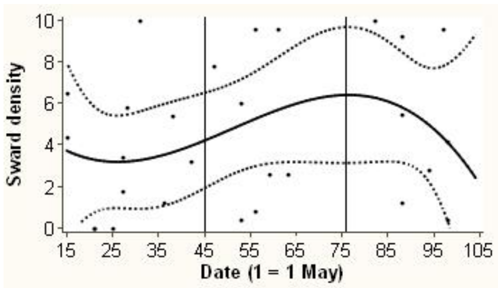
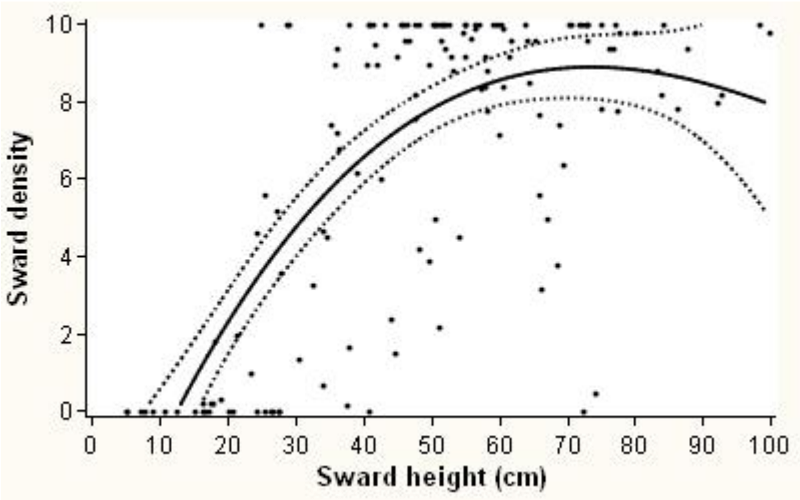


Figure 4.9. *Scatter plots showing the relationship between sward height and density in forage grasses (FG) and cereals (AC and SC). The solid line plots the mean predicted sward density score at any given crop height up to 100 cm, and dotted lines the 95% confidence limits (using the SAS GPLOT procedure, specifying a cubic regression equation).*

a) *FG.*



b) *Cereals (AC and SC combined).*

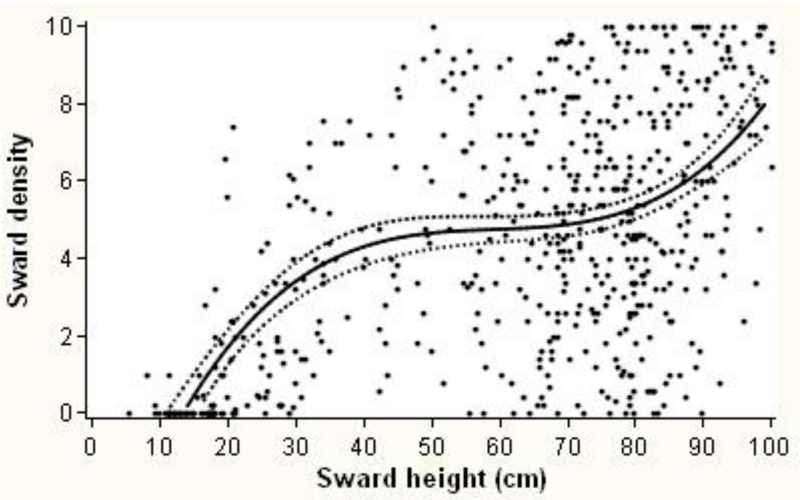
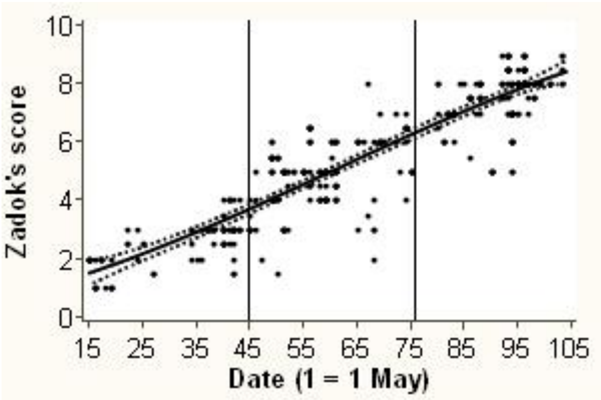
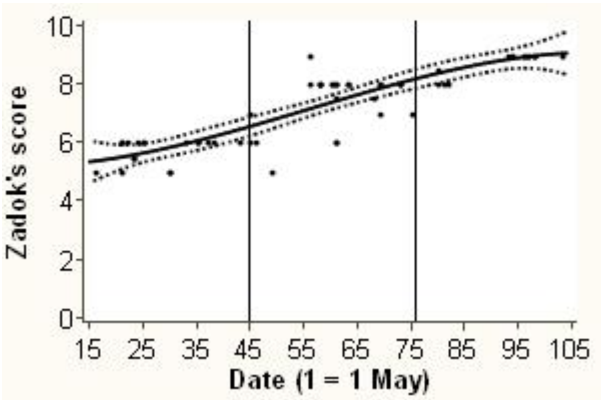


Figure 4.10. *Scatter plots showing seasonal pattern of crop maturity (Zadok's score) for each of the three main types of cereal grown in eastern Scotland. The solid line plots the mean predicted Zadok's score for each date, and dotted lines the 95% confidence limits (using the SAS GPLOT procedure, specifying a cubic regression equation). Vertical lines sub-divide the plots by period (early, mid and late season).*

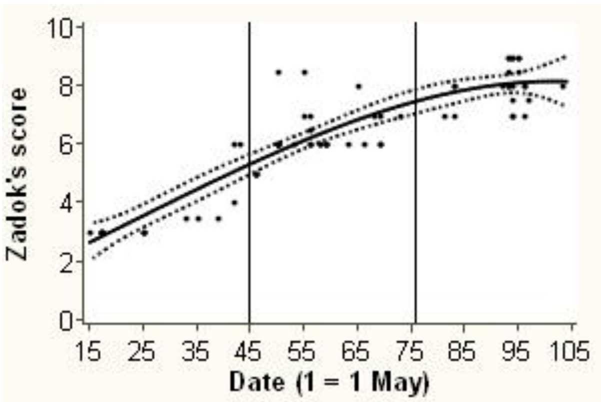
a) *Spring-sown barley.*



b) *Autumn-sown barley.*



c) *Autumn-sown wheat.*



CHAPTER 5. NEST SUCCESS AND TRIALS OF DELAYED MOWING TO INCREASE FLEDGING IN MEADOWS

5.1. Introduction

In northern Europe, Corn Buntings occur almost exclusively in farming landscapes where a preference for nesting within fields of growing crops and late onset of breeding makes nests, chicks and fledglings vulnerable to harvesting operations, especially where autumn-sown cereals predominate (Crick *et al*. 1994, Brickle & Harper 2002). Although most often associated with cereal cultivation, Corn Buntings also occupy extensively grazed and semi-natural grasslands in some areas of Europe, such as Iberia and the central-eastern countries (Diaz & Tellaria 1997, Báldi *et al*. 2005). Studies in Germany, the Netherlands and Poland have also shown strong associations with traditionally managed, herb-rich meadows (Hustings 1997, Golawski & Dombrowski 2002), and with large areas of set-aside dominated by grasses and herbs (Eislöffel 1997, Fischer & Schöps 1997).

Historically, hay meadows were widely used by Corn Buntings for nesting in the UK (Shrubb 1997), but recent studies recorded few nests in forage grasses mown for silage because these grass crops are usually harvested before Corn Buntings begin nesting (Gillings & Watts 1997, Murphy 2000, Brickle & Harper 2002). In mixed farming areas of eastern Scotland, however, forage grasses managed for hay or silage attract territorial male and nesting female Corn Buntings, especially during the early part of the breeding season, and the first cut is sufficiently late to allow widespread nesting in meadows in May and June (see Chapters 3 and 4). Populations here, though, are in rapid decline (Watson *et al*. 2009, and see Chapter 2), and one candidate cause is low reproductive success due to destruction of nests on a large scale during mowing of forage grasses for silage and hay (Wilson *et al*. 2007a).

Earlier and more frequent mowing associated with a switch from hay to silage (see Chapter 1) is thought to be responsible for declines in several species of meadow-nesting passerines, waders and game-birds in Europe and North America (Brennan & Kuvlesky 2005, Wilson *et al*. 2009). European species with declines driven by reduced reproductive success, caused by the direct effect of earlier and more frequent mowing destroying nests and killing chicks, include Corncrake, Yellow Wagtail and Whinchat (Green *et al*. 1997, Court *et al*. 2001, Müller *et al*. 2005). Grassland intensification can also affect birds indirectly by depleting seed and invertebrate food resources (Vickery *et al*. 2001), and a reduction in large invertebrate prey likely contributed to the virtual disappearance of Cirl Bunting and Red-backed Shrike from the UK (Evans *et al*. 1997, Brambilla *et al*. 2007). Direct and indirect effects of mowing often act simultaneously, as demonstrated in studies of Whinchat and Black-tailed Godwit (Britschgi *et al*. 2006, Kleijn *et al*. 2010).

In this chapter, I analyse and present data on the fledging success of Corn Bunting nests in different crop types, and test the effect of trial conservation interventions designed to increase Corn Bunting nest success in meadows cut for silage or hay by delaying mowing until after broods had fledged. I also compare nest success and the proportion of females nesting in forage grasses with other crop types, to measure the overall effect of delaying mowing on reproduction at the population scale. The viability of delayed mowing from a farming perspective, and alternative approaches, are discussed.

5.2. Methods

5.2.1. Study sites

We[5] monitored Corn Bunting nesting activity throughout the breeding season (May to early September) on 21 farms between 2004 and 2008, varying annually from eight in 2004 to 18 in 2007. These were the farms in Aberdeenshire and Inverness-shire (except for two) used in the study of habitat selection by nesting females (Chapter 4, Fig. 4.1). Most farms grew a mixture of arable and grass crops, mainly cereals, vegetables, forage grasses mown for silage and hay, and pastures grazed by beef cattle and sheep (Table 5.1, Plates 11–13). I did not use data from Fife and Angus in this chapter because study farms in those regions grew mainly arable crops with little grass, few Corn Buntings nested in meadows, and therefore meadow-management conservation interventions were rarely implemented in those regions.

5.2.2. Field management

In northeast Scotland, grass swards in meadows are typically rye grass *Lolium* spp. with varying amounts of clover *Trifolium* spp., re-seeded every 3 – 7 years, with inorganic fertiliser applied in April or May at a rate of 65 – 150 kg N ha^{-1}. Most meadows are rolled

[5] Several colleagues helped with this study. I designed the monitoring protocol for all farms, but surveys were shared with Hywel Maggs, Amanda Biggins, Ken Bruce, Alan Bull, Steven Coyne, Richard Firmin, John McMahon and Adam Watson. I personally undertook monitoring on eight of the 21 farms. I also designed and carried out all statistical analyses (with advice from Jeremy Wilson), and as lead author of the paper (not yet submitted), wrote the first draft and incorporated improvements suggested by the co-authors (Adam Watson, Jeremy Wilson, Hywel Maggs), and by John Deag.

during April – May to promote grass growth, and to flatten the ground surface to minimise the risk of damage to mowing machinery during cutting. Farmers usually take one or two cuts, rarely three, and meadows are often grazed after the final cut. Harvest dates of cereals and forage grasses (first cut only) are given in Fig. 5.1.

Conservation management interventions (administered by RSPB, with financial support from Scottish Natural Heritage (SNH) and the Scottish Government) to delay mowing of forage grasses (DM) began in 2005, and were subsequently implemented on twelve farms selected according to which farmers agreed to participate (Table 5.2). DM was usually across whole fields (n = 27), but on twelve occasions involved part fields, and areas ranged from 0.61 to 14.98 ha (mean = 6.63 ha, sd = 4.18). In most cases, selection of treatment meadows was based on nest locations of Corn Buntings in previous years. Farmers were paid to 1) delay mowing until 1 August; 2) ensure all livestock were removed from the field on or before 30 April; 3) ensure no rolling took place after 30 April. The latter two requirements were to prevent farmers from simply delaying all of their field management operations (in response to the requirement to cut late), as this may have prevented the development of a dense sward (c.30–60 cm tall) by late May when Corn Buntings begin nesting. We chose 1 August as the delayed mowing date because nest monitoring in 2004 suggested that almost all first broods should have fledged by then, confirmed by further monitoring since (Fig. 5.2 and see Chapter 4). It also matched the date in an existing option in Scotland's national agri-environment scheme to delay mowing for Corncrakes (SEERAD 2003), and any future use of this option for Corn Buntings outside of the Corncrake's range would be easier from an administrative aspect if the dates were the same. In practice, mowing often took place a few days after 1 August, dependent on weather.

Because this was a monitored trial conservation intervention, a certain degree of flexibility was possible regarding management agreements. In some cases, selection of treatment meadows took place during the breeding season once nests had been located. Further, fieldworkers allowed farmers to cut meadows up to 14 days earlier if nests had already fledged or failed. We also allowed earlier mowing if chicks were not due to hatch before 1 August. Although this action condemned clutches to failure through nest destruction or abandonment, it prevented chick deaths later on, as we could not reasonably expect farmers to delay mowing beyond early August. On one occasion, however, a farmer did delay mowing voluntarily until 12 August to allow a late brood to fledge. Clearly, in a wider agri-environment scheme without intensive monitoring by fieldworkers, such a flexible approach to management agreements would not be feasible.

5.2.3. Monitoring

We monitored the nesting activity of all female Corn Buntings on the study farms, except at three sites (farms 4, 7 and 17) where high densities made this impossible. Here, monitoring focused on territories that encompassed treatment (DM) or conventionally managed (SIL) meadows. Each farm was visited at least once per week from May to early September. Details of the methods used to find and monitor nests are in Chapter 4. Using behavioural observations of the nesting 'pair' (rather than visiting the nests, which would have risked causing predations or desertions) we monitored nests every three to seven days until the nest failed or chicks fledged. Nest failure was assumed in the absence of activity at or around the nest during at least two one-hour observations prior to expected fledging, and confirmed by later nest inspection (eggs or chick remains, or nest damage with no evidence of fledging). Observations of adults taking food into the crop nearby confirmed that chicks had left the nest. However, Corn Bunting chicks frequently leave the nest before they can fly, as young as nine days old (Harper 1995), when they remain vulnerable to agricultural operations. Therefore, a brood was considered to have fledged successfully only if at least one chick survived until it was capable of flight (15 days old). Observation dates of females incubating clutches or feeding broods were used to estimate the first egg date (FED) of each nest (Fig. 5.2), using the method presented in Chapter 4. Nest visits to ring chicks allowed the recording of brood size and unhatched eggs in a sample of nests, whilst in some cases, post-fledging observations allowed the number of young fledged from successful nests to be accurately determined (Table 5.3).

Corn Buntings can lay two or three clutches during a season, but most females were not individually identifiable so it was not always possible to determine whether a nesting attempt was a first clutch, repeat following failure of the first nest, or second attempt following successful fledging of the first brood. However, almost half of the 428 nests found were considered to be first attempts (Table 5.2), based on there being no earlier observations of nesting activity within a territory. At least 31 second clutches followed successful fledging of a first brood, where the identity of females were known from colour-rings or observations of nest building whilst still feeding fledglings. The period between fledging of the first brood and initiation of the next clutch was typically less than two weeks (mean = 12 days, sd = 8 days). No birds were recorded laying clutches following two successful, fledged broods.

5.2.4. Data analysis

5.2.4.1. Nest success in meadows and other crops

To obtain unbiased estimates of nest success, we used daily nest survival probabilities based on the number of days each nest was monitored (e.g. Mayfield 1961). This approach removes bias towards successful nests caused by the failure to detect some nesting attempts that fail at an early stage. We followed the methods of Shaffer (2004) to compare nest success between treatment meadows (DM) and conventionally managed meadows (SIL), and with other habitats, by fitting logistic-exposure models to nest observation interval data in SAS (version 9.2) using the GENMOD procedure. This method measures nest survival across multiple intervals between successive nest observations, and unlike traditional Mayfield analyses, does not require assumptions that nest survival is constant across time, or of exactly when nest losses occur within an interval. However, because observation intervals vary in length, SAS procedures are required that allow logistic regression to be done iteratively for each day in an interval (Rotella *et al*. 2004). Thus, daily nest survival was modelled using a generalized linear model (GLM) by specifying the response variable as the ratio of the outcome (*fate* = 1, survived or 0, failed) to the number of trials (1) for each nest observation interval, an inverse link function between an interval's fate and *field type* (eight classes), and assuming a binomial error distribution with a logit link function (Rotella *et al*. 2004, Shaffer 2004). *Year* was also fitted (fixed effect class variable) to control for annual variation in weather that may have affected harvest dates and nest success.

The LSMEANS command was used to obtain estimates of daily nest survival for each field type, which were then raised to the power 30 (15 days egg laying/incubation, and 15 days chick) to calculate fledging rates. Any nest observation interval where survival between the two (survived or failed) was uncertain was excluded from the analysis, as were nests found at the building stage that had apparently failed on the following visit (because the female may have abandoned the attempt before laying any eggs). Six late-season nests in SIL initiated after the first cut had taken place were also excluded. Analysis was therefore based on observations of 334 nests – 89 in forage grass meadows (52 DM and 37 SIL on 15 farms), 150 in spring-sown cereals including arable silage and unharvested crops (SC, 18 farms), 32 in autumn-sown cereals (AC, nine farms), 15 in vegetables, oilseed rape or daffodils (VEGRD, five farms), 30 in rough grass or set-aside (ROU, eight farms), nine in pasture (PAS, four farms), and nine in newly re-seeded grass (RES, five farms).

5.2.4.2. Effect of mowing date on nest success in meadows

We tested for differences in nest survival rates excluding losses to mowing between DM and SIL, by repeating the model using nest observation intervals up to the date of failure, but recording the outcome as survived where mowing was the cause of nest loss. We then pooled DM and SIL to obtain the underlying daily nest survival rate in forage grass meadows, which, together with estimated FEDs, was used to plot the proportion of nests active on each date throughout the season, to compare the number and proportion of nests vulnerable to cutting for a range of mowing dates. Similarly, we used estimated fledging dates (FED + 30 days) and the fledging rate in DM to plot the cumulative proportion of nests fledged against date.

Then, to formally test the effect on fledging rates of mowing earlier than 1 August, we modelled daily nest survival for DM assuming alternative mowing dates of 1 July, 10 July, 15 July and 24 July. We did this by re-coding successes as failures for observation intervals that extended beyond the nominal mowing date, and removing any subsequent observation intervals for that nest. For each of these dates, we assessed by how much delayed mowing would increase the overall proportion of first broods fledging across the study population. We did this for each habitat by multiplying the proportion of known first nesting attempts by the fledging rate, and summing these products. We then compared fledging rates for different percentages (10% increments from 0–100%) of forage grass nests in DM.

5.2.4.3. Predicted effect of delayed mowing on overall annual productivity

Finally, to estimate the effect of delayed mowing on overall annual productivity, taking into account repeat and second broods, we used a simple model to calculate the number of broods reared on each farm under various mowing regimes. Assuming a sex ratio of 1:1 (Watson *et al*. 2009 – Appendix 1), we used the mean annual number of territorial males recorded during population monitoring studies (see Chapter 7) as a surrogate for number of nesting females on the farm. The proportion of known first nesting attempts recorded in each field type was our starting point (Table 5.2). In the model, each female made three nesting attempts (or two successful ones), two in the habitat first used, with the third attempt always in spring cereal (SC).

The matrix of all possible nest outcomes per female is thus:

FLEDGE	FLEDGE	STOP
FLEDGE	FAIL	FLEDGE
FLEDGE	FAIL	FAIL
FAIL	FLEDGE	FLEDGE
FAIL	FLEDGE	FAIL
FAIL	FAIL	FLEDGE
FAIL	FAIL	FAIL

Table 5.4 shows a worked example for one site (farm 4). By applying alternative fledging rates (DM or SIL) to those nests in forage grass meadows, we estimated the number of broods likely to fledge successfully with and without delayed mowing on each farm, and summed these to give an overall total for the study population.

5.3. Results

5.3.1. Distribution of nests by habitat

Across all years, 115 of 428 nests found were in forage grasses, with 9–26% of meadows used per year (mean = 18%). Of 203 nests considered to be first attempts, 39% were in forage grasses, 16% in other types of grass or set-aside, 29% in spring cereals, 12% in autumn-sown cereals, 3% in vegetables, and 1% in other crop types (Table 5.2). Whilst farms with a larger area of meadow tended to have a greater overall proportion of nests in forage grasses (n = 21 farms, $r_s = 0.530$, $P = 0.014$), there was no such correlation with only those nests considered to be first attempts ($r_s = 0.288$, $P = 0.232$), indicating disproportionately greater use of meadows for nesting in early summer relative to their available area.

5.3.2. Nest success and causes of failure

Of the 428 nests found, 206 were known to have fledged at least one chick, 166 definitely failed, and the fate of 56 was unknown. Mowing and harvesting operations accounted for 61 of the 166 failures, predation a further 16, and chick starvation or exposure another six. For the remaining 83 nests that definitely failed, the exact cause of failure was unknown, but was not due to mowing or harvesting.

Controlling for bias towards greater detection of successful nests, the modelled survival rate across 334 nests gave a mean daily nest survival of 0.9673 (95% confidence limits = 0.9616–0.9723) across all habitats. Hence, the probability of a nest fledging successfully was 37% (cl = 31–43%). However, there was considerable variation between field types (Fig. 5.3), and nest success was particularly low in conventionally managed meadows of forage grasses (SIL – mean daily survival = 0.9207, cl = 0.8823–0.9473; fledging rate = 8%, cl = 2–20%) and in fields of newly re-seeded grass (RES – mean daily survival = 0.9249, cl = 0.8520–0.9634; fledging rate = 10%, cl = 1–33%). This was because mowing destroyed 23 of 37 nests (62%) in SIL, and five of nine nests (56%) in RES.

Spring cereals (SC), autumn cereals (AC) and set-aside or rough grass (ROU) all had similar daily nest survival (SC mean = 0.9838, cl = 0.9771–0.9886; AC mean = 0.9788, cl = 0.9590–0.9891; ROU mean = 0.9831, cl = 0.9673–0.9913), giving a fledging rate of 61% (cl = 50–71%), 53% (cl = 29–72%) and 60% (cl = 37–77%), respectively. In pastures (PAS) and other crops such as vegetables (VEGRD), nest survival was highly variable (Fig. 5.3). As well as in SIL and RES, nest losses to mowing or harvesting were also recorded in DM (three nests, 6%), ROU (two nests, 7%), AC (one nest, 3%), and SC (five nests, 3% – of which four nests were in fields harvested in early August for arable silage).

Across a sample of 83 nests, brood size was 3.11 ± 1.05 sd, and across 74 nests the number of fledglings per successful brood was 2.93 ± 1.09 sd (Table 5.3).

5.3.3. Effect of delayed mowing on nest success in meadows

With delayed mowing (DM), the daily survival rate of nests in meadows was significantly higher (0.9721, cl = 0.9579–0.9816) than in SIL (0.9207, cl = 0.8823–0.9473; $\chi^2_1 = 12.97$, $P = 0.0003$). Consequently, the probability of a nest fledging successfully (Fig.5.3) was five times higher in DM (mean = 43%, cl = 27–57%) than SIL (mean = 8%, cl = 2–20%). This was because mowing destroyed few nests in DM (three of the 52 nests in the analysis, or 6% – all second brood attempts), compared with 62% in SIL. When losses to mowing were excluded, daily nest survival rates in SIL (mean = 0.9840, cl = 0.9687–0.9919) were no different to those in DM (mean = 0.9768, cl = 0.9633–0.9854; $\chi^2_1 = 0.68$, $P = 0.4096$). The only field type whose daily nest survival was significantly higher than DM was spring cereals (χ^2_1 = 4.18, P = 0.041).

5.3.4. Effect of mowing date on nest success in meadows

Figure 5.4 plots the modelled proportion of nests active in forage grass meadows, and the cumulative proportion fledged, against date, assuming no losses to mowing. In an average year, only 8% of nests would have fledged by 1 July (date 62), increasing to 24% by 15 July and 40% by 1 August. Conversely, the proportion active and vulnerable to destruction by mowing on those dates declined from 53% to 27% and 4%, respectively. However, these figures vary between years according to the timing of nesting. For example, mean FED of first nests in 2007 (5 June) was eight days earlier than the overall average across all years, and in 2006 was five days later (18 June).

5.3.5. Predicted effect of delayed mowing on overall annual productivity

Figure 5.5a shows the modelled percentage of first broods fledged across the whole study population for a range of delayed mowing dates and proportions of forage grass nests in DM. If all meadow nests were in DM mown on or after 1 August, across all habitats the number of first broods fledged would increase by 35%, from 0.38 to 0.52 broods per female (a net increase of 21 fledged broods across the study population). This compares with a 15% increase to 0.45 broods per female (10 extra fledged broods overall) if only half of the nests were in DM, and almost no increase if the delay in mowing was only until 1 July.

With the inclusion of second broods and repeat attempts following nest failures into models (Fig. 5.5b), the estimated increase in total number of broods fledged throughout the breeding season was 16%, from 1.25 to 1.45 broods per female (an overall net increase of 31 fledged broods), if all forage grass nests were in DM mown on 1 August. However, these figures varied considerably between individual farms, from a 66% increase to no increase (Table 5.5). Across seven farms where more than half of recorded first nests were in meadows, delayed mowing to 1 August would increase the annual number of broods fledged by 43%, from 0.94 to 1.34 broods per female (a net increase of 17 fledged broods across these farms). By comparison, across ten farms where meadow nests were less common (≤50% of first nests found), and consequently fewer nests were vulnerable to mowing and success was already relatively high, the estimated increase was just 9%, from 1.36 to 1.49 broods fledged per female (a net increase of 14 fledged broods across these farms).

Figure 5.5b also shows the effect of various mowing regimes on annual productivity. DM with an earlier permissible mowing date than 1 August would give a smaller increase in total broods fledged, of almost no increase for 1 July, 7% increase to 1.34 broods per female for 10 July, 11% increase to 1.39 broods per female for 15 July, and 14% increase to 1.42 broods per female for 24 July. For every 10% reduction in the proportion of meadow nests targeted by DM, the number of broods fledged per female would fall by 0.1–1.4% (the magnitude varying according to the permissible mowing date).

5.4. Discussion

Over the entire dataset of 428 monitored nests, 48% definitely fledged, 39% definitely failed, and the fate of 13% was unknown. Of 166 definite nest failures, mowing and harvesting operations accounted for 37%, predation a further 10%, and chick starvation or exposure another 4%. We found dead chicks in 11 nests overall (6 complete broods and 5 partial broods), and such losses were often associated with prolonged periods of wet weather, when invertebrate food availability was likely to be low. For the remaining 49% of failed nests, the cause of failure was unknown, but was not due to mowing or harvesting. In these cases, predation was the most likely cause of failure. The identity of predators was unknown, but in a study examining BTO Nest Record Cards for causes of nest failure amongst Corn Buntings, Yellowhammers and Reed Buntings, recorded nest predators that were also present on our study sites included Red Fox *Vulpes vulpes*, Badger *Meles meles*, Stoat *Mustela erminea*, Weasel *Mustela nivalis*, mouse *sp*., Domestic Cat *Felis catus*, Carrion Crow and Black-billed Magpie *Pica pica* (Crick *et al*. 1994). Other potential predators of clutches or broods on our sites included Brown Rat *Rattus norvegicus*, Common Kestrel, Common Buzzard *Buteo buteo*, Eurasian Jackdaw and Black-headed Gull *Larus ridibundus*. Nest failures to mowing were most frequent in meadows of forage grasses, but also occurred in fields of newly re-seeded grass, and in set-aside. However, unlike in southern England (Brickle & Harper 2002), very few (seven of 235) nests in cereals failed due to harvesting operations.

The unbiased, modelled estimate for nest success was 37% overall, but varied between field types, being lowest in conventionally managed forge grass meadows (8%) and newly re-seeded grass (10%), and highest in spring-sown cereals (61%) and set-aside or rough grass (60%). In the other frequently used field type, autumn-sown cereals, the fledging rate was 53%. Compared with other studies (Table 5.3), only two (Crick 1997, Brickle 1998) reported lower nest success than our 37% rate across all habitats. However, the estimated values for nest success in cereals and set-aside or rough grass in our study are among the highest recorded for Corn Buntings in the UK. This suggests overall breeding productivity in northeast Scotland would be high were it not for the large number of nest losses in meadows.

Our only measure of clutch size was based on brood size plus the number of unhatched eggs during nest visits at the chick stage, so comparison with other studies is not possible. However, mean brood size and number of fledglings per successful brood are within the range of those reported in other UK studies, but are towards the lower end of the scale (Table 5.3).

5.4.1. Nest success in meadows and effect of conservation management interventions

More than one-third of Corn Buntings in northeast Scotland nest in meadows of forage grasses, with disproportionately greater use of these fields in early summer relative to their available area (see also Chapter 4). However, most nests fail when first cuts for silage or hay are taken between late-May and mid-July (Fig. 5.1 & 5.4). Failures are most likely due to direct destruction of clutches, broods and flightless young by the cutting machinery, rather than during subsequent processing of the mown grass or to predation (Humbert *et al*. 2009). On some of our study farms, this resulted in the failure of all first nesting attempts. Worse still, where mowing dates varied between fields, some females re-nested in forage grasses only to suffer the same fate when these meadows were cut, and did not fledge any young during the whole breeding season. If such high losses occur repeatedly each year, they are likely to be unsustainable and could lead to the extinction of some local populations (Watson *et al*. 2009 and see Chapter 2). By delaying mowing until 1 August, nest destruction in treatment meadows was reduced almost to zero. Only three nests were destroyed, and all were second attempts following successful fledging of first broods. Consequently, overall nest success was five times greater in meadows with delayed mowing (43%) than in those with conventional cutting dates (8%).

5.4.2. Predictions for annual productivity and population effects

Given the large number of meadow-nesting females, widespread adoption of delayed mowing could increase the number of first broods fledged by up to 35%, from 0.38 to 0.52 broods per female, giving a net increase of 21 fledged first broods across the study population. In addition, most cereals are harvested in late August or September (Fig. 5.1), so females fledging their first brood in meadows with delayed mowing in mid or late July have sufficient time to rear a second brood. We recorded 31 instances of females attempting a second brood, of which 13–15 were successful (see Chapter 4, Table 4.7), but these are minimum figures and the true number of second brood attempts was likely to have been much higher. Double-brooding can increase the annual fledgling production of female Corn Buntings by 50%, but is most frequent when first nests are started early (Hartley & Shepherd 1994b). In our study, the earliest nests were usually in forage grass meadows, so allowing these nests to fledge by delaying mowing can significantly increase annual reproductive success by allowing more females to rear two broods. Our calculations suggest that, with delayed mowing, the total number of broods fledged throughout the breeding season would increase by more than 60% on some farms (those with a high rate of meadow-nesting) and by 16% across the whole study population, from 1.25 to 1.45 broods per female, giving a net increase of 31 fledged broods across the study population. The biggest effect would be across the seven farms with > 50% of recorded first nests in forage grasses, where the number of broods reared per female would increase from 0.94 to 1.34 (equivalent to an increase from almost all females rearing just one brood, to one in three females also rearing a second brood). This may be important, as declines in southern England have been linked to a lack of opportunity to rear second broods where earlier harvesting of predominantly autumn-sown cereals leaves very little suitable nesting habitat after mid-August (Brickle & Harper 2002).

Population trends can be sensitive to modest changes in reproductive success. For example, regional population modelling of Savannah Sparrow *Passerculus sandwichensis* in northeast USA predicted that a 60% increase in annual productivity (from 1.3 to 2.1 fledglings per female) due to reduced nest losses to mowing would give a 34% population increase over 10 years, compared with an 8% decline under baseline conditions (Perlut *et al*. 2008). In their model, adult annual survival was 0.48–0.59, with juvenile survival (including the immediate post-fledging period) assumed to be 50% lower. Available data on annual adult survival for Corn Bunting (0.58) and other farmland passerines with similar ecology such as Yellowhammer (0.54), Reed Bunting (0.54) and House Sparrow (0.57) all fall within this range (Balmer & Peach 1996, Siriwardena *et al*. 1998). Applying a survival rate of 0.58 for adults and 0.29 for juveniles (50% of the adult rate) to our data, and assuming 2.9 young fledged per successful brood (mean = 2.93, Table 5.3), the 1.45 broods fledged per female with delayed mowing translates into a 19% population increase the following year across all study farms (Table 5.5). Without delayed mowing (1.25 broods fledged per female), the prediction is for a 10% increase overall, but with most of the increase occurring on farms where only a small proportion of Corn Buntings nested in meadows. However, across farms where more than half of recorded first nests were in meadows, the prediction without delayed mowing is for declines on six of the seven farms and for their combined population to decline by 2.4%, whereas with delayed mowing, numbers would increase on all seven farms, giving a population increase of 14.5% (from 86 to 98 breeding adults).

Clearly, these are merely predictions based on modelled estimates of annual productivity, survival estimates from other studies, assessments of the proportion of nests in forage grasses on each farm, and assuming, unrealistically, populations closed to immigration and emigration. They are also sensitive to small changes in parameter values. For example, a modest lowering of the juvenile survival rate from 0.29 to 0.25 would increase the magnitude of the predicted population decline for the seven farms with frequent meadow-nesting from 2.4% to 7.9% in the absence of delayed mowing, and similarly the magnitude of increase across all farms would reduce from 10% to 3%. Therefore, these predictions must be treated with caution. However, they do show the potential magnitude of effect that improved reproductive success through delayed mowing could have on population trends, and in Chapter 7, I examine the population response of Corn Buntings to delayed mowing and other conservation measures across a larger sample of farms in eastern Scotland.

5.4.3. Viability of delayed mowing in agri-environment schemes

Although Corn Buntings do nest in set-aside and rough grass in northeast Scotland, and nest success in this habitat is relatively high (Fig. 5.3), the denser swards of heavily fertilised forage grasses are more attractive, especially during early summer (see Chapter 4). This is unfortunate, as most farmers prefer to manage areas of land specifically for biodiversity rather than risk compromising the value of their crops by delaying harvest. The nutritional quality of silage deteriorates rapidly with each week’s delay in mowing beyond the optimum leaf-growth stage of the grass, adversely affecting energy intake and animal yield (Rinne *et al*. 1999, Dawson *et al*. 2002), and farmers delaying mowing in our study had to buy additional winter-feed to compensate. However, when asked about the viability of delayed mowing, individual farmers responded differently according to the type of livestock farmed. Silage quality was considered to be more critical, and hence late-cutting less viable, on farms where beef cattle were being fattened prior to slaughter than on those with sheep or suckler cows rearing calves (Ferguson & Grigor-Taylor 2009). This finding is similar to that of Nocera *et al*. (2005), who demonstrated that declines in nutritional quality of grass caused by delayed mowing to protect nesting Bobolinks *Dolichonyx oryzivorus* and Savannah

Sparrows was acceptable for some farmers, depending largely on the type of farm enterprise and livestock kept.

Based on our study, delayed mowing ("Mown Grassland for Corn Buntings") is now available in Scotland's main agri-environment scheme (Rural Priorities), as one of a package of options designed to provide food and safe nesting habitat for Corn Buntings (Scottish Government 2010a). However, with a payment rate of £224 ha^{-1}, only 64 management agreements (worth approximately £119 000 pa, covering 530 ha) had been secured by spring 2012, compared with 1105 agreements (worth £2.7m pa, covering almost 15 500 ha) for 'Mown Grassland for Wildlife', which permits mowing from 1 July for a lower payment of £175 ha^{-1} (Scottish Government 2012). Uptake of the Mown Grassland for Corn Bunting option could increase if the payment rate was higher or earlier mowing permitted, and from 2011 the option was modified to allow mowing one week earlier (24 July), although the payment rate for this is lower (£216 ha^{-1}).

5.4.4. Optimal mowing dates and scale of deployment

Our data show that in most years, advancing the mowing date from 1 August to 24 July would cause few (9%) additional nest losses (Fig. 5.4), but bringing the date forward further would be counterproductive. Mowing on 15 July would destroy a quarter of nests in those fields, despite increasing fledging rates of first broods by almost a quarter relative to conventional mowing dates, and mowing on 1 July would be far too early to allow most Corn Bunting broods to fledge. Setting a single optimum date, however, will always be difficult, as climatic variations between regions and years will affect the timing of both crop harvest and nesting. For example, in our study, mean FED of first nests in forage grass meadows varied annually from 5 – 18 June. Therefore, losses would be higher in years when nesting began late, or there was much re-nesting following failure of first attempts.

Whatever the mowing date, if management is to contribute to halting and reversing regional declines, it must be deployed on a sufficient scale to reach a large proportion of the population. For example, to achieve a 20% increase in first broods fledged across all study farms (and 10% increase in broods fledged throughout the breeding season), 60% of meadow-nesting females must be in fields cut on or after 1 August, rising to 70% with mowing permissible from 24 July, and 90% for 15 July. On some farms, deployment of delayed mowing on this scale would deliver even greater increases in the overall number of broods fledged, for example by 23–40% on the six farms with modelled population declines under conventional mowing, enough to halt the decline on each farm. Given that nests were found in one-fifth of meadows overall, this level of targeting could be achieved by implementing delayed mowing across approximately 12–18% of the area under forage grasses.

5.4.5. Targeting and alternative management

To ensure a high proportion of Corn Buntings benefitted from delayed mowing in our study, we selected meadows in which they were nesting or had nested in previous years. This would not be possible in a wider scheme without nest monitoring, so practitioners must assess which meadows to target for delayed mowing. Corn Buntings are most likely to use meadows in open areas away from woodland, and with overhead wires or isolated bushes that provide prominent song-posts for territorial males (see Chapter 3). In North America, Perlut *et al*. (2008) found that Bobolinks and Savannah Sparrows nested earlier in meadows that were subsequently cut earlier, due to differences in sward structure between field types at the onset of breeding. There was some evidence of this in our study, and further research should investigate how to make the swards of meadows selected for delayed mowing more attractive to nesting Corn Buntings at the start of the breeding season. Swards could also be improved to prevent them from collapsing following heavy rainfall, which sometimes led to females deserting clutches or young broods, and thus suppressed fledging rates relative to other habitats such as cereals (Fig. 5.3). One solution may be the inclusion of clover in meadow swards, whose thick stems and bushy structure make it less susceptible to collapse.

For farms where delayed mowing is not viable, alternative ways of reducing nest losses should be considered. One possible option is to take the first silage cut on the usual date, but leave patches or strips uncut to attract re-nesting birds (Buckingham *et al*. 2004, Masse *et al*. 2008). This would allow rapid re-nesting without birds having to wait for swards to re-grow, but does not prevent first nests from being destroyed. Therefore, such management could be combined with an earlier cut, before the onset of nesting, to minimise losses to cutting. Another alternative may be arable silage, an economically viable alternative to grass silage with potential benefits for farmland birds (Peach *et al*. 2011). In northeast Scotland, typically these are crops of spring-sown barley mixed with peas or clover, which are attractive to nesting Corn Buntings but harvested during late July or early August, so nest destruction remains a threat. Indeed, five of the six nests destroyed by harvesting operations in spring cereals were in fields cut early for arable silage. However, it is a much safer alternative to conventional forage grasses, with a significantly higher fledging rate (at least 12 of 21 arable silage nests were successful). Finally, habitat patches completely separate from agricultural crops have the advantage of remaining uncut throughout the breeding season. To attract nesting Corn Buntings, they should be periodically re-sown to ensure a dense flush of ground vegetation each summer. Cereal-based wild bird cover crops designed to provide winter seed food attracted nesting Corn Buntings in our study, and increased their reproductive success in southwest England (Setchfield *et al*. 2012).

5.4.6. Conservation implications

Increased reproductive success and reduced nest losses following the widespread adoption of delayed mowing of meadows through agri-environment schemes has led to population recovery in several European farmland birds. Like Corn Buntings, Corncrakes are double-brooded, and the number of fledglings per female increased almost threefold with delayed mowing, and by up to a quarter when meadows mown from the centre outwards (Green *et al*. 1997). These conservation measures have resulted in Corncrakes increasing by 5.6% p.a. in their core Scottish range during 1993–2004, compared with a decline of 3.4% p.a. during 1978–1993 (O'Brien *et al*. 2006). In western France, a single-brooded species, the Little Bustard, declined by 80% in one study area in just eight years, due to very low recruitment caused partly by nest destruction and female mortality during mowing of alfalfa (a fodder crop) and set-aside (Bretagnolle *et al*. 2011). Following the introduction of delayed mowing in 2004, almost 70% of nests were in treatment fields, and in combination with measures that increased the availability of grasshopper food for chicks, this caused breeding productivity to double, reversing the population trend. Also in France, delayed mowing in a quarter of a 3000 ha meadow area resulted in a doubling of meadow passerines over eight years, with population increases in Whinchat, Yellow Wagtail and Reed Bunting, and stability in Corn Bunting (Broyer 2011). However, the author concluded that these increases led to 'instable population dynamics', with breeding success declining at high territory densities, possibly associated with inter-specific competition for food. Such complexities were also apparent in a Swiss study of Whinchats (Müller *et al*. 2005), where 'sink' areas with low productivity caused by nest losses to mowing had misleadingly stable populations due to immigration.

Overall, each of these studies demonstrates that effective targeting of a simple conservation solution (delayed mowing) through agri-environment schemes can lead to rapid recovery of bird populations. In our study, we have shown that meadow-nesting Corn Buntings in northeast Scotland suffer high rates of nest loss, but that delayed mowing can substantially improve reproductive success. Modelled predictions suggest this has the potential to reverse population declines, especially on farms where a high proportion of females nest in forage grasses, and in Chapter 7 these predictions are tested by comparing population change on farms with delayed mowing and other agri-environment measures with trends on non-scheme control farms. However, to halt and reverse declines across the region, a large proportion of the population must benefit. The overall scale of deployment required would be in the order of one in eight meadows with delayed mowing to 1 August, rising to one in seven with mowing permissible from 24 July, and one in five for 15 July.

Table 5.1. *Land use composition (%) and total area (ha) surveyed in each year, and number of nests by habitat in early and late summer.*

Year	Area (ha)	SC	AC	VEG	OSR	FG	PAS	ROU	RES	OTH
2004	1674	32.7	15.5	2.5	3.9	15.5	17.6	4.6	1.0	6.8
2005	2688	36.5	14.0	2.6	1.9	14.1	20.0	5.1	1.2	4.0
2006	1574	30.7	7.8	5.1	3.5	13.7	28.2	6.9	1.1	3.0
2007	2898	31.2	12.4	3.2	2.7	15.5	23.4	5.0	1.7	5.6
2008	1597	34.9	13.7	5.4	2.8	13.7	18.3	3.1	0.0	8.1
May/June nests		82	34	8	1	99	10	28	0	2
July/August nests		114	5	5	0	16	0	13	10	1

SC = spring-sown cereals (barley and oats), legume/cereal mixture mown for arable silage, and unharvested crops; AC = autumn-sown cereals (barley, oats and wheat); VEG = root vegetables (potatoes, turnips, carrots and cabbages) and legumes (peas and beans); OSR = autumn-sown oilseed rape; FG = forage grasses mown for silage or hay (including meadows with delayed mowing); PAS = grazed pasture; ROU = rough grass and set-aside; RES = newly re-seeded grass; OTH = other habitats including soft fruit, daffodils, wetland and woodland.

Table 5.2. *Area of delayed mowing implemented on each farm in each year, with maximum annual area of forage grass meadows (FG) and mean annual number of territorial male Corn Buntings on each farm, and total number of nests found in each habitat. SIL = conventionally managed grass silage or hay; DM = delayed mowing; SC = spring cereals, arable silage, and unharvested crops; AC = autumn cereals; ROU = rough grass and set-aside; PAS = pasture; RES = newly re-seeded grass; VEG = root vegetables and legumes; ORD = oilseed rape and daffodils.* [a] *small patch around nest left uncut until after brood had fledged.*

Farm	FG area (ha)	Area p.a. with delayed mowing (ha)					Terr. males	Number of nests (known first attempts are shown in brackets)								
		2004	2005	2006	2007	2008		SIL	DM	SC	AC	ROU	PAS	RES	VEG	ORD
1	35.0	0	0	6.6	5.5	0	7	0	1 (1)	12 (1)	0	0	0	0	0	0
2	22.9	0	0.8	0	0	0	6	0	0	1 (0)	0	0	0	0	0	0
3	25.0	0	10.5	10.5	10.5	9.7	6	1 (1)	10 (4)	10 (0)	1 (1)	0	0	0	0	0
4	51.3	0	0	0	8.2	0	29	16 (13)	3 (3)	39 (16)	5 (4)	11 (2)	1 (1)	0	2 (0)	0
5	32.1	0	11.9	11.9	0	0	2	0	0	8 (7)	0	1 (1)	0	0	0	0
6	36.2	0	0	0	2.1	0	5	4 (3)	3 (2)	6 (3)	0	1 (0)	0	2 (0)	0	0
7	64.3	0	0	15.2	14.0	20.2	9	3 (1)	8 (7)	15 (1)	0	0	0	1 (0)	7 (2)	0
8	27.2	0	0	8.4	2.2	1.0	5	0	7 (4)	4 (0)	2 (1)	0	0	3 (0)	0	0
9	44.1	0	15.0	13.1	15.0	15.0	5	0	10 (5)	10 (0)	0	0	0	0	0	0
10	26.4	0	0	10.2	8.5	10.2	5	1 (1)	9 (7)	6 (2)	0	0	0	0	0	0
11	44.4	0	4.7	4.7	0	0	3	0	2 (2)	3 (2)	0	1 (1)	0	0	0	3 (2)
12	48.4	0	0	6.2	6.9	0	2	1 (0)	3 (1)	4 (3)	0	0	0	0	0	0
13	2.1	0	0	0	0	0	6	0	0	1 (0)	11 (9)	0	0	0	1 (1)	0
14	56.5	0	0	0	0	0	8	19 (13)	0	6 (2)	3 (1)	0	1 (1)	2 (0)	0	0
15	0	0	0	0	0	0	5	0	0	0	1 (1)	12 (9)	0	0	0	0
16	47.0	0	0	0	0	0	9	5 (4)	0	5 (3)	5 (2)	2 (2)	3 (3)	2 (0)	0	0
17	30.7	0	0	0	0	0	37	5 (4)	1[a] (0)	54 (17)	6 (4)	8 (5)	5 (5)	0	3 (3)	0
18	30.7	0	0	0	0	0	3	1 (1)	0	1 (1)	0	0	0	0	0	0
19	12.8	0	0	0	0	0	3	0	0	1 (0)	5 (1)	2 (1)	0	0	0	1 (0)
20	18.1	0	0	0	0	0	4	0	0	1 (0)	0	0	0	0	0	0
21	7.4	0	0	0	0	0	5	2 (2)	0	9 (2)	0	3 (1)	0	0	0	0
Total		0	42.9	86.8	72.9	56.1	164	58 (43)	57 (36)	196 (60)	39 (24)	41 (22)	10 (10)	10 (0)	13 (6)	4 (2)

Table 5.3. *Mean clutch size, brood size, and nest success in northeast Scotland and in other UK studies.* [E] *= England;* [S] *= Scotland.*

Study area	Years	Nests	Mean clutch size	Mean brood size	Mean fledglings per successful brood	Mean nest success	Reference
Cornwall [E]	1933–34	38	4.00	-	3.60	-	Ryves & Ryves 1934
Cornwall [E]	2006–08	200	-	-	3.30	54%	Setchfield 2012
East Anglia/Midlands [E]	1943–93	212	3.59/4.39 [a]	3.02/4.36 [b]	-	-	Crick 1997
Great Britain	1943–93	564	-	-	-	25/61% [c]	Crick 1997
Lancashire [E]	1999	19	4.58	4.11	3.55	c.50-60% [d]	Murphy 2000
Lincolnshire [E]	1994	28	3.82	3.28	3.00	48%	Gillings & Watts 1997
Sussex [E]	1995–97	120	4.65	4.45	4.45	25%	Brickle 1998
Sutherland [S]	1957–64	27	3.94	3.37	c.3.00	c.60%	Macdonald 1965
Western Isles [S]	1989–90	72	3.75/4.00 [e]	-	2.75/3.00 [e]	-	Hartley 1991
Western Isles [S]	1987–90	211	-	-	-	56%	Hartley & Shepherd 1994b
Northeast Scotland [S]	**2004–09**	**86**	**3.31 [f]**	**3.11**	**2.93**	**-**	**Present study**
Northeast Scotland [S]	**2004–08**	**334**	**-**	**-**	**-**	**37%**	**Present study**

[a] *non-significant increase over time, mean values shown are for 1955 and 1993.*
[b] *significant non-linear increase over time, mean values shown are for 1966 and 1991.*
[c] *mean values shown are for the decades with the lowest (1960–69) and highest (1990–93) nest success.*
[d] *recorded separately for two sites – 37% (n = 13 nests) and 85% (n = 6 nests).*
[e] *separate mean values shown for first nests and repeat or second nests.*
[f] *based on brood size plus number of unhatched eggs during nest visits at the chick stage, so likely to be an underestimate.*

Table 5.4. *Worked example calculating the predicted effect of delayed mowing on whole-season breeding productivity, incorporating multiple nesting attempts per female. Calculations are based on each female making three nesting attempts (maximum two successful), second attempts always being in the habitat used for the first attempt, and all third attempts in spring cereals. The example used is farm 4 (population = 29 breeding females), with comparison between conventional cutting dates (SIL) and delayed mowing until 1 August (DM), with all females that nest in forage grass exposed to the same mowing regime (SIL or DM). The effect of delayed mowing in this example is a 16.8% increase in total broods fledged (from 36.14 to 42.22).* ***Bold*** *= successful broods in A1, A2 or A3 that are summed to give the total broods fledged per year (B).*

Breeding parameter	Breeding parameter values by field type							Summed values for each regime	
	Forage grasses		SC		AC	ROU	PAS		
	SIL	DM						SIL	DM
% nesting females	41		41		10	5	3	100	
Fledging rate (FL)	0.08	0.43	0.61		0.53	0.60	0.39		
	First nesting attempt (A1)								
No. nesting females (NF)	11.90		11.90		2.97	1.49	0.74	29	
No. fledged A1 (FL x NF)	**1.00**	**5.08**	**7.30**		**1.56**	**0.89**	**0.29**	**11.04**	**15.12**
No. failed A1 (1-FL x NF)	10.90	6.82	4.60		1.41	0.60	0.45	17.96	13.87
	Second nesting attempt (A2)								
No. fledged A1+**fledged A2**	**0**	**0**	**4.48**		**0.82**	**0.53**	**0.12**	**5.95**	**5.95**
No. fledged A1+failed A2	1.00	5.08	2.82		0.74	0.36	0.17	5.08	9.17
No. failed A1+**fledged A2**	**0.91**	**2.91**	**2.82**		**0.74**	**0.36**	**0.18**	**5.01**	**7.01**
No. failed A1+failed A2	9.93	3.91	1.78		0.67	0.24	0.27	12.89	6.87
	Third nesting attempt (A3)								
			SIL	DM					
No. fledged A1+failed A2+**fledged A3**	**-**	**-**	**3.13**	**5.63**	**-**	**-**	**-**	**3.13**	**5.63**
No. fledged A1+failed A2+failed A3	-	-	1.96	3.54	-	-	-	1.96	3.54
No. failed A1+fledged A2**+fledged A3**	**-**	**-**	**3.07**	**4.30**	**-**	**-**	**-**	**3.07**	**4.30**
No. failed A1+fledged A2+failed A3	-	-	1.94	2.71	-	-	-	1.94	2.71
No. failed A1+failed A2+**fledged A3**	**-**	**-**	**7.94**	**4.21**	**-**	**-**	**-**	**7.94**	**4.21**
No. failed A1+failed A2+failed A3	-	-	4.95	2.66	-	-	-	4.95	2.66
Total broods fledged (B)	1.91	7.99	28.74	28.74	3.12	1.78	0.59	36.14	42.22

Table 5.5. *Predicting the population effect on each farm of delayed mowing until 1 August (DM), compared with conventional cutting dates (SIL). Mean annual number of breeding adults (assuming an equal sex ratio) and observed proportion of first nests in forage grasses across all years are used. In each scenario, all females nesting in forage grasses are exposed to the same mowing regime (100% in SIL or 100% in DM). The modelled percentage increase in broods fledged with delayed mowing, and percentage increase required for population stability is shown. Calculations of population change are based on an adult annual survival rate of 0.58, juvenile survival 0.29, and 2.9 young fledged per brood, and no immigration or emigration. Thus, across all farms (bottom row of table) under SIL: adult mortality = 308 x (1-0.58) = 129.36; recruitment = 191.47 x 2.9 x 0.29 = 161.02; Net change year ^{t} to year $^{t+1}$ = 161.56 – 129.36 = 31.66; Adult population in year $^{t+1}$ relative to year ^{t} = (308 + 31.66) / 308 = 1.1028 = 10.28% increase.*

Farm	No. of breeding adults	% first nests in FG	Predicted broods fledged	% increase in broods fledged with DM to 1 August		Population change year ^{t} to year $^{t+1}$	
			SIL	Predicted	For stable population	SIL	DM
Farms for which the proportion of first nests in forage grasses exceeds 50%							
3	12	83	5.37	48	12	- 4.4	+13.5
6	10	63	5.44	29	0	+3.7	+17.2
7	18	73	8.92	38	1	-0.3	+13.8
8	10	80	4.69	43	9	-3.5	+14.6
9	10	100	3.87	66	29	-9.5	+12.1
10	10	80	4.71	43	6	-2.4	+14.8
14	16	76	7.57	41	6	-2.2	+14.2
Total	**86**	**61**	**40.47**	**43**	**6**	**-2.4**	**+14.5**
Farms for which the proportion of first nests in forage grasses is 50% or less							
1	14	50	8.34	21	0	+8.1	+18.9
4	58	41	36.14	17	0	+10.4	+19.2
5	4	0	3.21	0	0	+25.6	+25.6
11	6	29	3.34	26	0	+4.8	+17.1
12	4	25	2.80	9	0	+16.9	+22.3
13	12	0	9.00	0	0	+21.1	+21.1
15	10	0	7.91	0	0	+24.5	+24.5
16	18	29	11.58	11	0	+12.1	+18.3
17	74	11	54.11	4	0	+19.5	+21.8
18	6	50	3.58	21	0	+8.1	+18.9
19	6	0	4.63	0	0	+22.9	+22.9
21	10	40	6.36	16	0	+11.5	+20.1
Total	**222**	**29**	**151.00**	**9**	**0**	**+15.2**	**+20.5**
Overall	**308**	**39**	**191.47**	**16**	**0**	**+10.3**	**+18.8**

Figure 5.1. *Timing of mowing of forage grass meadows (first cut) and harvesting of cereals (excluding arable silage) in northeast Scotland, with approximate main period in England and Wales shown for comparison. n = total number of fields.*

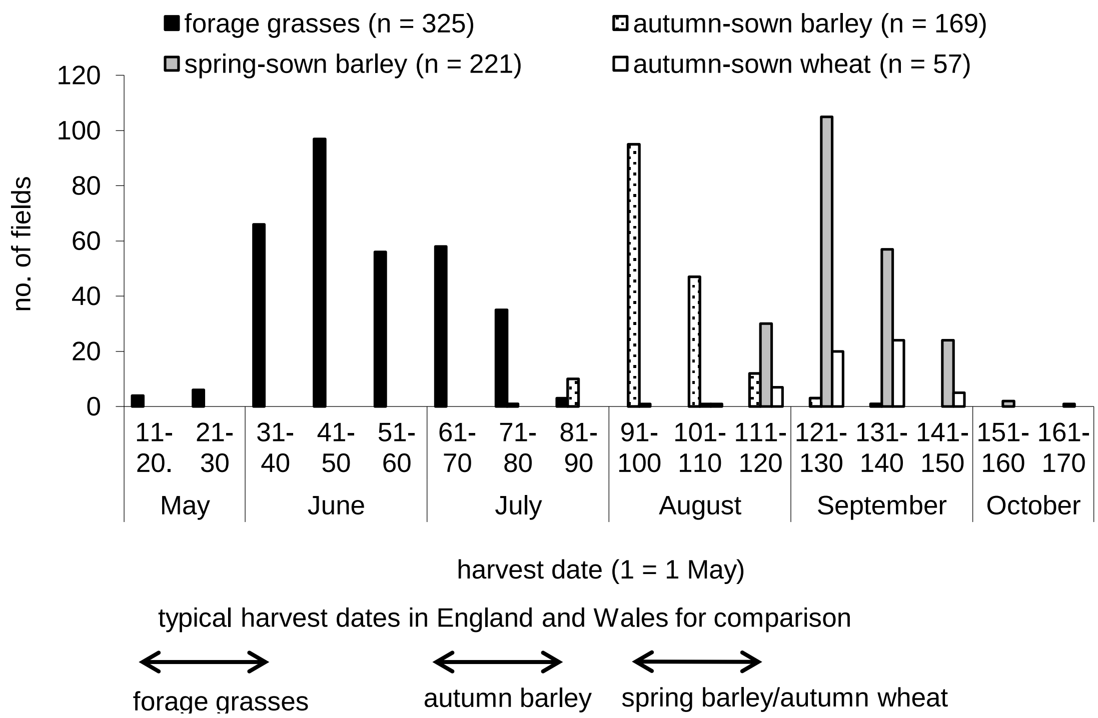


Figure 5.2. *First-egg dates of nests in forage grass meadows and spring- and autumn-sown cereals (excluding arable silage and unharvested crops), and approximate range of first-egg dates in those crops in southern England shown for comparison.*

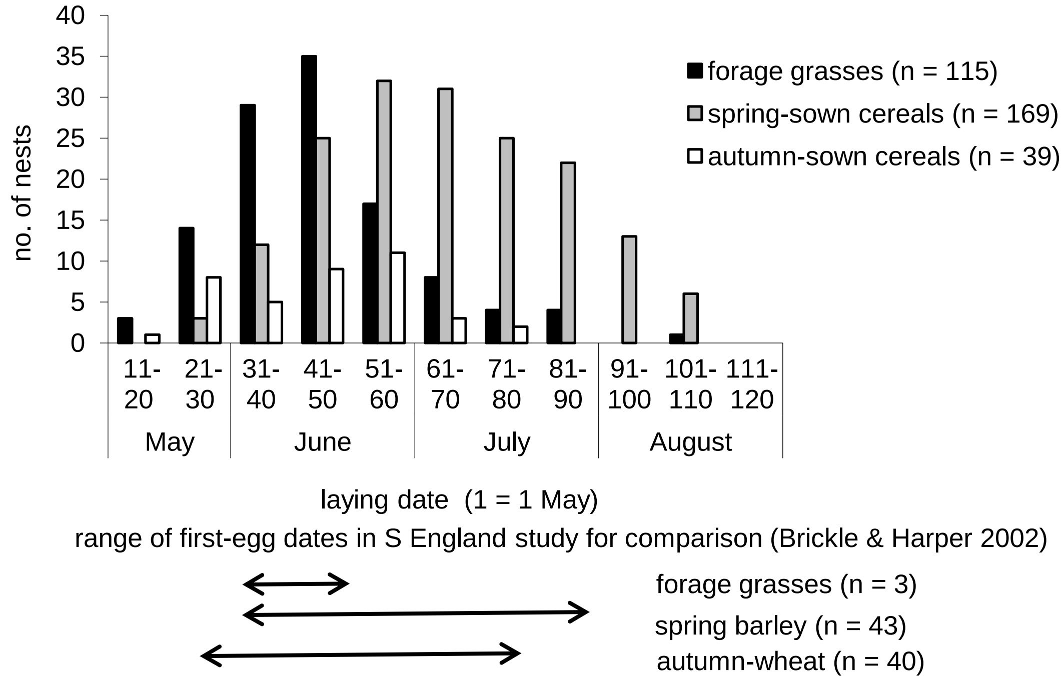


Figure 5.3. *Probability of broods fledging (≥ 1 chick alive at 15 days old) in conventionally managed forage grasses (SIL) compared with meadows with delayed mowing (DM) and other habitats (SC = spring cereals including arable silage and unharvested crops; AC = autumn cereals; ROU = rough grass and set-aside; PAS = pasture; VEGRD = vegetables, oilseed rape and daffodils; RES = newly re-seeded grass). Mean ± 95% confidence limits are shown. n = number of nests in the analysis.*

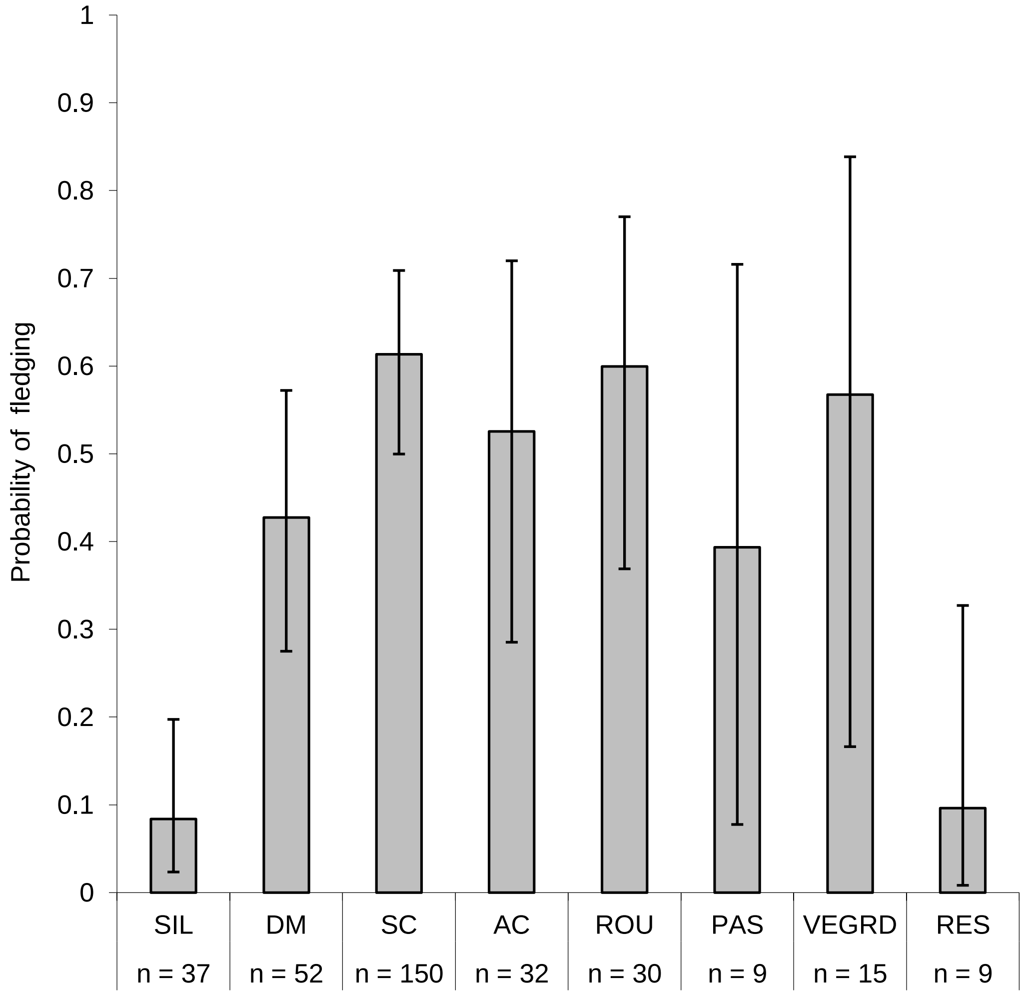


Figure 5.4. *Modelled proportion of nests in forage grass meadows that were active (and therefore vulnerable to destruction by mowing) on each date throughout the breeding season, and the cumulative proportion of broods fledged (safe from mowing) on each date. To produce the 'nests active' curve, estimated FEDs and underlying daily nest survival (i.e. excluding losses to mowing) in forage grass meadows were combined, and to produce the 'nests fledged' curve, estimated fledging dates and daily nest survival in DM were used. Six late season nests initiated after the first cut were not included in calculations.*

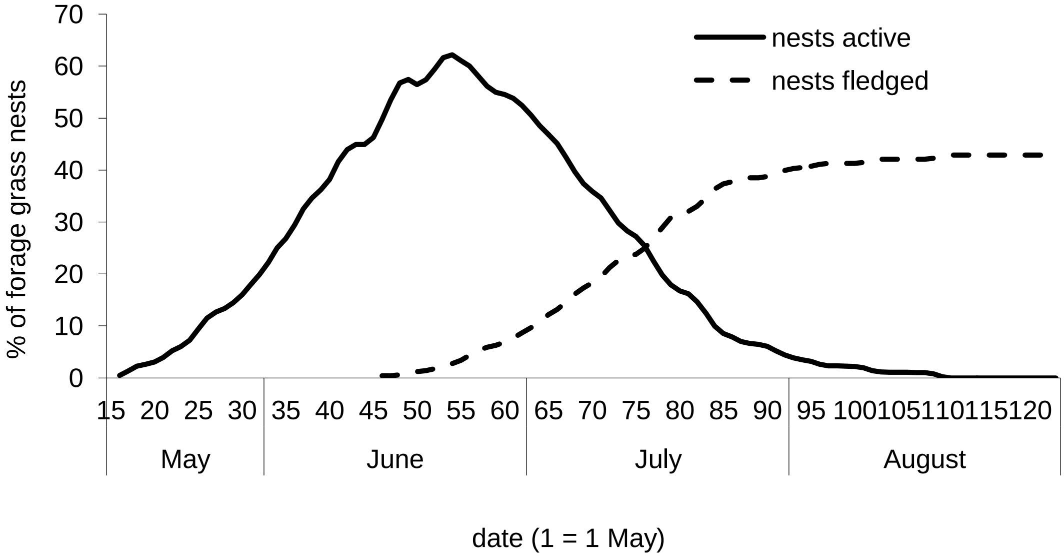
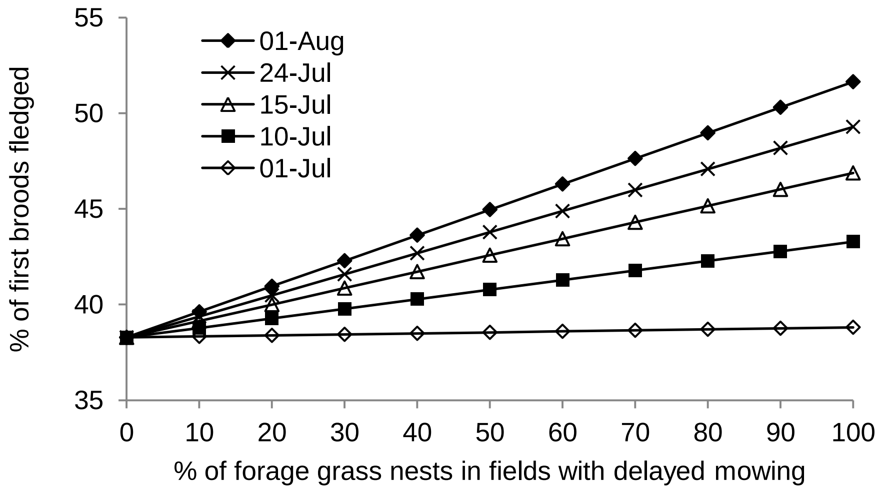
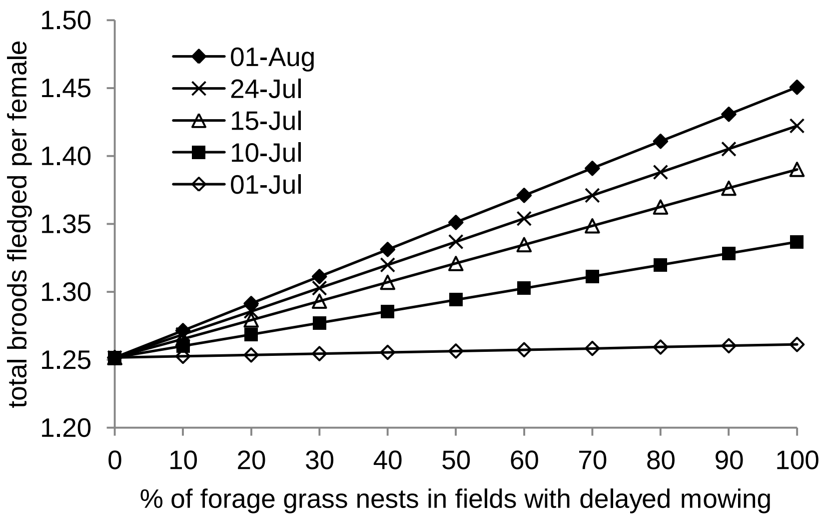


Figure 5.5. *Predicted percentage of first broods fledged for a range of mowing dates and proportions of forage grass nests in meadows with delayed mowing, across 19 farms with a mean annual breeding population of 154 territorial males. The plots in (a) were produced by combining the fledging rate and the proportions of known first nesting attempts in each habitat with various fledging rates in forage grass meadows, calculated for a range of delayed mowing dates and proportions of forage grass nests in meadows with delayed mowing (from 0–100%). Baseline on y-axis = 59 first broods fledged, or 0.38 per female. Plots of overall productivity in (b) incorporates multiple nesting attempts based on a simple model where each female can rear a maximum of two broods from up to three nesting attempts, the first two in the habitat first used, with the third attempt in spring cereal (SC). Baseline on y-axis = 193 broods fledged, or 1.25 per female.*

a) *Broods reared from first nesting attempts only.*



b) *All broods, including those from repeat and second attempts (assuming each female lays up to 3 clutches per season, and fledges a maximum of 2 broods).*



CHAPTER 6. WINTER HABITAT USE BY CORN BUNTINGS AND OTHER SEED-EATING BIRDS

6.1. Introduction

Outside of the breeding season, Corn Buntings are predominantly granivorous, specialising in cereal grains and the seeds of grasses and arable weeds, foraged mainly from the ground and in low vegetation (Wilson *et al*. 1999, Brickle & Harper 2000, Perkins *et al*. 2007). Several other UK and European farmland species of conservation concern (e.g. larks, sparrows, buntings, and some of the finches) have a similar diet and ground-foraging behaviour (see Chapter 1). Many studies have shown the importance of post-harvest crop stubbles and fallow land as foraging habitats for these species during winter (e.g. Wilson *et al*. 1996, Buckingham *et al*. 1999, Moorcroft *et al*. 2002, Hancock & Wilson 2003, Suárez *et al*. 2004, Orlowski 2006).

Cereal stubbles (Plate 16) and fallows are especially favoured by Corn Buntings, as shown by studies in the UK (e.g. Donald & Evans 1994, Brickle & Harper 2000, Mason & Macdonald 2000a) and in other European countries (e.g. Diaz & Tellaria 1997, Stoate *et al*. 2000, Orlowski 2006). In a national survey of wintering Corn Buntings in Britain, Donald & Evans (1994) reported that 50% of 222 flocks and 60% of almost 3000 birds were in stubbles, and this was the most strongly selected habitat. Further, weed-rich stubbles held twice as many birds and flocks as 'clean' stubbles. Whilst Corn Buntings generally avoided winter cereals and improved grasslands, they used other habitats such as bare fields and unimproved or semi-improved grasslands in proportion to their availability. Two recent studies in southern England showed seasonal shifts in habitat use by Corn Buntings as the winter progressed. In one study, birds selected ploughed land in early winter, stubbles in late winter, and showed strong selection of grassland throughout the winter (Mason & Macdonald 2000). In another study, they switched from stubbles in early winter to cattle-grazed fields from late December, following the introduction of feed troughs containing cereal grain for the cattle, and finally to spring-sown barley fields from mid-February onwards where they fed on the newly drilled grains lying on the soil surface (Brickle & Harper 2000).

In recent decades, changes in land use associated with agricultural intensification have resulted in a reduction in winter seed food availability for farmland birds, which is thought to have contributed to the population decline Corn Buntings in eastern Scotland (see Chapter

2), and of several granivorous species throughout the UK (e.g. Peach *et al*. 1999, Siriwardena *et al*. 1999). The widespread switch from spring to autumn-sown cereals has removed overwinter stubbles from large areas of the farming landscape, whilst on many farms polarisation of farming systems has seen the removal of livestock, along with associated overwinter fodder crops and grain feed (see Chapter 1). For example, in Scotland, one of the main causes of the Corn Bunting's range contraction from northern and western areas, including extinctions on most of the islands, has been the loss of cereals in favour of sheep grazing and grass silage-based feed for cattle (Forrester *et al*. 2007 and see Chapter 2). The only population remaining in western Scotland is in crofting areas of the Western Isles where cereals are still grown for cattle feed, but even here Corn Buntings are declining due to loss of winter food associated with changes in cereal management (Wilson *et al*. 2007b). Traditionally, cereals were harvested once fully mature and stored as stacks made of ripe sheaths, but since the 1980s there has been an increase in the practice of harvesting cereals early before grains ripen, stored as bales wrapped in plastic to make arable silage.

All of the main UK agri-environment schemes include management options designed to provide overwinter seed food for farmland birds, and one such option involves sowing mixtures of seed-bearing plants during spring and leaving them standing for one or two winters (Stoate *et al*. 2004). In Scotland, this management option is known as *unharvested crops* (SEERAD 2003; Plate 17) but is known elsewhere as wild bird cover, wildlife seed mixture or game cover, as its traditional use was to provide cover and food for gamebirds (Pheasant *Phasianus colchicus*, Grey Partridge and Red-legged Partridge *Alectoris rufa*). Agri-environment schemes also provide post-harvest stubbles retained overwinter as a management option.

In this chapter, we[6] first tested whether the provision of seed-bearing crops in Scottish agri-environment scheme options was effective in providing over-winter seed for birds, by measuring the relative use by granivorous birds of unharvested crops and other seed-rich habitats on farms in eastern Scotland. Further, to determine how best to manage seed-bearing crops for birds, especially when targeting the option at individual species, we measured seed availability in crop patches at different ages of establishment, and compared bird use of these

[6] This study was undertaken with the help of several colleagues. I designed the monitoring protocol for RSS and non-scheme farms and undertook all surveys on these 39 farms. The 14 FBL farms were surveyed by Hywel Maggs, Alan Bull, Karen Cunningham and Andy Wight. With advice from Jeremy Wilson, I designed and carried out all statistical analyses, and as lead author of the published paper (Appendix 3), wrote the first draft and incorporated improvements suggested by the co-authors (Jeremy Wilson and Hywel Maggs) and by Richard Bradbury, Adam Watson, the journal editor and two anonymous referees.

patches between their first and second winter of establishment. Seasonal change in seed availability within unharvested crop patches, and in bird use, was also measured. We did not seek to compare bird populations at the farm scale before and after entry into agri-environment schemes, or between scheme and non-scheme farms, because most granivorous species in the winter are highly mobile and flocking, and such comparisons are best made using territorial birds during the breeding season (see Chapter 7).

6.2. Methods

6.2.1. Agri-environment schemes operating across study sites

In the UK, each of the devolved countries has its own agri-environment scheme, and in Scotland, from 2001–2006 (but since replaced – see Chapter 7) this was the Rural Stewardship Scheme (RSS). The RSS was a voluntary scheme that operated on a competitive basis, such that farmers had to apply if they wished to join the scheme. Each application was required to include a whole-farm plan that incorporated several of the 33 management options available (SEERAD 2003). Because the scheme was competitive, applications were ranked according to an overall score accumulated from points awarded for each management option proposed, and for habitats and species present on the farm that were UK or local conservation priorities. The proportion of applications accepted varied annually, according to the funds available and the number of applications submitted. Once approved, each agreement was for a minimum of five years. By 2006, there were 3280 RSS management agreements across Scotland, covering 1.3 million ha of land (Scottish Executive 2006). In addition, a more targeted intervention scheme, Farmland Bid Lifeline (FBL), has been operating in eastern Scotland since 2002, aimed specifically at Corn Buntings in eastern Scotland (see Chapter 2). FBL offers payments to a small number of farmers for similar management options to those available in the national government-run schemes.

6.2.2. Study sites

Between 2002 and 2005, surveys were carried out on 53 arable and mixed lowland farms covering 6074 ha in Aberdeenshire and Moray, eastern Scotland (Fig. 6.1). Of these, 23 were in the RSS, 14 were in FBL, and 16 were not in any agri-environment scheme. The RSS farms were selected from those that joined the scheme in autumn 2002, as part of a wider study to assess the effects of the RSS on Corn Bunting breeding populations (see Chapter 7). At the same time, non-scheme farms with similar land use were selected within 10 km of the

RSS farms. None of these farms had applied to join the RSS, and there was no evidence to suggest that they were any less likely to gain entry to the scheme, or differed in biodiversity. The FBL farms were selected as participants in the RSPB intervention scheme in 2001 because they held breeding Corn Buntings, and all of those in Aberdeenshire were surveyed.

Land use on the farms during winter included autumn-sown cereals, oilseed rape, turnips, linseed, vegetables (carrots and mixed vegetables) and their stubbles, soft fruit, cattle- and sheep-grazed pasture, ungrazed grass, and long-term (non-rotational) set-aside (Table 6.1). In addition, some fields were ploughed and left bare in preparation for spring-sowing of cereals from mid-March. Stubbles were managed as agri-environment options on four FBL farms, and turnips on six, but most such fields were part of conventional farm management. Unharvested crop patches were planted on 31 of the 37 farms in RSS or FBL, and were the main difference between these and the non-scheme farms. To improve the dataset to address the second aim of the study, additional surveys of unharvested crops were undertaken in winters 2002/03 – 2003/04 (13 patches on eight farms), 2004/05 (46 patches on 25 farms), and 2006/07 (nine patches on eight farms).

6.2.3. Characteristics of unharvested crop patches

Two types of unharvested crops were grown. One-year crops consisted of a cereal (barley, oats or triticale *Triticosecale*) and at least one oilseed (mustard *Sinapis* spp., oilseed rape, linseed, or sunflower *Helianthus annuus*) or quinoa *Chenopodium quinoa*, and remained standing for one winter only. Two-year crops also included kale *Brassica oleracea* and stood over two consecutive winters. Kale is a biennial that does not produce seed until the second winter. Seeding plants from crops grown in previous year were sometimes present, especially wheat or oilseed rape. All unharvested crops were spring-sown, and 74 of 162 patches sown during 2002 – 2004 were two-year crops. In the RSS, the maximum patch size was 1 ha, but in FBL larger plots were sown, and overall patch size ranged from 0.14 – 7.34 ha (mean = 1.38 ha, sd = 1.18). Management agreements required crops to remain standing until at least 15 March, after which they could be destroyed.

6.2.4. Bird monitoring

In each winter, farms were visited twice between November and early April. During each visit, the habitat type of every land compartment on a farm was recorded on a 1:25 000 map. The area of each compartment was obtained from digitised versions of these maps using

MapInfo Professional version 6. All seed-rich habitats (see Table 6.1 or 6.2) were surveyed by walking transects across each compartment. All granivorous passerines and gamebirds seen on or flushed from the ground (i.e. considered to be using the compartment, as opposed to birds simply flying over or only recorded in boundary features such as hedgerows) were recorded. To ensure a similar likelihood of detecting birds, the distance between transects varied between habitats, to account for differences in vegetation structure (Atkinson *et al*. 2006). Transects with 50 m separation were used for stubbles, and 20 m for turnips, vegetables and non-rotational set-aside. Unharvested crop patches were particularly tall and dense, so were watched for 10 minutes before walking slowly back and forth through the crop to within 10 m of each point. Double counting was minimised by taking into consideration birds that were flushed to another compartment or to other parts of the compartment being surveyed. All surveys were carried out at least one hour after sunrise and completed by at least one hour before sunset, and avoided periods of strong wind (> Beaufort Scale 4) or heavy rain.

6.2.5. Seed availability in unharvested crop patches

In two winters (2004/05 and 2006/07), seed abundance was measured in 35 two-year crop patches (18 first-winter and 17 second-winter) and seven one-year crop patches. First-winter patches were visited three times and second-winter patches twice between December and March. At ten random sampling points in each patch, the observer detected seeds by visually searching an area of 1 m radius for 30 seconds, and for each crop component (plant type sown as part of the crop), recorded seeds as ‘absent’ (0), ‘present’ (1) or ‘abundant’ (2), separately for seeds on the plant and on the ground. Abundance categories were then summed across the ten sampling points to give an overall seed score for each crop component.

6.2.6. Data analysis

Because birds tend to be aggregated into flocks during the winter, and because some species occurred only on a small proportion of compartments, distributions of counts (where a count is defined as the number of birds of a given species recorded in one compartment during one visit) were highly skewed with many zeros and a long tail of large counts. To overcome this problem, re-sampling procedures (Crowley 1992) were used to determine whether the total number of birds counted in a given habitat type differed from that expected if the birds were distributed randomly with respect to the area of each habitat type available (‘null’ model).

Counts and habitat areas from the two visits in each winter were combined, and analysis carried out where there were at least ten counts for a species in a given winter. In each analysis, 999 re-samples of the observed sample of counts were used as independent estimates of the expected distribution of birds across habitat types under the null model. Re-sampling was with replacement so that expected counts on different habitat types were independent. The distribution of the 999 re-sampled values, plus the observed data, was used to calculate the probability that the proportion of birds of that species counted in a given habitat type was no more extreme than predicted by the null model. If the observed value fell within either of the 2.5% tails of this distribution, then the two-tailed probability of it having arisen by random distribution of birds with respect to habitat type was < 0.05. Analyses were carried out in MINITAB release 14.

A general linear mixed model (GLMM), with binomial error structure, was used to analyse the effect of *Age* of unharvested crop patch, *Season* (early or late winter period), and the *Age|Season* interaction on probability of species presence. *Crop patch identity* and *Year* were fitted as random effects, to control for repeated measures of the same patches, and the presence (1) or absence (0) of a species in a patch at each visit was the response variable. Each of the ten species was analysed separately, combining data from four winters. Analyses were carried out using the GLIMMIX procedure in SAS version 9.1. Because it was measured only in two winters, the additional effect of *Seed abundance* was analysed separately for each of two species guilds classified by food preference (cereal grains: buntings and sparrows, and oilseeds: finches). These analyses were carried out as above, but with the addition of cereal grain or oilseed abundance in the model, and presence or absence of one or more individuals of the guild as the response variable.

Finally, we used a non-parametric two-sample test (the Kruskal-Wallis test) to compare the seed abundance scores of two-year unharvested crops at different ages of establishment. A separate analysis was conducted for each crop component. Further, we used a non-parametric paired sample test (the Wilcoxon signed rank test) for significant change in seed abundance scores of crop patches between visits throughout the winter. Separate analyses were conducted for each crop component and each crop age. Analyses were carried out in SAS version 9.1.

6.3. Results

6.3.1. Bird counts and habitat availability

More than 36 000 birds of ten species were recorded during the study (Table 6.2), and 82% were of the four most abundant species, Linnet, Skylark, Chaffinch *Fringilla coelebs* and Yellowhammer. Many other species were recorded, but these data are not presented. Unharvested crops and cereal stubbles were the two habitats most heavily used, and 28% of birds of the ten species recorded were in unharvested crops, despite this habitat occupying less than 5% of the seed-rich area surveyed. Cereal stubbles held 44% of birds and occupied 71% of the seed-rich area surveyed. Of 867 Corn Buntings recorded, 44% were in unharvested crops, 46% in cereal stubbles, 6% in other stubbles, 2% in non-rotational set-aside, and there were fewer than five birds each in farmyards, vegetables and turnips.

6.3.2. Re-sampling analyses

Results of the re-sampling analyses are given in Table 6.2. Unharvested crops were selected (used more than expected from their area) by nine species in at least one winter (Table 6.2), including Corn Bunting, and five species (Reed Bunting, Tree Sparrow, Linnet, Goldfinch and Greenfinch) recorded in greater numbers than in any other habitat. Cereal stubbles were selected by five species exploiting seeds in open habitats, including three (Yellowhammer, Skylark and Grey Partridge) recorded in greater numbers than in any other habitat.

6.3.3. Seed abundance in unharvested crop patches

Cereal grains were present in all 18 first-winter, but only four second-winter two-year unharvested crop patches, reflecting the presence of cereals only as a self-sown 'volunteer' in second-winter crops (Fig. 6.2). Kale was present in all 35 crop patches, but being a biennial, only shed seeds during the second winter. Mustard, quinoa, oilseed rape, linseed and sunflower were restricted to few crop patches and, as annuals, only provided seed in their first winter. Overall, the clear pattern was for the seed abundance of all components other than kale to be higher in the first winter than the second. Seed depletion of crop patches throughout the winter varied between components, but was significant for four (Fig. 6.3).

6.3.4. Bird use of unharvested crop patches

For five species (all four bunting and sparrow species, plus Skylark), probability of encounter was significantly higher in first-winter than second-winter unharvested crop patches (Table 6.3a), differences that were also apparent from count data (Fig. 6.4a). Chaffinch was the only species that showed the opposite relationship. Three finch species (Chaffinch, Greenfinch and Goldfinch) were more likely to be present in crop patches in early winter, and Skylark in late winter (Table 6.3a). For Linnet, occurrence was more likely in second-winter crop patches in early winter, and in first-winter patches in late winter. In most cases, differences in a species' mean counts between early and late winter reflected seasonal variation in presence/absence (Fig. 6.4b). Seed abundance had an additional effect on the probability of encounter of species guilds within unharvested crop patches. For the buntings and sparrows guild, presence was more likely in crop patches with high cereal grain abundance, and for finches, there was very weak evidence for a positive effect of oilseed abundance on probability of occurrence (Table 6.3b).

6.4. Discussion

6.4.1. Bird use of unharvested crops

Three winters of surveys showed that more than 10 000 birds of ten species used seed-bearing crops provided by agri-environment schemes. This was over a quarter of the birds recorded, despite crop patches occupying less than 5% of the area surveyed, and all species except for Skylark selected them in at least one winter.

For all three buntings, Tree Sparrow and Skylark, birds were more likely to be present in first-winter crop patches than second-winter patches. These species feed extensively on cereal grain (Wilson *et al*. 1999), and differences in patch use with crop age were most likely due to a greater abundance of cereal grain in first-winter patches than in second-winter patches. This was supported by the species guild analysis, which showed that buntings and sparrows were more likely to be found in crop patches with high cereal grain abundance, although Tree Sparrows and Reed Buntings also take the seeds of quinoa and oilseed rape present in first-winter crops (Henderson *et al*. 2004b). In contrast to buntings and sparrows, patch use by finches either did not differ with crop age (Linnet, Greenfinch, Goldfinch), or was more likely in second-winter patches (Chaffinch). Finches readily take brassica seeds (Wilson *et al*. 1999), so were able to exploit those of oilseed rape and mustard in first-winter patches, and kale in second-winter patches. The species guild analysis confirmed that patch use by finches was more strongly associated with oilseed abundance than with crop age. Similarly, the lack of relationship between probability of Grey Partridge in crop patches and crop age probably reflect this species' varied diet, including both cereal grains and the seeds of kale in first- and second-winter patches, respectively. Results of previous studies were similar to those presented here. For example, Parish & Sotherton (2004a) recorded higher bird densities in seed-bearing crops than in other habitats, with the exception of Skylark, and Henderson *et al*. (2004b) found significant selection of seeding kale, mustard, quinoa and oilseed rape by finches and Tree Sparrows, and of seeding cereals by buntings.

Seed-bearing crops in agri-environment schemes have the potential to provide large quantities of seed food for farmland birds during the winter. In Scottish agri-environment schemes, unharvested crops can be grown as either a one-year or two-year crop. From this study, we recommend that for grain specialists such as Corn Bunting and Yellowhammer, one-year crops are grown, or that patches of two-year crops are sown in alternate years to ensure that cereal grain is available in each winter. For species with a more generalist diet, such as Tree Sparrow and Reed Bunting, or those that favour oilseeds, such as finches, sowing either a one-year or a two-year crop would be appropriate.

Seed-bearing crops may also provide seed and insect food for birds during the summer (Parish & Sotherton 2004b, Pywell *et al*. 2007), but this is of secondary importance to the provision of seed during winter. Although not considered in this study, to maximise bird use, crop patches should be located next to cover or a safe retreat such as a hedge, bushes, isolated trees or overhead wires that the birds can use between feeding bouts within the crop. By experimentally manipulating seed density at varying distances from a hedgerow, Robinson & Sutherland (1997) showed that Corn Buntings and Yellowhammers preferentially foraged closer to cover, suggesting a strong influence of perceived predation risk on foraging locations for these species. Locating crop patches next to hedgerows may also deter large species such as geese, whose flocks may destroy the crop and consume the seed source quickly in some areas.

6.4.2. Seed depletion and seasonal bird use of unharvested crops

Some crop patches retained very few seeds at the end of the winter, as a probable consequence of the combination of depredation by birds, mammals and insects, and rotting or germination of seeds when shed. In accordance with previous studies (Henderson *et al*.

2004b), quinoa and kale showed the greatest retention of seeds into late winter whilst seeds of cereals, oilseed rape and mustard were significantly reduced by late winter. For some species, birds were more likely to be present (e.g. Chaffinch, Greenfinch, Goldfinch), or counts higher (e.g. Tree Sparrow, Linnet, Yellowhammer), in seed-bearing crops during the early than late winter period. This may at least partly have been due to a reduction in the availability of cereal and brassica seeds, and other studies have shown a decline in crop use by birds following a mid-winter peak in response to seed depletion, although this varied with crop type (Stoate *et al*. 2004, Sage *et al*. 2005).

However, seasonal variation in presence/absence did not always reflect counts (e.g. Chaffinch, Yellowhammer), suggesting that flock size in crop patches may have differed seasonally. Weather conditions such as temperature and snow cover, and changes in availability of other seed-rich habitats such as the ploughing of stubbles also affect seasonal bird use of seed-bearing crops. The latter may explain why patch use by Skylarks was most likely during the late winter period, although the effects of weather may also have created a more attractive, open crop structure in some patches in late winter. Although not statistically significant, patch use by Yellowhammer and Grey Partridge also tended to be higher in late winter. In an experimental study providing supplementary seed food in farmland, Siriwardena *et al*. (2008) found that use of feeding stations by Yellowhammer, Reed Bunting and Chaffinch peaked in February or later, when few cereal stubbles remained unploughed, and most game cover crops were destroyed following the end of the shooting season.

6.4.3. Farming practicality of unharvested crops

Seed-bearing crops are popular with some farmers because patches are small and are separate from commercially grown crops, and the payment is high relative to other management options (Boatman *et al*. 2007). Unsurprisingly, they are particularly attractive to farmers with game-rearing or shooting interests. However, there are potential problems with seed-bearing crops that may deter some farmers from sowing them. Crops can attract pests such as Brown Rats *Rattus norvegicus*, Woodpigeons and corvids, and act as a reservoir for insect pests, weeds and disease, although such problems can be alleviated through careful location and rotating crop patches around the farm (Pywell *et al*. 2007). Another potential problem is that farmers sometimes view land managed through agri-environment schemes as low priority and requiring little management compared with land managed conventionally for food production. Unlike commercial crops, payments for seed-bearing crop patches are not based on yield, and there is no financial incentive for farmers to grow crops that produce large quantities of seed. This was evident from some farms in the RSS where unharvested crops had failed to establish and produced few seeds, often due to patches sown late in spring and on poor ground, and in some cases, despite being a requirement of RSS agreements, patches were not re-sown on at least a twice-yearly cycle. If these issues are widespread (5% of farms in Scottish agri-environment schemes are inspected by government officials per year, but information on the recorded rate of non-compliance is unavailable), it could severely limit the effectiveness of the option, and other studies have reported similar problems relating to deployment of agri-environment options (e.g. Chamberlain *et al*. 2009). One possible solution would be to allow payment for leaving a proportion of a commercially grown crop unharvested. In schemes targeted at buntings and sparrows, this would be a cereal, and where finches are the target species, a brassica such as oilseed rape.

6.4.4. Bird use of other seed-rich habitats

Whilst birds heavily used seed-bearing crops, this study also reaffirmed the importance of other seed-rich habitats. Cereal stubbles were strongly selected by species exploiting grain available in open habitats (e.g. Skylark, Yellowhammer), and brassica (oilseed rape) stubbles and turnips by at least one species specialising in oilseeds (Linnet). Open-country species such as Skylark prefer to feed in stubbles than in the tall, dense vegetation typical of seed-bearing crops. However, the higher frequency of encounter in crop patches in late winter accords with observations that Skylarks tend to feed closer to the edge of stubble fields as the winter progresses (Robinson & Sutherland 1999), perhaps reflecting the effects of food depletion in field centres. In other parts of Europe, stubbles are also important for large granivorous species such as Great Bustard *Otis tarda* (Lane *et al*. 2001). Some species selected seed-bearing crops but not stubbles, and a combination of the two is most likely to provide foraging habitats for the widest range of species. Even when crops were spring-sown, many stubbles were ploughed during autumn and winter and left bare prior to sowing (Fig. 6.5), demonstrating the need for agri-environment incentives for farmers to retain stubbles through the winter. Similarly, Mason & Macdonald (2000a) also found that despite 26% of their study area in southeast England being under spring cultivation, by early January only 3–4% of the land area remained as stubble.

Alternative options that provide winter seed food for birds include extensively managed fodder root crops, in which broad-leaved weeds are left to set seed (Hancock & Wilson 2003), or the direct provision of supplementary grain or seeds (Siriwardena *et al*. 2007). In grass-dominated farmland, manipulation of cutting or grazing regimes to allow rye-grasses to set seed can also provide winter food, as recent trials have shown for Yellowhammers and Reed Buntings (Buckingham *et al*. 2011). In some parts of Europe, medium to long-term fallow can support high densities of wintering seed-eating birds (e.g. Suárez *et al*. 2004, Orlowski 2006), especially where agricultural practices are less intensive (Van Buskirk & Willi 2004). In our study, Skylark and Linnet selected non-rotational set-aside where they exploited grass and weed seeds, and even on intensively managed land, short-term (two-year) fallow can provide large quantities of seed from volunteer cereals and weeds (Stoate & Moorcroft 2007). For Corn Buntings and other cereal grain specialists, grains lying on the surface of spring-sown cereals also provide an abundant food source at the end of the winter, when seed resources in other habitats, including seed-bearing crop patches, may have been depleted (Brickle & Harper 2000).

6.4.5. Population effects

Whilst habitat selection studies such as ours cannot determine the impact on over-winter survival and subsequent breeding populations, the high densities of birds recorded using seed-bearing crops suggest that the local benefits for farmland bird populations may be considerable. In the UK, research has shown that availability of over-winter stubbles can help to increase subsequent breeding densities of Skylarks and Yellowhammers (Gillings *et al*. 2005, Whittingham *et al*. 2005), and targeted provision of stubbles has contributed to halting and reversing the population decline of Cirl Buntings (Peach *et al*. 2001, Bradbury *et al*. 2008). Studies have also demonstrated that provision of supplementary seed food can increase over-winter survival of farmland birds (Hole *et al*. 2002), and positively influence local population trends (Siriwardena *et al*. 2007). However, only recently have studies begun to investigate the amount of seed-rich habitat required for a population effect, and how it should be distributed within the farming landscape (Siriwardena *et al*. 2006).

Throughout Europe, six of the nine species that selected seed-bearing crops in our study (Grey Partridge, Tree Sparrow, Linnet, Yellowhammer, Reed Bunting, Corn Bunting) have undergone long-term (1980–2009) population declines (PECBMS 2011 and see Chapter 1). In the UK, these are all conservation priority species (Gregory *et al*. 2002), and almost the entire populations of Corn Bunting and Grey Partridge are dependent on farmland. Given that, for the first time since its introduction in 1992, there was no compulsory set-aside in 2008 or since, it is likely that there has been a further reduction of seed-rich habitat on farms throughout the EU, especially of stubbles previously retained as rotational set-aside.

Therefore, it is crucial that such habitats are replaced to ensure that vulnerable and declining farmland bird populations can persist, and the targeted provision of seed-bearing crops and other seed-rich habitats through agri-environment schemes is the most effective way of achieving this.

In the next chapter, I measure the effect of deployment of agri-environment management, including the provision of seed-bearing crops, on Corn Bunting populations at the farm scale, and assess the potential for such schemes to halt and reverse declines across eastern Scotland, and the level of implementation required.

Table 6.1. *Farms surveyed and percentage area of land occupied by each habitat type (visit 1) in each winter.*

Scheme	Year	Farms	Area (ha)	CST [c]	OST [d]	TUR [e]	VEG	UC [f]	NSA	FY	PL	WC	OSR	GRA	OTH
FBL [a]	2002–03	14	1731	28.5	4.4	1.1	0	2.2	3.6	1.2	1.8	15.0	6.7	30.9	4.6
FBL	2003–04	14	1731	20.4	3.7	1.5	0	2.9	3.2	1.1	8.3	15.3	6.8	32.7	8.3
FBL	2004–05	13	1658	29.1	4.0	1.3	0	3.4	2.0	1.1	2.8	14.8	4.5	32.6	4.4
RSS [b]	2002–03	23	2736	24.2	0.5	0.7	0.1	0	1.5	1.1	10.2	19.8	4.1	30.0	7.8
RSS	2003–04	23	2736	20.4	1.1	0.1	0.1	1.3	2.6	1.1	11.3	18.5	5.9	30.8	6.8
Non-scheme	2002–03	16	1607	17.8	0	1.0	0	0	4.7	1.2	21.7	7.0	2.5	32.4	11.7
Non-scheme	2003–04	16	1607	25.6	0.8	1.6	0.6	0	3.3	1.2	19.1	12.2	0.6	33.0	2.0

Field types categorised as seed-rich habitats and surveyed for birds: CST = cereal stubble; OST = other stubble; TUR = turnips; VEG = vegetables; UC = unharvested crop; NSA = non-rotational set-aside; FY = farmyard.
Field types not surveyed for birds: PL = ploughed land; WC = winter-sown cereal; OSR = oilseed rape; GRA = grass; OTH = other, mostly non-farmed land.

[a] *options established on 11 FBL farms in summer 2002 and on a further three farms in summer 2003.*
[b] *options established on RSS farms in summer 2003.*
[c] *some cereal stubbles were FBL options (17 ha in 2002/03, 18 ha in 2003/04, 24 ha in 2004/05).*
[d] *some oilseed rape stubbles were FBL options (2 ha in 2002/03, 7 ha in 2003/04).*
[e] *some turnips were FBL options (9 ha in 2002/03, 15 ha in 2003/04, 15 ha in 2004/05).*
[f] *all unharvested crops were FBL or RSS options (25 patches on eight farms in 2002/03, 63 on 31 farms 2003/04, 31 on 13 farms 2004/05).*

Table 6.2. *Bird counts summed across visit and year for ten seed-eating species recorded in each seed-rich habitat, and results of re-sampling analyses. For each year analysed: proportion of birds using habitat greater (G) or less (L) than that expected if birds were distributed randomly with respect to the area of each habitat available (P < 0.05). For example, G, GG, GGG = significant selection of that habitat in one, two and three winters, respectively.*

	Cereal stubble	Other stubble	Turnips	Vegetables [b]	Unharvested crops	Non-rotational set-aside	Farmyards	Total
Corn bunting [a]	403	54	1	3	G 381	21	4	867
Yellowhammer	GGG 2175	0	9	G 20	GG 1128	39	GG 296	3667
Reed bunting	GG 567	16	G 57	0	GGG 1397	66	7	2110
Tree sparrow	L 289	0	51	0	G 543	33	GGG 420	1336
Chaffinch	685	GG 909	GG 989	G 20	G 955	82	GG 871	4511
Linnet	G 4857	G 1387	GGG 1880	G 99	GGG 5019	G 1281	121	14644
Greenfinch	L 151	11	61	G 1	GG 543	0	114	881
Goldfinch	LL 159	96	26	G 34	GG 194	59	G 118	686
Grey partridge	G 386	G 51	G 65	G 17	G 104	12	0	635
Skylark	GGG 6664	GG 254	G 197	0	124	G 277	1	7517
Total Count	16336	2778	3336	194	10388	1870	1952	36854
Area 2002/03 (ha) [c]	1039	91	57	2	38	178	68	1473
Area 2003/04 (ha) [c]	1322	108	55	11	87	180	66	1829
Area 2004/05 (ha) [c d]	483	66	20	0	57	33	18	677

[a] *insufficient data 2004/05.*
[b] *absent 2004/05.*
[c] *areas refer to visit 1 (some stubbles ploughed between visits).*
[d] *only FBL farms surveyed 2004/05.*

Table 6.3. *Results of GLMM analyses, showing the effects of crop age and season on probability of encounter of seed-eating bird species in unharvested crop patches, and the additional effect of seed abundance. N = number of crop patch visits (out of 237 and 66) on which the species or guild, respectively, was encountered. * P < 0.05; ** P < 0.01; *** P < 0.001.*

a) *Variation in bird use of unharvested crop patches with crop age and season.*

Species	N	Significant effects	Probability of encounter			
			First-winter crop		Second-winter crop	
			Early	Late	Early	Late
Corn bunting	16	Age*	0.073	0.052	0.005	0.003
Yellowhammer	57	Age**	0.241	0.326	0.071	0.104
Reed bunting	97	Age***	0.581	0.560	0.128	0.119
Tree sparrow	22	Age*	0.135	0.058	0.021	0.008
Chaffinch	61	Age* Season*	0.230	0.127	0.407	0.249
Linnet	58	Age|Season**	0.192	0.303	0.308	0.113
Greenfinch	33	Season*	0.177	0.071	0.169	0.067
Goldfinch	23	Season***	0.159	0.019	0.114	0.013
Grey partridge	33		0.120	0.205	0.054	0.097
Skylark	35	Age** Season**	0.115	0.304	0.011	0.036

b) *Additional effect of seed abundance on probability of encounter of guilds (cereal grains: buntings/sparrows, and oilseeds: finches).*

Guild	N	Additional seed score effect	
		Parameter estimate	*P*
Buntings/sparrows	29	+0.129	0.006
Finches	26	+0.080	0.070

Figure 6.1. *Map showing location of the RSS (squares), FBL (triangles) and non-scheme (circles) farms surveyed. Open triangles indicate additional FBL farms where only unharvested crop patches were surveyed.*

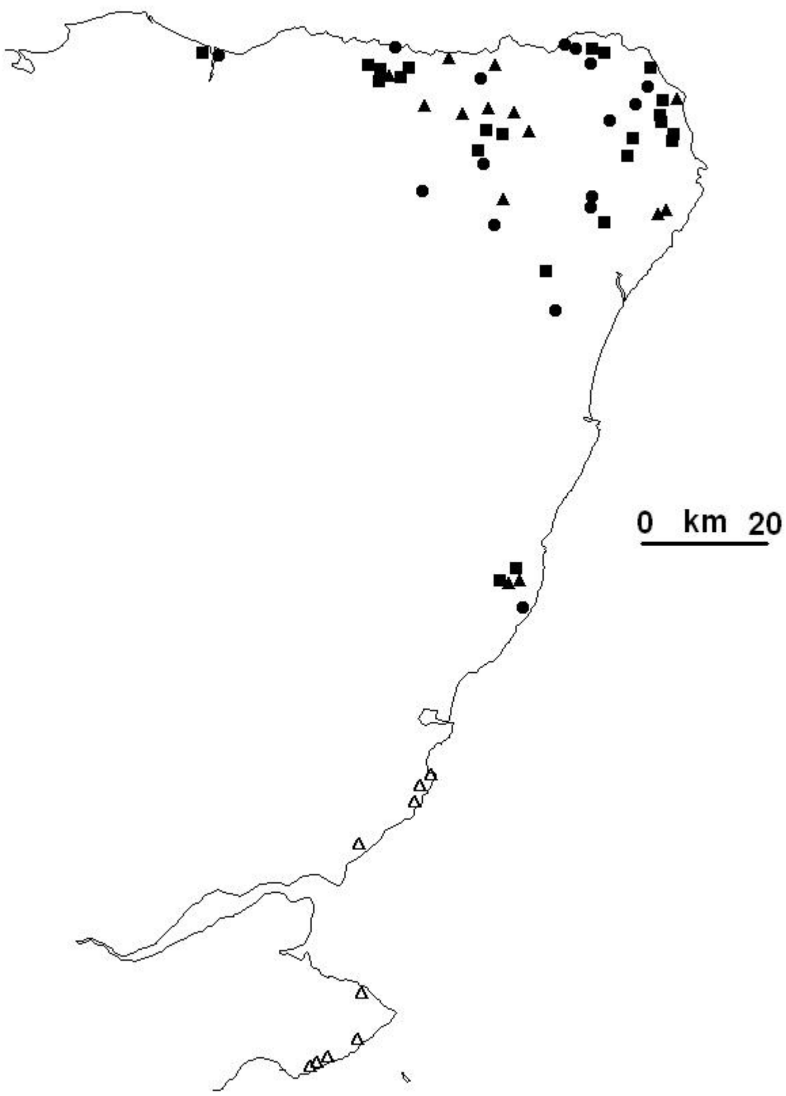


Figure 6.2. *Differences in seed availability between two-year unharvested crop patches in their first and second winter of establishment (mean seed abundance score on visit 1 ± 1 se). The number of crop patches in which each component was present (n) and the significance of Kruskal-Wallis tests are shown below each bar.*
** P < 0.05; ** P < 0.01; *** P < 0.001; ns P > 0.05.*

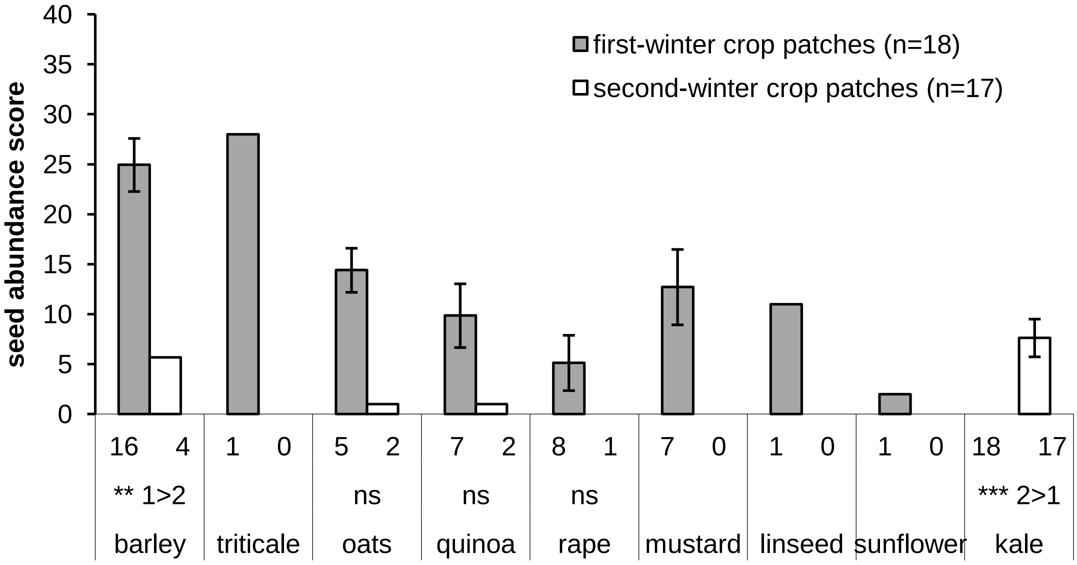


Figure 6.3. *Change in seed abundance within unharvested crop patches throughout the winter (mean proportion of seed abundance score on visit 1 ± 1 se). The number of crop patches in which seeds of each component were present on visit 1 (n), and the significance of Wilcoxon signed rank tests are shown below each bar.*
** P < 0.05; ** P < 0.01; *** P < 0.001; ns P > 0.05.*

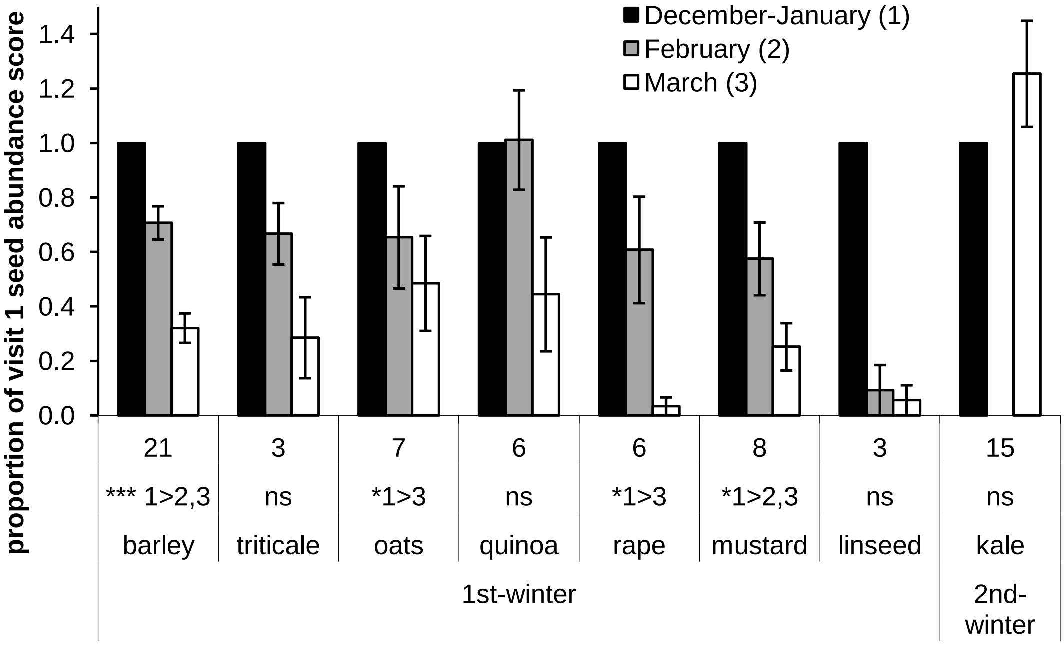
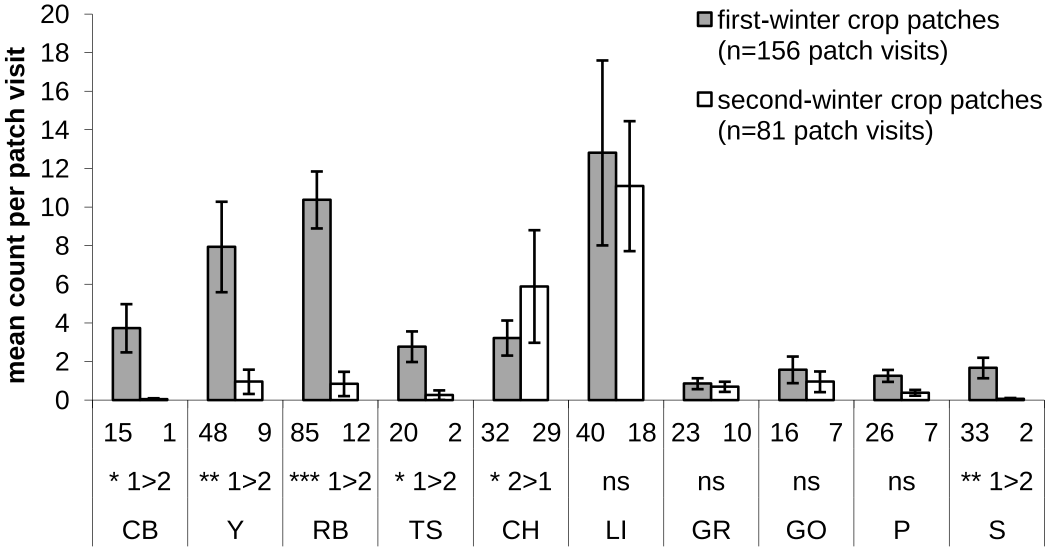


Figure 6.4. *Comparison of bird counts (including zero counts, mean ± 1 se) in two-year unharvested crop patches at different ages of establishment (first and second winter) and with season (early and late winter). For each species, the number of first- and second-winter crop patch visits on which it was present (n) is shown, along with the direction and strength of significant effects in the GLMM. * P < 0.05; ** P < 0.01; *** P < 0.001; ns P > 0.05.*

CB = Corn Bunting; Y = Yellowhammer; RB = Reed Bunting; TS = Tree Sparrow; CH = Chaffinch; LI = Linnet; GR = Greenfinch; GO = Goldfinch; P = Grey Partridge; S = Skylark.

a) *Age of crop patch (first and second winter of establishment).*



b) *Season (early and late winter).*

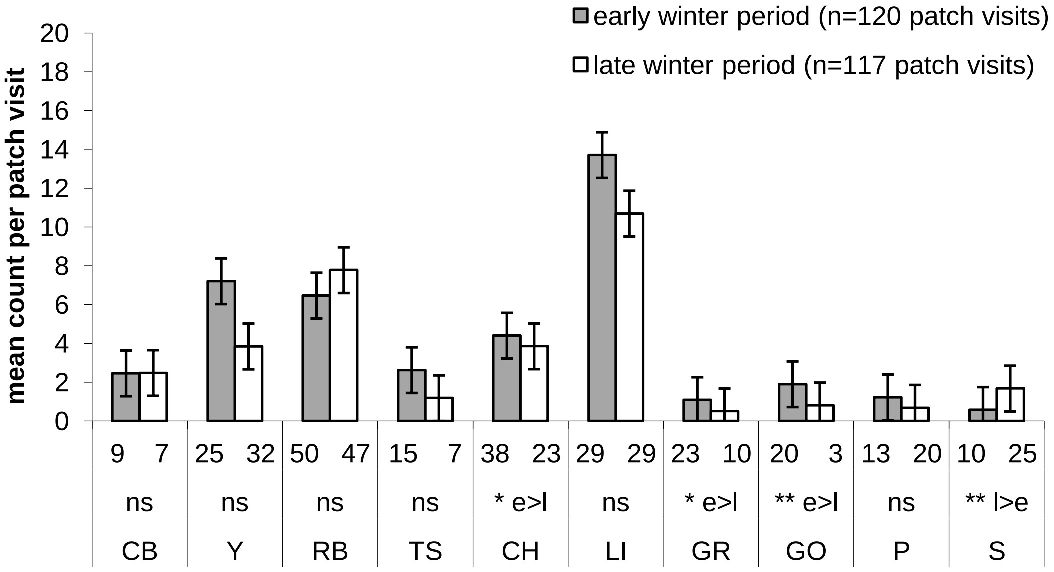
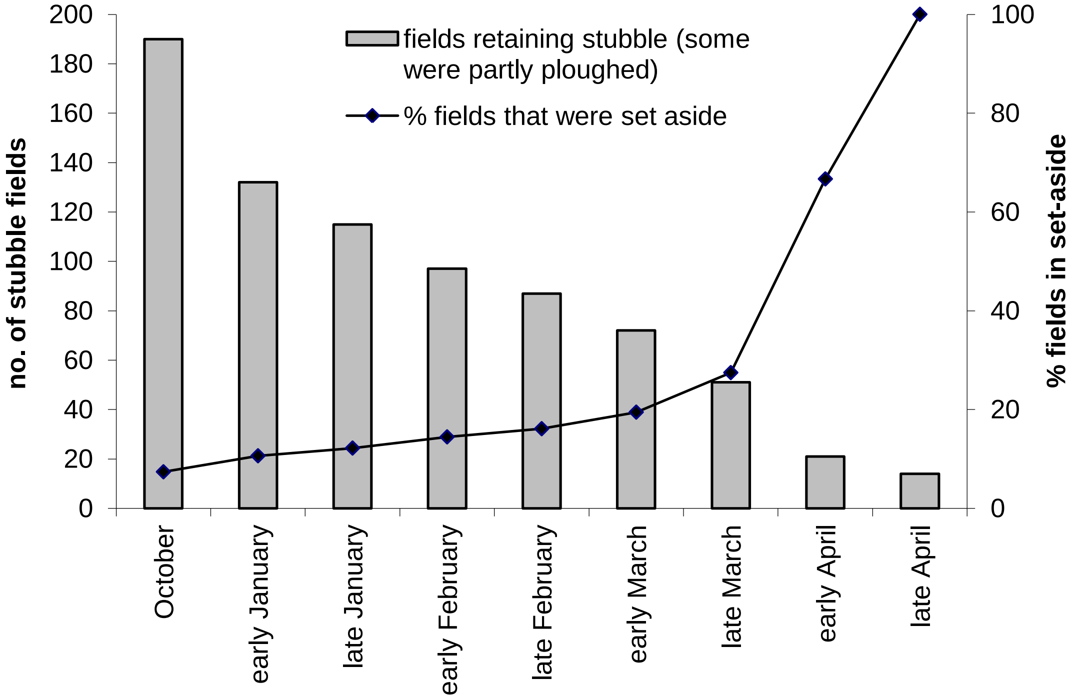


Figure 6.5. *Number of fields (grey bars) retaining stubble at approximately fortnightly intervals throughout the late-winter period, across 190 fields (182 cereal stubbles and 8 rape stubbles) in seven areas of Aberdeenshire during winters 2003/4 and 2004/5. The black line represents the percentage made up by rotational set-aside fields (i.e. fields retained as stubbles into the following summer).*



CHAPTER 7. POPULATION RESPONSE TO AGRI-ENVIRONMENT SCHEMES

7.1. Introduction

In Chapters 5 and 6, I showed how Corn Buntings and other farmland species had benefitted from management provided through agri-environment schemes, in terms of increased breeding success due to reduced nest losses in meadows with delayed mowing, and exploitation of winter seed food in unharvested crops and stubbles. Whilst many studies have shown positive effects of management on breeding success in farmland birds, and habitat associations in winter and more generally (see Chapter 1), few studies have demonstrated population increases in response to agri-environment management. This is despite a 'greening' of European Union (EU) agricultural policy since the late 1980s, involving vast financial expenditure on agri-environment schemes. A review by Kleijn & Sutherland (2003) found relatively little evidence of the effectiveness of agri-environment schemes (AES) for biodiversity conservation and, worse, found that many schemes lacked robust monitoring programmes to make such assessments possible. Since then, studies have demonstrated some benefits of AES interventions for a range of taxa (e.g. Kleijn *et al*. 2006, Knop *et al*. 2006, O'Brien *et al*. 2006, Pywell *et al*. 2006, Birrer *et al*. 2007, Carvell *et al*. 2007, Maes *et al*. 2008, Davey *et al*. 2010a), and maximising biodiversity conservation from AES is now a key policy challenge (Sutherland *et al*. 2006).

However, studies which quantify large-scale, long-term population response to AES remain scarce (e.g. Peach *et al*. 2001, Wilson A. *et al*. 2007, La Haye *et al*. 2010, Bretagnolle *et al*. 2011), and Memmott *et al*. (2010) note the continuing lack of applied ecological studies which implement and test effectiveness of management in an adaptive approach. Adaptive management that combines research with action on the ground, enabling practitioners to learn from successes and failures and adapt actions accordingly, is essential if we wish for better conservation (Salafsky *et al*. 2002). AES monitoring studies provide opportunities to carry out such tests, to understand how the design of measures may need to vary across species' ranges (Whittingham 2007), to adapt measures over time, and to estimate what proportion of a given species' population must be targeted to halt its decline (Wilson *et al*. 2010).

Here we[7] use a seven-year monitoring study of Corn Bunting population response to agri-environment management in eastern Scotland to test its effectiveness. We compared two different AES, one with general and one with targeted deployment of measures, across a wide range of arable and mixed farming systems. The targeted scheme was adapted during the study by incorporating a novel measure in response to initial monitoring results from the same study populations. We also use the observed annual growth rates of Corn Bunting populations on farms within and outside AES schemes to make initial estimates of the proportion of the total population that AES management will need to target in order for the national population decline to be halted and reversed. Finally, we also took the opportunity to monitor two other bunting species (Yellowhammer and Reed Bunting) during the study, to measure changes in their breeding populations in response to management in the general AES.

7.2. Methods

7.2.1. Agri-environment schemes used as the basis for the study

In Scotland, the national agri-environment scheme in 2001–2006 was the Rural Stewardship Scheme (RSS). The scheme was voluntary but competitive, such that not all applicants succeeded (for further details, see Chapter 6). Each application (for a five-year funding agreement) included a whole-farm plan incorporating several of 33 management options (SEERAD 2003), some of them designed to provide resources for farmland birds (Table 7.1; Plates 14–17). The RSS was our 'general' AES. Our 'targeted' AES was Farmland Bird Lifeline (FBL), an intervention scheme for Corn Buntings, operating since 2002. In FBL, farmers whose holdings were in areas with known breeding populations are invited to enter annually reviewed management agreements, backed by face-to-face advisory support. Management options are similar to those in the RSS, but have been adapted to the specific needs of Corn Buntings as our knowledge of the birds has improved. For example, because Corn Buntings use "unharvested crop" patches (Table 7.1) mostly in the first winter when

[7] Several colleagues helped with this study. I designed the monitoring protocol for RSS/control farms and undertook all surveys on approximately half of them (c.20 farms). All other farms were surveyed by Hywel Maggs, and seasonal fieldworkers Amanda Biggins, Alan Bull, Steven Coyne, Paul Doyle, Richard Firmin, Clive McKay, and John McMahon, with the assistance of Stuart Benn, Ken Bruce, Karen Cunningham, Chris Smout and Adam Watson. I designed and carried out all statistical analyses (with advice from Jeremy Wilson), and as lead author of the published paper (Appendix 4), wrote the first draft and incorporated improvements suggested by the co-authors (Adam Watson, Jeremy Wilson, Hywel Maggs), and by John Deag, Mark Whittingham, and two anonymous referees.

they are rich in cereal grain (Perkins *et al*. 2008a and see Chapter 6), these patches are sown annually with a cereal-rich mixture (since 2006) to ensure that grain is available throughout each winter. In the RSS, however, re-sowing was required only every two years. Secondly, findings from the first two years of nest monitoring studies (2004–2005) in Aberdeenshire revealed that, in contrast to elsewhere in the species' UK range (e.g. Brickle *et al*. 2000), up to 30% of first nesting attempts were in forage grass meadows managed for silage, with high nest-loss rates caused by subsequent mowing (see Chapters 4 and 5). Consequently, delayed mowing ("late-cut grass") was added to FBL from 2005 (Table 7.1; Plate 19), which has increased the fledging success of Corn Bunting nests in mown grasslands (see Chapter 5).

7.2.2. Farm selection

Between 2003 and 2009, surveys were carried out on 71 arable and mixed lowland farms covering 8845 ha in eastern Scotland (Aberdeenshire, Moray, Inverness-shire, Angus and Fife), and split between RSS, FBL and 'control' groups (Fig. 7.1). Because we were studying real agri-environment schemes and their uptake by farmers, the distribution of farms between treatment groups changed between years (Table 7.2).

The RSS farms were selected from those in Aberdeenshire and Moray that joined the scheme in autumn 2002, and were within or adjacent to a 2-km square recently occupied by Corn Buntings during the breeding season (Francis & Cook 2011). At the same time, non-scheme 'control' farms with similar land use to the RSS farms, and within 10 km of them, were also selected. Seven 'control' farms later joined the RSS (two each in autumns 2003 – 2004 and three in autumn 2005). Management options were established during the spring and summer following entry into the scheme. In autumns 2007 and 2008, management options were removed from seven and two farms respectively, following expiry of five-year RSS agreements, and these farms were considered as controls in subsequent years.

The FBL farms were selected in areas known to hold breeding Corn Buntings, and management options on 14 farms in Aberdeenshire, one in Inverness-shire, four in Fife and three in Angus were first implemented in the springs of 2002 and 2003. This was extended to a further seven farms in Aberdeenshire, two in Inverness-shire, three in Angus, and one in Fife in springs 2006 – 2007 (Table 7.2). Two of these farms had prior RSS agreements, and joined the FBL treatment group from 2006.

Land use (determined from digitised maps) was mainly a combination of autumn- and spring-sown barley and oats, autumn-sown wheat and oilseed rape, potatoes, turnips, grass mown for silage or grazed by cattle and sheep, and land left fallow as 'set-aside' (Table 7.3). In Fife and Angus, there was less grass, and more land was used for vegetables, including carrots, cabbages and broccoli.

7.2.3. Monitoring

All FBL farms were monitored in each year that AES options were implemented, and eight were surveyed as controls in at least one year prior to them joining FBL. It was not possible to monitor every RSS and control farm in every year due to other data collection commitments (see Chapters 4 and 5). However, all of these farms were surveyed in summers 2003, 2004, 2006 and 2008, with additional partial surveys in 2005 (n = 7 farms), and 2007 (n = 5 farms) during detailed studies of breeding Corn Buntings. In 2009, we surveyed 17 of these farms, together with eight farms not monitored since their transfer from FBL to RSS in 2004 – 2005. Our study therefore had strong 'control-intervention' design over several years, but weaker 'before-after' design. However, preliminary studies of Corn Bunting response to FBL management had previously shown that population changes were consistent across control and intervention study areas before management began (Perkins *et al*. 2008b – Appendix 5).

Surveys took place between May and August, on mornings with no or light rain and calm or light winds, and usually involved three visits to each farm. However, in three years (2004, 2008 and 2009), some farms (47%, 23% and 59%, respectively) were visited only twice. Each survey route was selected to pass within 250 m of all points on the farm, and mainly followed field boundaries. During each visit, locations and activities of all Corn Buntings were recorded on a 1:10 000 map. On RSS and control farms in Aberdeenshire and Moray, we also recorded all Yellowhammers and Reed Buntings. For all three species, locations of birds on land surrounding the farm, but within 250 m of the farm boundary, were also recorded. The number of territorial males was counted from clusters of map records (Marchant *et al*. 1990). Corn Buntings sing frequently and conspicuously during all daylight hours, up to a rate of five to seven songs per male per minute in early morning (Møller 1983, Olinkiewicz & Osiejuk 2003), and during calm conditions, observers can hear songs up to 500 m away. They sing mostly from prominent song-posts such as overhead wires, fences, tops of trees or bushes, or tall plants within crops, so detection rates are high. In our study, data from those farms on which more intensive Corn Bunting nest monitoring fieldwork was carried out (so that territory count is known with certainty) show that across 98 farm-years and 300 recorded territories, 94% of territorial males were detected by the survey method, so it was not necessary to account for effects of imperfect detection (Gonzalo-Turpin *et al*. 2008) in population trend modelling. This was also the case for Yellowhammer and Reed Bunting, as studies of the closely related Cirl Bunting have shown that two visits are likely to record 84% of territories (Peach *et al*. 2001).

7.2.4. Data analysis

7.2.4.1. Corn Buntings

We excluded eight farms that held no territorial Corn Buntings in any year (Table 7.2). Data analysis was therefore based on 63 farms with at least one territorial male Corn Bunting in one of the years 2003–2009. However, because 30 farms changed treatment group (in one case, twice) during the study due to entering or leaving AES agreements, we considered data after the treatment change as being a new time series of data from a ‘new’ farm.

To assess population change in response to AES, we modelled the density of territorial males on each farm (response variable = *territory count*; offset = *farm area*) as a function of three fixed effects:

(i) *farm type* (1 = RSS; 2 = FBL; 3 = Control),
(ii) years since a farm joined its 'farm type' group (covariate *duration*),
(iii) number of survey visits (*visit*, two or three),

plus the *farm type|duration* interaction term, in a generalized linear mixed model (GLMM) framework. This was specified by the SAS 9.1 GLIMMIX procedure with a log-link function, Poisson error distribution, and standard errors adjusted for over-dispersion, fitting *farm identity* (n = 94 'farms' after including treatment changes) as a random effect. Denominator degrees of freedom for tests of fixed effects (see Chapter 2) were calculated using the Kenward-Roger method (Littell *et al*. 1996). First we fitted *visit*, *farm type* and the *duration|farm type* interaction term, from which back-transformation of regression coefficients to the scale of the response variable gave an estimate of the annual percentage rate of change in the density of territorial males for farms in each treatment group. We then added *duration* to the model to assess differences in trend between each farm type. Finally, to check for confounding of treatment effects with calendar year, we repeated the analysis, replacing *duration* with the covariate *calendar year* (2003–2009). Differences in model output between the two approaches proved negligible.

Secondly, we repeated the analyses, but replaced *farm type* as a descriptor of scheme identity with an alternative three-level fixed effect (*option type*) that described the resources offered by these options. The three levels were food (standard RSS or FBL options), food plus safe nesting habitat (as above but including FBL "late-cut grass"), and no options (control farms). Because safe nesting habitat was implemented mainly in Aberdeenshire and Inverness-shire due to few fields of grass silage occurring in Fife and Angus (Table 7.3), this analysis was done separately on the two areas to investigate regional differences in the influence of the management options.

7.2.4.2. Yellowhammers and Reed Buntings

To assess population change in response to RSS management, we used the same modelling procedure as for Corn Buntings, with the exception that *farm type* was a two-level effect (1 = RSS, 2 = Control). Data analysis was based on 38 and 36 farms with at least one territorial male Yellowhammer or Reed Bunting, respectively, in one of the years 2003–2009 (n = 51 and 48 'farms', respectively, after including treatment changes).

7.2.4.3. Differences in cropping between treatment groups and years

Finally, because differences between the three treatment groups other than AES management may have influenced population trends, we tested (Kruskal-Wallis test) whether agricultural land use (proportions of each major crop) differed between the three groups in 2003 and 2008/09, and whether there was significant change in each farm group between these years (Table 7.3). Further, for 62 farms where cropping was recorded in both 2003 and 2008/09, we tested for significant change in these proportions using a Wilcoxon signed rank test.

7.3. Results

7.3.1. Corn Bunting population changes

Corn Buntings were recorded on 63 of the 71 farms surveyed. The maximum number of males on an occupied farm varied from 1–46 and their density from 0.21–14.71 km^{-2}. Modelled population trends differed between FBL and control farms (t_{317} = 4.53, P <

0.0001), and between RSS and controls (t_{317} = 2.85, P = 0.0047). The density of territorial males on control farms declined at 14.5% per annum (t_{317} = -3.66, P = 0.0003), and on FBL farms increased at 5.6% per annum (t_{317} = 2.73, P = 0.0066). On RSS farms, the rate of decline of 2.0% per annum did not differ significantly from zero (t_{317} = -0.89, P = 0.377), but the difference between RSS and FBL trends was significant (t_{317} = 2.52, P = 0.0122; Fig. 7.2a).

On farms without delayed grass mowing in Aberdeenshire, Moray and Inverness-shire ('food options only' in Fig. 7.2b), populations showed no significant trend (2.3% pa decline: t_{255} = -0.89, P = 0.376), but revealed weak evidence of increase (6.3% pa) on farms with late-cut grass (t_{255} = 1.73, P = 0.0840; 'late-cut grass plus food options' in Fig. 7.2b). This difference was significant (t_{255} = 1.98, P = 0.0491). In Fife and Angus, populations increased (17.8% pa) on FBL farms with food options only (t_{55} = 4.22, P < 0.0001), but declined (33.6% pa) on control farms (t_{55} = -2.38, P = 0.0209; Fig. 7.2c), and the difference between these two trends was significant (t_{55} = 3.25, P = 0.0020).

7.3.2. Yellowhammer and Reed Bunting population changes

Of the 38 farms surveyed for these two species, Yellowhammers were recorded on all 38 and Reed Buntings on 36. The maximum number of males on an occupied farm varied from 2–63 Yellowhammers (density = 1.44–13.33 km^{-2}) and 1–39 Reed Buntings (density = 0.41–9.92 km^{-2}). The density of territorial male Yellowhammers on RSS farms increased at 4.5% per annum (t_{161} = 3.66, P = 0.0003) and on controls increased at 6.2% per annum (t_{161} = 2.76, P = 0.0064; Fig. 7.3a). For Reed Bunting, the density of territorial males increased at 9.4% per annum on RSS farms (t_{150} = 4.37, P < 0.0001), whilst on control farms, the rate of increase of 2.0% per annum did not differ significantly from zero (t_{150} = 0.56, P = 0.5792; Fig. 7.3b). For both species, modelled population trends did not differ significantly between RSS and control farms (Yellowhammer: t_{161} = 0.67, P = 0.5031; Reed Bunting: t_{150} = 1.69, P = 0.0922).

7.3.3. Land use and farming systems

In both 2003 and 2009, farms in Fife and Angus had less grass (χ^2_1 = 7.14, P = 0.0075; χ^2_1 = 10.29, P = 0.0013) and more vegetables (χ^2_1 = 19.12, P < 0.0001; χ^2_1 = 24.46, P < 0.0001), and in 2009 more autumn-sown cereals (χ^2_1 = 5.53, P = 0.0187) than those in Aberdeenshire, Moray and Inverness-shire. Within each region, land use was similar across the three treatment groups, although in Aberdeenshire, Moray and Inverness-shire in 2003, the proportion of autumn-sown cereals was greater on RSS than control farms ($\chi^2_1 = 8.02$, P = 0.0046), and of vegetables greater on FBL than on RSS farms ($\chi^2_1 = 7.06$, P = 0.0079). In 2008/09, FBL farms in this region had a larger proportion of grass than control farms (χ^2_1 = 4.01, P = 0.0453). In Fife and Angus, the only significant difference between treatment groups was in 2003 when FBL farms had a greater proportion of rough grass and set-aside than controls ($\chi^2_1 = 3.85$, P = 0.0499). Across 62 farms surveyed in 2003 and 2008 or 2009, the only crop type whose proportion changed was set-aside and rough grass (Wilcoxon signed rank test $P < 0.0001$), which declined in all treatment groups in both regions due to withdrawal of compulsory set-aside in late 2007 (Table 7.3). Although data were not analysed, other land use factors such as availability of song-posts, non-farmed habitats, field size, and the nature of field boundary features (other than AES measures listed in Table 7.1) are unlikely to have differed between the three treatment groups.

7.3.4. Identifying a population target for AES management for Corn Buntings

If a closed Corn Bunting population of size N is divided into a proportion, p, which benefits from AES and a proportion, 1-p, which does not, and the annual population growth rates of these two proportions are a and b, respectively, then for inter-annual population change t to t+1:

$$N_{t+1} = apN_t + b(1-p)N_t$$

which by re-arrangement for the case where $N_{t+1} = N_t$ (i.e. a stable population) gives:

$$p = (1-b) / (a-b)$$

Substituting in the best estimate values of a = 1.056 (from FBL farms) and b = 0.855 (from control farms), gives p = 0.72. In other words, given a current rate of decline of 14.5% per annum in the wider countryside outside AES, then at least 72% of the Corn Bunting population would need to benefit from agri-environment management to halt the overall decline, assuming that all of this was at FBL standard. Given that a = 1 (or at least does not differ significantly from 1) for RSS farms, then the entire Corn Bunting population would need to benefit from Corn Bunting-relevant RSS management to halt the overall decline. These estimates assume that the annual rate of population change observed on control farms is representative of the population as a whole. Given that our study sites were spread throughout the remaining breeding range of the species in mainland Scotland, this seems a reasonable assumption. By 2009, farms under FBL-type management supported 167 Corn Bunting territories, 24% of the remaining Scottish mainland population of *c*.700 territories. The mean annual cost of AES options on these 16 farms over four years, covering 186 ha, was approximately £40 000 (see Table 7.1 for payment rates). Data were unavailable to allow us to assess the total extent of population coverage by RSS agreements.

7.4. Discussion

We monitored the population response of breeding Corn Buntings to AES management implemented through national (RSS) and locally targeted (FBL) schemes in eastern Scotland over a seven year period. Both schemes included management options known to provide food for Corn Buntings and other farmland passerines. Unharvested crops and weedy over-winter stubbles provide cereal grain and weed seed as food during winter, and Corn Buntings are known to exploit these habitats (Donald & Evans 1994, Brickle & Harper 2000, Perkins *et al*. 2008a, and see Chapter 6). The schemes also provided insect-rich habitats beneficial to Corn Buntings during summer, including grass margins around arable fields, cereal crops with no herbicide applications, species-rich grassland, and set-aside left unsprayed and uncut throughout the summer (Brickle & Harper 1999, Brickle *et al*. 2000, and see Chapter**s** 3 and 4). Management in FBL was adapted during the study by including delayed mowing of forage grasses to protect Corn Bunting nests in meadows, and by ensuring that unharvested crop patches were re-sown annually to provide cereal grain food throughout each winter.

7.4.1. Population response to agri-environment schemes

The number of territorial male Corn Buntings remained stable on farms where AES management with options broadly designed to benefit farmland birds (RSS) were implemented, but increased (5.6% pa) where AES management was specifically targeted at Corn Buntings (FBL). Adaptive improvement of FBL by adding delayed mowing of grass grown for silage in fields where Corn Buntings were nesting may have been a critical addition to the scheme. Before this was introduced, preliminary monitoring of FBL farms revealed that populations were maintained, but did not increase (Perkins *et al*. 2008b, and see Appendix 5), and late cutting is known to increase nest success rates (see Chapter 5). Outside AES management, Corn Bunting populations continued to decline at a rate (14.5% pa) now greater than that (10.3% pa) observed on a partially independent sample of study areas over a longer period (1989–2007) by Watson *et al*. (2009) (see Chapter 2 and Appendix 1). Of course, within these overall trends, there was variation between individual farms. For example, both the largest increase (from five to 15 territorial males over four years) and the biggest decline (from ten to zero territorial males over seven years) were on FBL farms, and this may reflect variation in the degree to which management options were implemented successfully at individual sites. In addition, although non-AES land use did not vary greatly between farm types and years, some differences could have influenced Corn Bunting populations. Notably, the reduction in set-aside and rough grass over the years of study may have contributed to declines on control farms that did not benefit from the ameliorating effect of AES measures.

7.4.2. Effective targeting of management

Overall, these results suggest that the AES available during the study were capable of reversing Corn Bunting declines in eastern Scotland. However, success requires AES management with the biological and spatial targeting found in FBL to be made available to approximately three-quarters of the current population, a large increase on the current level of availability that targets only 24%. Adaptive improvement to management options has been possible in FBL because agreements are flexible and renewed annually. Participants also received regular advice, which may be critical for options that require frequent interventions to recreate or maintain a habitat in good condition, as some farmers may not fully understand the aims of the management or how to maximise the effectiveness of their AES (Morris 2004). In the RSS, however, agreements were fixed for five years, precluding annual adjustments to improve the effectiveness of options, and expert advice was usually lacking. These were key differences between the two schemes that may have contributed to larger responses on FBL than on RSS farms.

Some FBL options, notably late-cut grassland, necessitate substantial payments for profit foregone (Table 7.1). The finer-scale breakdown of the trend analyses suggests that it may be possible to achieve successful outcomes more cost-effectively by targeting different combinations of management options in different areas. Thus, in arable-dominated Fife and Angus where few Corn Buntings nest in grassland (see Chapter 4), management to provide food (e.g. one-year unharvested crops) may be sufficient, but in the mixed farming systems typical of Aberdeenshire, Inverness-shire and Moray, provision of safer nesting habitat via late-cut grass silage fields is likely to be a crucial additional measure. More generally, causes of population declines may vary regionally and between farming systems, according to which resources (e.g. safe nesting habitat, invertebrate food, seed food) are constrained by modern farming, and AES will be most effective if tailored to fill these gaps. For example, schemes targeting Corn Buntings in other farming systems elsewhere in the UK may need to address a lack of late-season nesting habitat, or look for alternative ways of providing winter seed food (Wilson *et al*. 2007a). However, studies which demonstrate this need for geographically targeted variation in agri-environment management are rare, although Batáry *et al*. (2010a) show that effectiveness of grassland extensification schemes for bees is high only in countries (e.g. Switzerland) with intermediate farming intensity. Tryjanowski *et al*. (2011) also use Grey Partridge and Red-backed Shrike as examples of species whose population trends are driven by different factors in the extensive farmlands of central-eastern Europe than in the intensive farming systems of western Europe.

The findings of our study also reaffirmed the view that, for Corn Bunting at least, management agreements must include within-field options, such as late-cut grass and unharvested crops, for schemes to be effective in halting and reversing population declines (Butler *et al*. 2007). Most farmers prefer field boundary options or to use land that is not agronomical than those requiring changes to their management of crops or livestock (Butler *et al*. 2007). Although many of our RSS farmers had opted for unharvested crops, patches were small and often on land that was less productive for growing conventional crops. Few undertook options involving changes to grassland grazing or mowing regimes, although in the latter case this was fortuitous, as the permissible mowing date of 1 July in the RSS option was too early to allow most Corn Bunting nests to fledge (see Chapter 5). None chose the "extensive cropping" option, potentially one of the most beneficial for Corn Buntings, and had more farmers adopted this option, as many did in FBL, populations may have responded more positively to RSS management.

7.4.3. Benefits of agri-environment schemes to other species

Whilst this study focused on Corn Buntings, we also monitored breeding populations of two other bunting species on our RSS and control farms across Aberdeenshire and Moray. Reed Bunting populations increased (9.4% pa) on RSS farms, but showed no significant trend on controls (2.0% pa increase), whilst Yellowhammers increased on RSS farms (4.5% pa) and controls (6.2% pa). Both species made great use of unharvested crops and stubbles on our study farms during winter (Perkins *et al*. 2008a, and see Chapter 6), and would also benefit from RSS field margin options that provide insect-rich foraging habitats (Bradbury *et al*. 2000, Brickle & Peach, 2004). Unlike Corn Buntings, both species also nest in tall herbaceous vegetation along field boundaries, and are therefore not at risk from within-field crop harvesting operations.

The fact that both Yellowhammer and Reed Bunting populations increased or remained stable on controls perhaps reflects the favourable nature of the mixed farming mosaic of spring- and autumn-sown crops and grass fields that is typical of northeast Scotland. Farming systems here may have a greater capacity to maintain populations of some farmland bird species without the intervention of agri-environment schemes than in more intensively farmed arable or pastoral dominated landscapes. Perhaps supporting this conclusion, national population trends for Yellowhammer, Reed Bunting, Skylark and Linnet are all more positive in Scotland than in England (Risley *et al*. 2011 and see Chapter 1). In eastern Scotland, AES resources should therefore target those species most in need of conservation interventions, such as the Corn Bunting.

Other species of high conservation concern likely to have benefited from the AES management implemented include Tree Sparrow, Linnet and Grey Partridge (see Chapter 6), whilst delayed mowing of grass can provide safe nesting habitat for Skylarks, and refuges for a wide variety of invertebrates after conventional fields are mown (Wilson *et al*. 1997, Woodcock *et al*. 2009). Field margin management similar to that in our study can benefit arable flora, butterflies and small mammals (Askew *et al*. 2007, Aviron *et al*. 2007, Walker *et al*. 2007), as well as arthropods and soil macrofauna, which may themselves enhance pollination, pest control and improved soil structure (Smith *et al*. 2008, Albrecht *et al*. 2010).

7.4.4. Scheme uptake and scale of deployment required to halt declines

We have shown that current AES options have the potential to reverse losses of one of Scotland's most rapidly declining farmland birds. However, geographical targeting and flexible, adaptive improvement of measures, backed by advice from experts with sound knowledge of the species are likely to be crucial. Fulfilling this potential depends upon increasing the proportion of the current population targeted by these measures, from approximately a quarter (in 2009) to around three-quarters. Based upon the scale of land management and financial cost of measures provided through FBL (but excluding costs associated with expert advisory input), meeting this target would require 500 – 600 ha of land managed appropriately, at a total cost of around £120 000 per year. This amounts to 0.02% of agricultural and agri-environment subsidies currently paid to Scottish farmers annually (Scottish Government 2009), and 0.5% of land in the current mainland range of

Corn Buntings in Scotland. The potential for success certainly exists. In 2008, the RSS was replaced by Rural Development Contracts (RDCs), which fund relatively simple "Land Managers Options" available to all farmers, and more demanding and expensive Rural Priorities (RPs), which like RSS, operate on a competitive basis (Scottish Government 2010a). However, unlike RSS, the RP scheme is structured to deliver national and regional agri-environment priorities, including biodiversity, through ‘packages’ of options tailored to achieve specific outcomes. One such ‘package’ targets Corn Buntings, offering those management interventions that we have implemented and tested in FBL (Table 7.1), and several of our FBL farms have now transferred into the RP scheme. Contrary to conclusions of a recent report on the future of agriculture funding in Scotland (Pack 2010), our results show such targeting of AES is essential to reverse species declines.

7.4.5. Conclusions

Monitoring in future years will be critical to test whether this adoption of FBL measures by the national AES does deliver targeted management for Corn Buntings on a sufficient scale to reverse the national population decline. Our study illustrates the value of AES monitoring, not only to test scheme effectiveness, but also to allow adaptive improvement of implementation, and to estimate the scale of provision needed. Since the review by Kleijn & Sutherland (2003), studies recommending improvements to AES management are more common (e.g. Douglas *et al*. 2009, Smith *et al*. 2009), but those estimating the scale of intervention needed to reverse large-scale population decline remain rare (Vickery *et al*. 2004). Perhaps the best example is Aebischer & Ewald's (2004) estimate that recovery of British Grey Partridge populations to 1990 levels would require management of 5% of arable land as insect-rich brood-rearing habitat through reduced use of agrochemicals, and 6.9 km km^{-2} of field boundary nesting habitat. Many agri-environment monitoring studies have compared biodiversity trends on AES and control farms over several years (e.g. Kleijn & van Zuijlen 2004, Swetnam *et al*. 2004, Stevens & Bradbury 2006, Birrer *et al*. 2007, Wilson A. *et al*. 2007, Roth *et al*. 2008, Davey *et al*. 2010a). However, we are not aware of any that estimated the proportion of the population that must be targeted to halt population decline at the national scale. Such estimates have practical value because, accompanied by data on distribution and abundance of target species, they help AES administrators to assess the extent, cost and spatial targeting of AES implementation necessary to meet conservation targets for species of high conservation concern. This is particularly important in current times of financial pressure, when it is essential to ensure that public funds are used to best effect.

Table 7.1. *Management implemented on 30 RSS and 35 FBL study farms. Those most likely to provide food (winter seed or summer insects) or safe nesting habitats for Corn Buntings are in italics. For each option, payment rate, frequency of uptake and area managed per farm (mean ± 1 sd) is shown.*

Management option	Main Resource	Payment (£ ha^{-1})		No. of farms		Area (ha)	
		RSS	FBL	RSS	FBL	RSS	FBL
Unharvested crops [L R]	Winter seed	600	[e] 160 / 450	24	30	1.8 ± 0.9	2.9 ± 1.8
Introduction or retention of extensive cropping followed by over-winter stubble [a L R]	Summer insects / winter seed	[f] 120 / 140	[g] 120 / 150	0	17	0	5.1 ± 3.6
Management of conservation headlands [L]	Summer insects	[h] 70 / 150	70	14	11	0.8 ± 0.3	1.1 ± 0.6
Management of grass margin or beetle-bank in arable fields [L R]	Summer insects	736	533	28	12	1.9 ± 1.5	1.3 ± 1.4
Provision of supplementary food in winter	Winter seed		50		13		<1
Late-cut grass for Corn Buntings [b R]	Safe nesting		260		14		7.6 ± 4.8
Delayed spraying and / or cutting of set-aside [c]	Safe nesting		0		14		6.6 ± 5.5
Extensive management of mown grassland for birds	(Safe nesting) [d]	150		6		16.4 ± 17.1	
Creation and / or management of species-rich grassland [R]	Summer insects	250 / 100		13		1.6 ± 1.7	
Management of water margin [R]		400		24		0.9 ± 0.6	
Management of open grazed grassland for birds		100		8		5.0 ± 4.4	
Creation and / or management of wetland		250 / 100		15		3.0 ± 3.7	
Management of wet grassland for waders		100		2		3.5 ± 2.2	
Management of flood plain		25		2		16.5 ± 0.7	
Management of scrub (including tall herb communities)		55		2		1.7 ± 1.5	
Creation and / or management of hedgerows [L]		5000		16		0.2 ± 0.2	
Management of extended hedges		500		10		0.5 ± 0.5	
Management of native or semi-natural woodland [L]		100		1		5.1 ± 0.0	
Management of a site of archaeological or historic interest [L]		80		4		9.1 ± 16.9	
Pond creation		18000		4		0.3 ± 0.3	

[a] extensively managed spring cereal or rape followed by over-winter stubble that could not be sprayed or ploughed before 28 February (RSS) or 31 March (FBL), or extensively managed turnips; [b] available in FBL from 2005; [c] until 15 September; [d] dependent on mowing date (permissible date of 1 July was too early to allow most Corn Bunting nests to fledge before mowing); [e] lower payment rate for crops grown on set-aside; [f] higher payment rate for management applied to the same field for three years or more; [g] lower payment rate for extensively managed turnips; [h] higher payment rate if nitrogenous fertiliser not applied; [R] since 2008, available within the Corn Buntings package of the Rural Priorities scheme; [L] since 2007, available as a Land Managers Option; grey shading = not available in this scheme.

Table 7.2. *Farms monitored in each treatment group (RSS, FBL and Control).*

	Aberdeenshire & Inverness-shire			Fife & Angus		
Year	RSS	FBL	Control	RSS	FBL	Control
2003	23 (4)	15 (2)	17	0	7 (2)	4
2004	25 (4)	15 (2)	16	0	7 (2)	4
2005	4	7	5	0	7 (2)	3
2006	30 (4)	12	9	0	10 (2)	1
2007	4	13	1	0	11 (2)	0
2008	23 (2)	12	16 (2)	0	10 (2)	1
2009	14 (2)	11	12	0	10 (2)	1

( ) = farms with no Corn Buntings in any year during 2003–2009 and excluded from analyses.

Table 7.3. *Area (ha) surveyed and land use composition (%) per farm in each treatment group and region (mean ± 1 sd), 2003 and 2008/09.*

Region	Farm type	Year	No. of farms [a]	Total area (ha) [b]	SC	AC	VEG	OSR	GRA	ROU
Aberdeenshire,	RSS	2003	23	2641	30 ± 17	20 ± 16	4 ± 10	6 ± 13	24 ± 18	8 ± 8
Moray &	RSS	2008–09	29	3455	37 ± 23	17 ± 20	2 ± 5	5 ± 10	29 ± 22	3 ± 4
Inverness-shire		*P*			0.28	0.25	0.98	0.84	0.38	0.06
	FBL	2003	15	1982	28 ± 17	13 ± 15	6 ± 5	7 ± 11	33 ± 20	6 ± 7
	FBL	2009	11	1399	29 ± 18	13 ± 18	6 ± 6	5 ± 10	41 ± 24	3 ± 3
		P			0.81	0.82	0.83	0.55	0.44	1.00
	Control	2003	17	1611	44 ± 25	9 ±19	3 ± 4	1 ± 4	32 ± 25	9 ± 11
	Control	2008–09	19	1667	37 ± 23	18 ± 22	4 ± 9	6 ± 20	24 ± 20	4 ± 11
		P			0.41	0.14	0.91	0.66	0.35	0.02
Fife & Angus	FBL	2003	7	1243	42 ± 19	17 ± 23	19 ± 10	1 ± 3	11 ± 12	4 ± 4
	FBL	2009	10	1993	28 ± 23	32 ± 20	21 ± 8	2 ± 5	9 ± 12	1 ± 2
		P			0.17	0.20	0.70	0.66	0.80	0.15
	Control	2003	4	781	26 ± 30	22 ± 31	28 ± 6	7 ± 7	8 ± 9	0 ± 0
	Control	2009	1	72	67 ± 0	18 ± 0	15 ± 0	0	0	0
		P			0.18	0.66	0.18	0.35	0.35	0.56

P-values are for Kruskal-Wallis tests for differences in proportions of each crop type within each treatment group between 2003 and 2008–09.
SC = spring cereals, including barley/legume mixture mown for arable silage; AC = autumn cereals; VEG = root vegetables or legumes (peas, beans); OSR = oilseed rape; GRA = grazed pasture, grass mown for silage or hay, or newly sown; ROU = rough grass or set-aside (rotational and non-rotational).
[a] *some farms switched treatment groups between years.*
[b] *total area includes non-cropped land and minor habitat categories not presented in table.*

Figure 7.1. *Distribution of study sites (black dots) and Corn Buntings (10-km squares occupied during 2002–09) in mainland Scotland.*

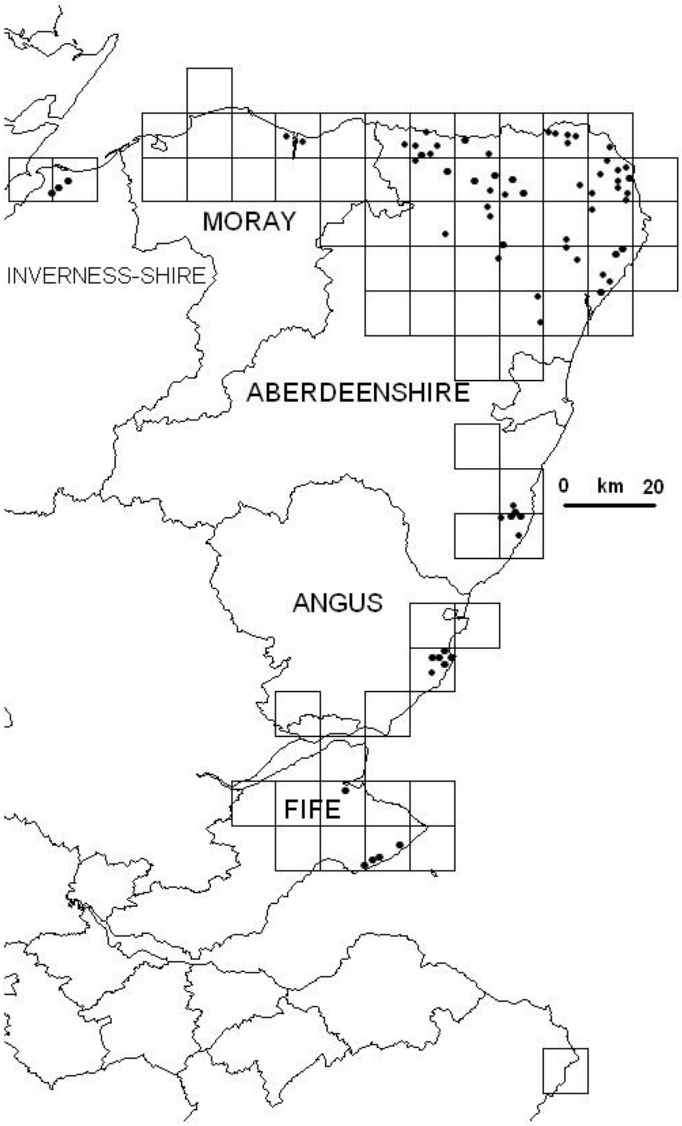
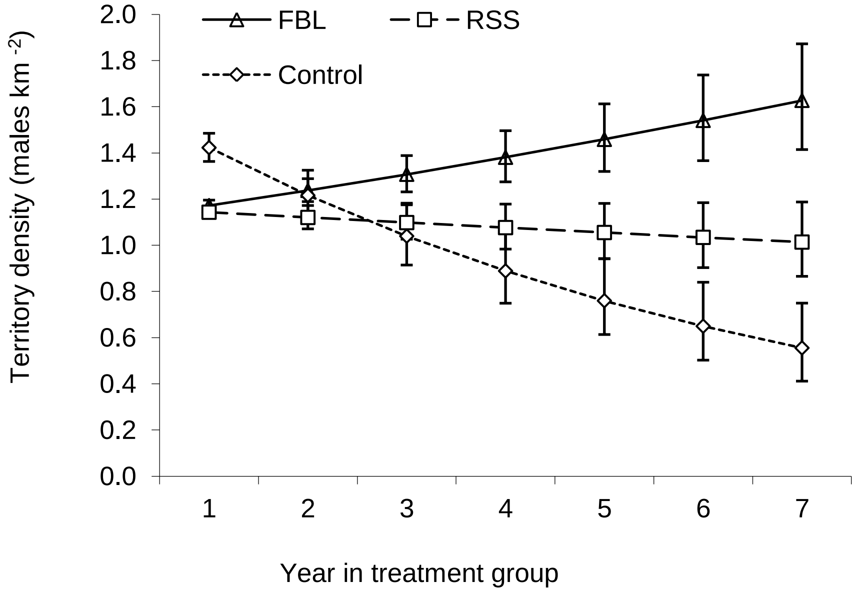


Figure 7.2. *Corn Bunting population trends on agri-environment scheme and control farms, plotting model estimates for mean density of territorial males per farm type (± 1 se) in each treatment year (defined as year of management for AES farms, and year of control within 2003–09 for control farms).*

a) *All farms (n = 63 farms).*



b) *Aberdeenshire, Moray & Inverness-shire (n = 52 farms).*

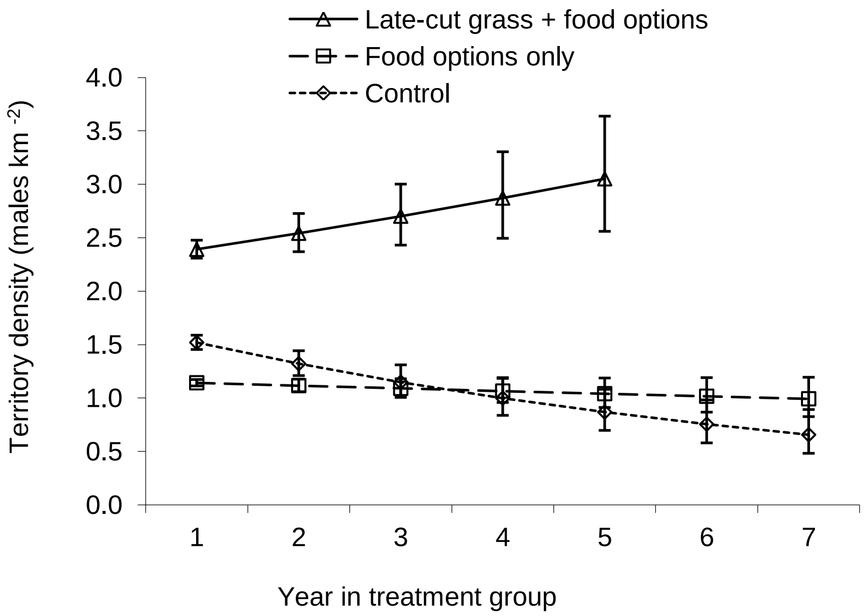


Figure 7.2 cont.

c) *Fife & Angus (n = 11 farms) Note that for clarity, standard errors not plotted for late-cut grass + food options (year 1: lower se = 0.42, upper se = 0.66; year 2: lower se = 0.89, upper se = 2.17).*

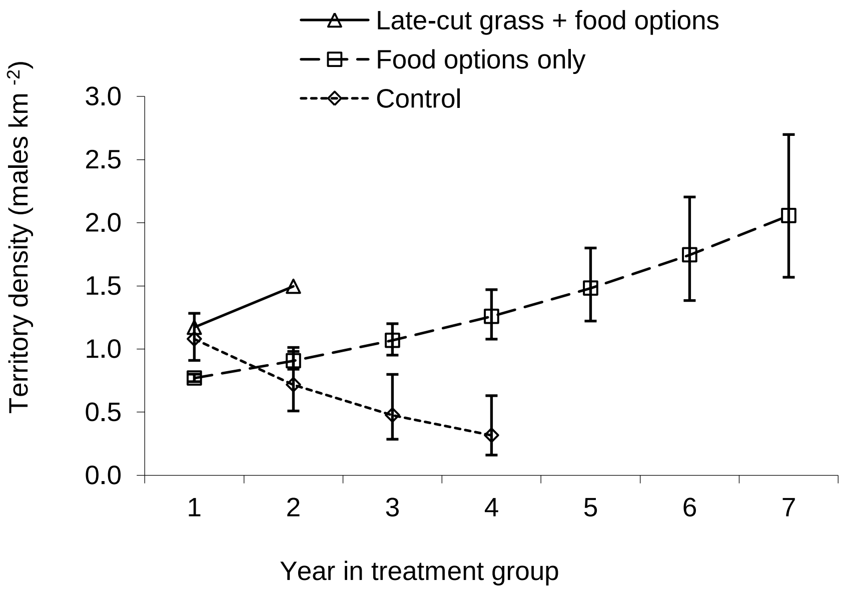
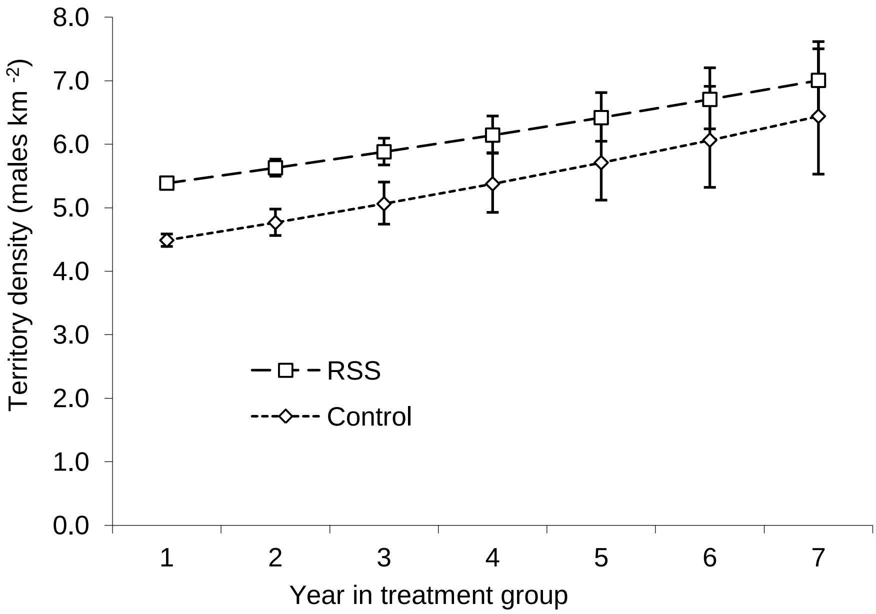
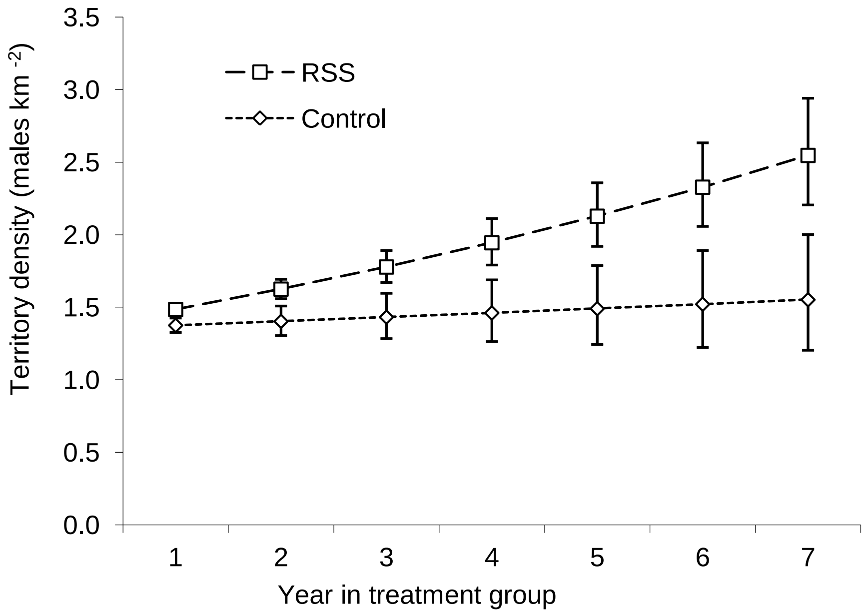


Figure 7.3. *Yellowhammer and Reed Bunting population trends on RSS and control farms in Aberdeenshire and Moray, plotting model estimates for mean density of territorial males per farm type (± 1 se) in each treatment year (defined as year of management for RSS farms, and year of control within 2003–09 for control farms).*

a) *Yellowhammer (n = 38 farms).*



b) *Reed Bunting (n = 36 farms).*



CHAPTER 8. GENERAL DISCUSSION

In this thesis, I aimed to determine the causes of decline and test conservation solutions for one of Europe's most severely declining farmland birds, the Corn Bunting, in the core of its Scottish range. In Chapter 1, I reviewed changes that have taken place in agriculture over the past few decades, and their effects on farmland bird populations throughout Europe. For Corn Bunting, the findings of several detailed studies from across Europe include general negative associations with intensification of arable management, positive associations with crop diversity and the area of semi-natural habitats, and mixed associations with grassland management intensity (see Table 1.5). In the following sections, I synthesise the findings presented in Chapters 2–7, with reference to the information presented in Chapter 1 to place our findings into the wider context of farmland bird research across Europe.

8.1. Corn Bunting population trends in Scotland

In the 1900s, the Corn Bunting, also known then as the Common Bunting, occurred throughout the British Isles wherever farmers grew cereals (Donald *et al*. 1994, Holloway 1996). In Scotland, the species' distributional range included northern and western areas and most of the human-inhabited islands, but there were large declines throughout the twentieth century (Forrester *et al*. 2007). Between the two British breeding bird atlases of 1968–72 and 1988–91, the number of occupied 10-km squares declined by 32%, the largest range contraction of any UK FBI species (Sharrock 1976, Gibbons *et al*. 1993, and see Chapter 1, Table 1.1.). Further local extinctions have since occurred, such that in Scotland the species now occupies just two areas, the east coast lowlands and the Western Isles (Forrester *et al*. 2007, and see Chapter 2, Fig. 2.1.).

Consequently, the Corn Bunting is too rare and localised for the national monitoring scheme (Breeding Bird Survey) to derive a Scottish population trend. Therefore, the study presented in Chapter 2 is the only measure of recent population trend we have for this species in the core of its mainland Scotland range. Across 30 study areas in eastern Scotland between 1989 and 2007, the number of territorial male Corn Buntings declined in all but one (to extinction in all but four), and by 83% overall. Since then, continued declines have reduced these populations even further. In one area (group 5 – see Chapters 2 and 3), just one territorial male remained in 2011, compared with 12 in 2007, and in another area (group 13), just two remain, down from four in 2007. Further south, annual monitoring of the Corn Bunting population in Fife showed a fall from 120 territories in 1995 when surveys began, to 75 in

2001, but following the introduction of targeted conservation measures, the population has recovered in recent years to 101 territories in 2011 (T.C. Smout unpubl. data). The current Scottish population of territorial males is considered to be 800–900, of which 85% occur in eastern Scotland. Of the 19 species whose trends make up the UK Farmland Bird Index (FBI), Corn Bunting is one of the two (the other being Yellow Wagtail, with a breeding population of just 25–35 pairs) most vulnerable to extinction in Scotland (Forrester *et al*. 2007). The country has already lost one FBI species, the Turtle Dove, which formerly occurred in small numbers in southern Scotland (10–20 pairs in the 1950s and 1960s), but following declines throughout its UK range, no longer breeds in Scotland (Forrester *et al*. 2007).

8.2. Corn Bunting habitat associations in eastern Scotland

Despite recent and ongoing population declines, the fact that Corn Buntings still occur in the lowlands of eastern Scotland is in itself of great interest. This is because across most of the rest of Scotland, Wales, Ireland and large parts of England such as the southwest, Corn Buntings are now extinct (or almost so) where once they were common (Donald *et al*. 1994 and see Chapter 2). Lowland eastern Scotland is characterised by arable and mixed farming systems with a predominance of spring-sown cereals, mainly barley (Table 8.1). Moving south from Aberdeenshire to Fife, the farming landscape becomes more arable-dominated, including greater production of vegetables (mostly for human consumption), and a larger proportion of cereals are autumn-sown. However, even here, 45% of cereals across the study sites (5 farms and 7 years) were spring-sown, compared with only 10–20% of cereals in southern and eastern England (Table 8.1). Thus, the studies presented in this thesis are from an area where some aspects of agricultural intensification (farm specialisation, and a switch from spring to autumn sowing of cereals) have been less apparent than in other parts of the UK and Europe. They therefore provide a valuable insight into Corn Bunting habitat associations from an agricultural landscape where mixed farming systems create a mosaic of arable and grassland, and in which most cereals are spring-sown.

8.2.1. Summer

Habitat associations of Corn Buntings during the breeding season were determined in Chapter 3 using a pre-existing 20-year dataset of annual distribution and mating status of territorial males and land use across 36 km^2 of farmland, and in Chapter 4 using new data on nest locations of females in relation to sward structure across 32 study farms. In addition, in

2007 I collected data (55 watches totalling 67.4 hours from 30 nests across seven farms) on foraging locations of adults when provisioning chicks, but do not present these data in a chapter because they have not yet been formally analysed. However, because they add further, supporting information regarding Corn Bunting breeding habitat associations, I refer to some of the preliminary findings here. None of these findings suggest additional habitat associations that are not supported by data in other chapters, and in most cases are consistent with those of published studies on chick-provisioning foraging locations of Corn Buntings and related species.

8.2.1.1. Cereals (autumn- and spring-sown)

Territorial male Corn Buntings were strongly associated with cereals during the breeding season, but specifically with autumn-sown barley in early summer, and with spring-sown cereals throughout the summer (Chapter 3). Mid-summer abandonment by males was more likely in territories with more autumn-sown barley, and less likely in those with more spring-sown cereals. These associations reflected favoured nesting habitats of females, which laid clutches from mid-May to mid-August, but whose crop use varied during the breeding season (Chapter 4). Overall, 61% of nests were in cereals, but female Corn Buntings increasingly used fields of spring-sown cereals for nesting as the season progressed, whereas their use of autumn-sown cereals for nesting declined. This was because spring cereal swards became taller and denser as crops matured, and thus more attractive as nesting habitat later in the season (see Chapter 5, Fig. 5.1). By contrast, the effects of crop plants dying once they had ripened, pre-harvest spraying with herbicides from July onwards, and the onset of harvesting, made autumn-sown cereals increasingly less attractive as nesting or foraging habitats. However, unlike previous UK studies (Crick *et al*. 1994, Brickle & Harper 2002), we recorded very few instances of direct nest failures caused by cereal harvesting operations (Chapter 5).

Relative use of spring- and autumn-sown cereals by parents provisioning chicks followed a similar seasonal pattern. Of 158 foraging trips recorded in June with known destination, 19% were to autumn-sown cereals and 9% were to spring-sown cereals. By August, of 88 foraging trips, none were to autumn-sown cereals despite some crops still standing, whilst 78% were to spring-sown cereals. The increase in use of spring cereals was concurrent with crops maturing and bearing part-ripe cereal grain from mid-July onwards (Chapter 4), which is an important food source for bunting chicks, particularly during cold wet weather when invertebrate activity and availability is low (Watson 1992, Douglas *et al*. in press). On our study sites, autumn barley was the only cereal crop offering part-ripe grains during early summer (from mid-June), supplemented by autumn wheat from late June and by spring barley from mid-July (Chapter 4). Douglas *et al*. (2010), whose study sites overlapped with mine, also showed that autumn-sown barley supported twice the abundance of arthropods in May, and 50% more in June, than spring-sown barley. However, by late summer when the spring-sown crops had matured, and many autumn-sown crops had become fully ripe and begun to die, spring barley held 25% more arthropods than autumn barley (Douglas *et al*. 2010).

Autumn-sown barley and spring-sown cereals therefore provide a good combination of crops for Corn Buntings, offering nesting, grain-rich and potentially insect-rich (dependent on levels of pesticide use) foraging habitat throughout summer, and increasing the possibility for females to rear two broods.

8.2.1.2. Arable weeds

By far the biggest influence on territory location was weed abundance within fields (Chapter 3), suggesting, in accordance with studies in other regions, that Corn Buntings are strongly associated with ‘extensive’ farming regimes in which crops receive few or no pesticide applications. In addition, weed abundance within crops was strongly associated with late-summer territory occupancy by males, and with polygyny (Chapter 3), and in Chapter 4 I showed that females were two to three times more likely to nest in cereal fields with high weed abundance than in those with few or no weeds. Weedy fields are preferred for two main reasons. First, they provide insect-rich foraging habitat, so females nesting within them have less far to travel when provisioning chicks (Brickle *et al*. 2000), and second, dense weed cover can help to conceal nests from predators. A recent English study also showed that female Corn Buntings were more likely to re-nest in cereals with high weed abundance than in crops with few weeds, thus increasing the likelihood of females attempting to rear second broods (Setchfield *et al*. 2012).

8.2.1.3. Forage grasses

Territories were strongly associated with forage grasses in early summer (Chapter 3), and as with cereals, this most likely reflected the preferred nesting habitats of females. Overall, meadows of forage grasses accounted for one-fifth of nests, but half of the nests initiated in May – early June in the mixed-farming regions of Aberdeenshire and Inverness-shire

(Chapter 4). This was because forage grasses offered the densest swards during early summer, especially on farms where cereals were spring-sown. However, rates of nest loss in meadows were high during cutting operations in June and July (Chapter 5).

Intensively managed grasslands that are frequently cut or heavily grazed do not generally support high numbers of the larger invertebrates that form the main prey of buntings, and therefore buntings rarely use these habitats for foraging during summer (Brickle *et al*. 2000, Morris *et al*. 2001). However, meadows that are cut late and only once per year may offer insect-rich habitats (Vickery *et al*. 2001), although the dense swards may reduce accessibility to this food (Atkinson *et al*. 2005). In my study, of 618 foraging trips made by Corn Buntings when provisioning chicks, 13% were to meadows. Use of this habitat peaked in July (19% of visits), when grasses were flowering and densities of chick-food invertebrates within these swards were perhaps at their greatest just prior to mowing. Douglas *et al*. (2009) showed that Yellowhammer chick-food arthropods in uncut grass margins around arable fields steadily increased in abundance throughout the summer.

8.2.1.4. High perches and song-posts

Territories were strongly associated with overhead wires (Chapter 3), an association that grew stronger in later years as the study population declined. Pole-mounted telephone and electricity wires running across fields or along treeless field boundaries offer high perches with an unobstructed view of surrounding areas, and this appears to be important for territorial male Corn Buntings. They sing from these high perches and use them for watchful behaviour, such as observation of neighbouring males and looking out for potential predators, giving alarm calls whenever raptors, crows, mammalian predators and humans approach nests within their territory. During the nest building, egg-laying and incubation periods, male Corn Buntings also follow their female partners whenever they leave the nest. Probably because of this behaviour, nests were relatively close to the male’s main song-post, on average the two being approximately 70 m apart, and less than 100 m apart in 80% of cases.

8.2.1.5. Field-boundary features

There was a weak association between territory locations and field-boundary features. Fences, walls, bushes and small trees provide males with elevated perches and song-posts, whilst overhead wires also frequently followed roads, tracks and other field-boundary features. Although Corn Buntings did not generally nest in field-boundary habitats such as ditches or grass margins (just 1% of nests – see Chapter 4), or close to the field edge (only 17% of nests were less than 10 m from a field boundary), few nests were more than 50 m (14%) from a field boundary. One possible explanation is that females preferred to nest in reasonably close proximity to the male's main song-post, and thus benefit from his vigilance and warning calls of approaching predators.

In accordance with other studies of this (Brickle *et al*. 2000) and related species (e.g. Yellowhammer – Morris *et al*. 2001, Perkins *et al*. 2002, Douglas *et al*. 2009), Corn Buntings made great use of field-boundary features as foraging habitats when provisioning chicks. Overall, 30% of foraging trips were to locations within grass margins, road- and track-verges, or along the edges of fields, despite such habitats forming a small proportion of the total area available. The proportion of foraging trips to field-boundary habitats was highest during June (37%) and July (33%), but declined to just 9% in August. The seasonal decline in use of field-boundary habitats may have been due to vegetation growth making grass swards increasingly dense and inaccessible to foraging birds, as demonstrated by Douglas *et al*. (2009) in studies of Yellowhammer foraging use of experimentally created short, sparse patches within grass margins. However, in both my and Douglas's study, the decline in use of field-boundary habitats during the summer corresponded with an increase in foraging trips to within-field locations in spring cereals, likely reflecting greater availability of food (ripening cereal grains and invertebrates) in these crops as they matured (Douglas *et al*. 2010). In each of the months May – August, Douglas *et al*. (2009) found that Yellowhammer chick-food arthropods were more abundant in grass margins than spring barley, but the magnitude of this difference in abundance was greatest in May (fourfold) and smallest in July (25%).

8.2.1.6. Fallow land

Territories were increasingly associated with fallow land such as set-aside and rough grass (positive in early summer, negative in late summer) in later years. Although female Corn Buntings did nest in these habitats, cereal fields and forage grass meadows were used far more frequently (Chapter 4). Use of set-aside or rough grass by foraging adults provisioning chicks was also relatively low (approximately 10% of all foraging trips), although these habitats were used in greater proportion to their available area. In southern England, almost one fifth of Corn Bunting foraging trips recorded by Brickle *et al*. (2000) were to patches of non-rotational set-aside, and this rough grassland habitat held the third highest abundance of chick-food invertebrates (second only to 'unintensified' chalk grassland and grass margins around arable fields). In studies of foraging Yellowhammers across three English regions, patches of non-rotational set-aside were selected but summer fallows (stubbles) were not, probably because the latter were usually sprayed with herbicides in early-summer (Morris *et al*. 2001).

8.2.1.7. Other crop types

Root vegetables, legumes and oilseed rape had little influence on territory location, although autumn-sown rape was negatively associated with late-summer territory occupancy (Chapter 3). We recorded few nests in any of these crop types, although use of vegetables increased as the breeding season progressed, and as these crop plants grew bigger to provide greater physical cover for nests (Chapter 4). I also recorded few foraging trips to vegetables or rape, although such crops were scarce on these particular study farms. Use of vegetables by foraging Corn Buntings may have been higher on farms where their area was greater, such as in Fife and Angus.

Territory associations with grazed pasture were not formally tested (Chapter 3), but its mean proportional area was significantly greater in territory circles at null sites than at occupied sites, suggesting a general avoidance (or at least lack of selection) by territorial Corn Buntings. However, some females did nest in pastures that had not recently been grazed, particularly in early summer, although their probability of use was much lower than that of forage grasses (Chapter 4). Newly re-seeded grass, with its dense weedy swards, was a scarce but strongly selected field type by females for nesting in mid to late summer, although most nests failed due to cutting to prevent weeds from setting seed (Chapter 5).

8.2.2. Winter

Winter habitat associations of Corn Buntings and nine other seed-eating farmland birds were determined in Chapter 6 using new data collected across 53 study farms over three winters. Below, I summarise the use by Corn Buntings and other species of each of the habitats surveyed.

8.2.2.1. Cereal stubbles

Almost half of the Corn Buntings, and of the 36000 birds of all ten species combined, were in cereal stubbles, consistent with the findings of other studies that have demonstrated the importance of this habitat for overwintering seed-eating farmland birds. These stubbles provide weed seeds and cereal grains, both of which are exploited by buntings, sparrows, larks and some finch species (Wilson *et al*. 1999). Corn Bunting territory associations with spring cereals in early summer (Chapter 3) may have been partly due to birds establishing breeding territories around seed-rich stubbles used during the preceding winter months. Studies of Skylarks and Yellowhammers (which selected cereal stubbles in all three winters in our study) have shown that availability of stubbles can have a positive effect on subsequent breeding densities (Gillings *et al*. 2005, Whittingham *et al*. 2005).

8.2.2.2. Other stubbles

Non-cereal stubbles such as those of oilseed rape and vegetables were used infrequently by Corn Buntings (6% of birds), but selected in at least one winter by four other species (Grey Partridge, Skylark, Chaffinch and Linnet).

8.2.2.3. Unharvested crops

The agri-environment scheme management option “unharvested crops” held 44% of Corn Buntings and 28% of birds across all ten species. Only Skylark, a species that tends to avoid foraging in tall, dense vegetation typical of these crops, did not show significant selection of unharvested crops in at least one winter. Use of unharvested crops was especially high by Reed Bunting and Greenfinch, for which this crop type held more than 60% of birds recorded. Unharvested crop patches in their first winter of establishment held more buntings and sparrows than patches in their second winter. This was because one year old crops held much more cereal grain than two year old crops, whose main seed-food provision was the oily seeds of kale. Species such as Corn Bunting and Yellowhammer feed predominantly on cereal grain, so favour cereal-rich one-year crops, whereas finches readily take oily brassica seeds, enabling them to exploit kale-dominated two-year crops.

8.2.2.4. Non-rotational set-aside

Just 2% of Corn Buntings were recorded in the rough grassland habitat of non-rotational set-aside fields, and this habitat was selected in at least one winter by just two species, Skylark and Linnet.

8.2.2.5. Vegetables and fodder crops

Only four Corn Buntings were recorded using vegetables and fodder crops such as turnips. However, across all ten species, turnips held 9% of birds recorded, despite occupying just 2–3% of the seed-rich habitat surveyed, and were selected by Linnet and Chaffinch in three and two winters, respectively. Turnip crops often receive few herbicide applications, so tend to be weedy and heavily used by birds such as finches that feed predominantly on the seeds of arable weeds (Hancock & Wilson 2003).

8.2.2.6. Other sources of cereal grain

Although they covered less than 2% of the seed-rich habitat area surveyed, farmyards held 5% of birds across all ten species, but this included just four records of Corn Bunting. Tree Sparrows were particularly strongly associated with seed sources in farmyards, with 31% of birds recorded in this habitat and significant selection in all three winters. Some species including Tree Sparrow, Goldfinch, Greenfinch and Chaffinch readily take seed provided specifically for birds in gardens, but others such as Corn Bunting, Skylark and Linnet rarely do so (Chamberlain *et al*. 2005), perhaps because they prefer to feed in open areas well away from human habitation. Corn Buntings do exploit artificial grain sources such as livestock feed troughs (Brickle & Harper 2000), and provision of such grain-filled troughs for Corn Buntings on the Western Isles has met with some success (Wilson *et al*. 2007b). Provision of supplementary seed food through scattering grain or harvest waste (often referred to as ‘tailings’) has also proved effective at attracting Corn Buntings and other seed-eating farmland passerines, with some positive effects on local breeding population trends (Siriwardena *et al*. 2007).

Although not part of formal analyses, on several occasions I recorded Corn Buntings and other grain-eaters such as Yellowhammers and Chaffinches exploiting newly drilled grains in spring-sown cereal fields in March and April. This behaviour by Corn Buntings has been reported previously (Brickle & Harper 2000). As with winter use of stubbles, birds settling next to fields exploited for food during early spring may be one of the reasons for the early summer territory association with spring cereals (Chapter 3).

8.3. Farming changes associated with population declines in eastern Scotland

In Chapter 3, we quantified changes in the habitat attributes and distribution of Corn Bunting breeding territories as the study population declined, from 134 territories in 1990 to just nine in 2008. We also determined that in later years, fewer males held territories into late summer, and more males were either polygynous or unmated. Both of these changes suggested that habitat quality declined and became more variable spatially in later years. Here, I summarise the aspects of agricultural intensification in eastern Scotland likely to have had the greatest influence on local Corn Bunting population trends.

8.3.1. Reduced weed abundance in cereal crops

Weed abundance within fields declined during the 20-year study, with mean scores in spring cereals falling by 50% between the first three and last three years (Chapter 3). This decline was likely due to an increase in herbicide use over the same period, as demonstrated by national statistics on annual rates of herbicide use in cereals, which showed a 79% increase in the active substance treated area across Scotland between 1990 and 2010 (Table 8.1). Effects on Corn Buntings are fourfold. First, weed plants host invertebrate food for adults and chicks, so crops with fewer weeds tend to have fewer invertebrates. A lack of invertebrate food can lead to Corn Bunting chicks being in poorer condition, and ultimately can reduce survival rates of nestlings and fledglings (Brickle *et al*. 2000). Douglas *et al*. (in press) showed that Yellowhammer chicks in eastern Scotland fed mostly on cereal grain were in poorer condition than chicks fed invertebrates. Second, reduced invertebrate abundance can adversely affect adult survival through them having to work harder to maintain breeding success by flying further to find food (Siriwardena *et al*. 2000a). Third, weeds provide physical cover for nests within crops, and a reduction in weed cover means nests are likely to be more vulnerable to predation (Chapter 4). Less weed cover in cereal crops may also encourage Corn Buntings to nest in the dense swards of forage grasses where they suffer high rates of nest loss to cutting (Chapters 3, 4 and 5). Fourth, Corn Buntings and other farmland passerines feed on arable weed seeds during winter (Wilson *et al*. 1999), so stubbles that follow crops with few weeds have less over-winter seed food for granivorous farmland birds (Bradbury *et al*. 2008).

8.3.2. Switch from spring- to autumn-sown cereals

Across the 36 km^2 of the 20-year study (Chapter 3), there was no clear pattern of change between the proportion of cereals sown in spring and autumn, or the types of cereals grown. National statistics (Table 8.1), however, show that across Scotland, wheat increased from 16% to 26% of cereals grown between 1985 and 2010, whilst the share of barley declined from 79% to 68%. The proportion of Scottish cereals spring-sown declined slightly, from 65% in 1985 to 61% in 2010, but because the overall area of cereals declined by one fifth during this period, the area of spring-sown cereals was 25% lower in 2010 than in 1985, whilst the area of autumn-sown cereals was 8% lower. Across northeast Scotland between 2000 and 2010, the overall area of cereals remained constant, whilst changes in the types of cereal grown were similar to, but weaker than, the national trends (Table 8.1).

The timing of sowing of cereals affects Corn Buntings in several ways. First, spring-sown cereals are the most widely used crop for late-summer nesting attempts, giving female Corn Buntings greater opportunities to rear a second brood. In Fife and Angus, where up to 65% of cereals are autumn-sown, the breeding season is around three to four weeks shorter than in Aberdeenshire and Inverness-shire, where only a quarter of cereals are autumn-sown, and fewer Corn Buntings rear two broods (Chapter 4). A low incidence of second broods is thought to have contributed to Corn Bunting population declines in English regions dominated by autumn-sown cereals, where little available nesting habitat remains by mid-August after most cereals have been harvested (Brickle & Harper 2002). Second, in winter, Corn Buntings are strongly associated with cereal stubbles (Chapter 6), and a switch from spring to autumn sowing removes the possibility of retaining over-winter stubbles, thus reducing the availability of winter seed food, and potentially reducing annual survival rates. Even in Aberdeenshire where most cereals are spring-sown, farmers often plough stubble fields several weeks (and sometimes months) before they sow new crops in March and April (Chapter 6). I have no information on whether ploughing has become earlier, but the timing does vary considerably between winters, according to the weather. Farmers cannot easily use heavy machinery in fields when the soil is waterlogged, so in wet autumns and winters (e.g. winters with prolonged periods of snow cover, such as 2009/10 and 2010/11), ploughing tends to be later. Third, Corn Buntings and other grain-eaters exploit newly drilled grains in spring-sown cereal fields in March and April (Brickle & Harper 2000, and see Chapter 6), so switching to autumn-sown cereals removes this seed source at a time of year when other sources may have become depleted and granivorous farmland birds struggle to find food (Siriwardena *et al*. 2008). Finally, sowing some cereals in autumn, particularly barley, is beneficial for Corn Buntings because these crops provide nest sites and a source of cereal grain and insect food during early summer, at a time when spring-sown crops are too short and sparse to conceal nests (Chapter 4), and do not provide grains or support large numbers of invertebrates (Douglas *et al*. 2010).

8.3.3. Earlier mowing of forage grasses

Across the Chapter 3 study site, the total area of grassland remained approximately constant over the 20 years, making up 30–35% of the study area. The proportion of grass fields cut for silage or hay varied from 30% to 50% p.a., with no overall trend across years. In recent decades, throughout the UK there has been a widespread switch in the management of forage grasses from taking a single late cut for hay to earlier and multiple cuts to make silage (Chapter 1). Unfortunately, I do not have data demonstrating this on our study sites, or more generally across Scotland, but in Aberdeenshire, most meadows are now cut twice per year, with the timing of first cuts ranging from mid-May to late July, and second cuts from early July to late August (Chapter 5). Forage grasses currently act as a trap for nesting Corn Buntings in northeast Scotland, because their tall dense swards provide attractive nest sites in early summer, but most first cuts for silage are taken in June or early July, when nests still contain eggs or chicks (Chapter 5). In meadows cut late (on or after 1 August), very few nests are destroyed by mowing, and nest success is five times higher than in meadows cut between mid-May and mid-July.

8.3.4. Loss of arable fodder crops

Although still widely grown in northeast Scotland, arable fodder crops are increasingly being replaced by grass silage. Turnips are one of the main fodder crops, and between 1990 and 2006, the area grown in northeast Scotland declined by two-thirds, from approximately 12000 ha to 4000 ha (Wilkinson *et al*. 2010). Across the whole of Scotland between 1985 and 2010, the area of fodder turnips declined by 87% (Scottish Government 2010b), whilst the area across the Chapter 3 study site declined from 63 ha in 1989–91 to 31 ha in 2006–08, which may be accounted for by a similar magnitude of increase in the area of forage grasses.

Corn Buntings did not show strong associations with turnips, but other crops grown for livestock feed include cereals, and these were strongly selected by the species. In Aberdeenshire, up to one-third of cereals grown supply harvested grain fed to livestock on the same farm (Cook 2008). Switching from spring-sown cereals, which are often weedy when grown for livestock feed, to forage grasses cut for silage in early summer replaces a crop type strongly selected by breeding Corn Buntings throughout the summer with one that attracts nesting females, but acts as a trap because most nests are destroyed when fields are mown (Chapter 5). Replacing cereals with grass also removes an important winter seed-food source for Corn Buntings and other birds, which often exploit grain provided for livestock during winter (Brickle & Harper 2000, Wilson *et al*. 2007b), a foraging behaviour that I have observed on several occasions in Aberdeenshire. Any reduction in the area of spring-sown cereals will also result in the loss of seed-rich over-stubbles and spring-drilled grain, both of which are heavily used by Corn Buntings during winter and early spring (Brickle & Harper 2000, and see Chapter 6).

8.3.5. Loss of set-aside

Our long-term study (Chapter 3) spanned the 'set-aside' period (1992–2007) when throughout the EU it was compulsory for farmers growing cereal and protein crops to remove a proportion of their land from crop production. Consequently, the area of summer fallow (including 'rough' grass) doubled across the study area between 1990 and 1993, and then remained high (200–300 ha) in most years before falling back to pre-1993 levels in 2008 following the end of compulsory set-aside. Similarly, across 62 farms elsewhere in eastern Scotland, there was a significant decline in the area of set-aside and rough grass between 2003 and 2008/9 (Chapter 7), and across northeast Scotland, the area of arable fallow fell by 82% between 2000 and 2010, from 34000 ha to 6000 ha (Table 8.1).

During our long-term study (Chapter 3), territorial males became increasingly associated with set-aside and rough grass, suggesting these were important habitats for breeding Corn Buntings in later years, especially during early summer. Set-aside and rough grass can also provide valuable late-summer breeding habitats, if patches remain uncut and not sprayed with herbicides. Although females did nest in set-aside and rough grass, their selection of this habitat was not strong (Chapter 4). However, loss of set-aside and rough grass reduces the availability of insect-rich foraging habitat for buntings provisioning chicks, potentially leading to poorer chick survival and lower fledging rates (Brickle *et al*. 2000).

The end of compulsory set-aside also means that fewer stubbles remain unploughed right through the winter into early spring, and most of those retained after the end of March in northeast Scotland were in set-aside (Chapter 6). The loss of set-aside could therefore suppress survival rates of Corn Buntings and other farmland birds dependent on seed-food in stubbles throughout the winter and into early spring.

8.3.6 Field enlargement and loss of boundary features

During our 20-year study (Chapter 3), the mean size of arable fields increased by 18% (from 7.14 ha to 8.45 ha), due to the removal of field-boundary features such as fences, walls, bushes and ditches. The maximum field size recorded across all of our study sites (AW's area 5 in 2006–08) was 59.7 ha. Despite tighter regulations to discourage farmers from removing certain types of field-boundary features (farmers receiving agricultural subsidies must obtain written permission from the Scottish Government's agriculture department or other statutory agencies before removing or destroying walls, hedges, field-boundary trees or watercourses), this process of field enlargement is ongoing. Larger fields create a more homogenous landscape, reducing the diversity of crop types for nesting, and the availability of field-boundary song-posts and invertebrate-rich foraging habitats (Chapter 4).

8.3.7. Technology

The driving force behind most of the changes listed above is technological advancement (Chapter 1). Crops have become less weedy over time not only because farmers are now applying herbicides over a wider crop area or more times per year than previously (Table 8.1), but also because the chemical products used are becoming increasingly efficient at controlling weeds (e.g. Ewald & Aebischer 2000). The area of wheat grown in northern areas such as Scotland has expanded (Table 8.1) partly because of the development of new hardy varieties tolerant of colder growing conditions (Wilson *et al*. 2009). One of the main reasons for field enlargement and associated removal of field-boundary features is to accommodate and make the most efficient use of increasingly large agricultural machinery, whilst modern combine harvesters spill little grain, leaving less seed food in stubbles for birds (Wilson *et al*. 2009). In livestock systems, earlier and more frequent mowing of forage grasses is possible partly because of the rapid growth of grass swards in response to inorganic fertilisers, whilst the development of a method to wrap cut grass in bales also encouraged the switch from hay to silage (Shrubb 2003). Regardless of CAP policy, agricultural technology will continue to improve, and first on the list of priorities in Cook's (2008) report looking to the future of agriculture in Aberdeenshire is 'major technological improvement'.

8.4. Conservation solutions

Whilst the future persistence of Corn Bunting populations in eastern Scotland and in other regions will ultimately depend upon the type of farming systems practised, agri-environment schemes have a key role to play. They are the main policy tool for delivering conservation measures on farmland, and with proper implementation and targeting, agri-environment schemes are effective at halting and reversing population declines. In Chapter 7, we demonstrated the success of targeted schemes for Corn Buntings in eastern Scotland. However, to maximise success, it was necessary to adapt the scheme during the study to incorporate delayed mowing of forage grasses, and to ensure annual sowing of cereal-based unharvested crops. Delayed mowing of forage grasses to late July or early August substantially increases Corn Bunting reproductive success, and its widespread adoption could increase overall annual productivity from 1.25 to 1.45 successful broods per female (Chapter 5). Cereal-rich unharvested crops are valuable foraging habitats for Corn Buntings throughout winter (Chapter 6), and with the inclusion of triticale, these crops now retain cereal grains and attract bunting flocks into late March.

8.4.1. Management options

Based on our findings and with reference to other studies, the following types of management should benefit Corn Buntings in arable or mixed farming systems in the UK and northwest Europe. Delivery would be through agri-environment schemes as specific management options, in combination with conventional cropping:

Delayed mowing of forage grasses to late July – protects nests otherwise destroyed during mowing in early summer; insect-rich foraging habitat in early-mid summer (Plates 7, 19).

Weedy (or under-sown) spring-sown cereals followed by overwinter stubble – weed-rich nest sites in mid-late summer; insect-rich foraging habitat in mid-late summer; seed-rich stubble in winter; newly-drilled fields give grain-rich habitat in early spring (Plates 3, 8, 14–16).

Summer fallow after overwinter stubble (not sprayed or cut until late summer) – insect-rich foraging habitat throughout summer; seed-rich habitat in late winter and early spring.

Unharvested cereals (conventional crop, or annually-sown mix including triticale, barley and oats) – winter seed food; weed-rich nest sites in mid-late summer (if no herbicides applied); insect-rich foraging habitat in mid-late summer (if no herbicides applied) (Plates 14, 17).

Rough grassland (field margins or larger patches) – insect-rich foraging habitat throughout summer; potential nesting habitat if patches large enough (at least 10–20 m wide); refuge for over-wintering invertebrates (Plate 15).

Field-boundary features (fence-lines, ditches, walls, bushes, small trees) – perches and song-posts for males; insect-rich foraging habitat throughout summer; refuge for over-wintering invertebrates (Plates 9, 15).

Supplementary feeding with cereal grain (for livestock or for wild birds) – provides additional seed food in winter or early spring.

Autumn-sown barley – nest sites in early summer; source of cereal grain in early summer; potentially insect-rich foraging habitat in early summer if weedy.

8.4.2. Targeting

Whilst some of the management options listed above are likely to prove beneficial for Corn Buntings in almost any farming system (e.g. weedy cereals, rough grassland, field-boundary features), others should not be regarded as universal solutions for all farming landscapes. Instead, conservation practitioners should consider what resources for Corn Buntings (and other species) the current farming system already provides, and use agri-environment measures to fill any resource gaps. For example, in Aberdeenshire and Inverness-shire where a high proportion of Corn Buntings nest in grasslands, delayed mowing of forage grasses was essential to achieving population increase, whereas in arable-dominated Fife and Angus, provision of cereal-based unharvested crops was sufficient to reverse population declines (Chapter 7). Other examples might include encouraging farmers in areas dominated by autumn-sown cereals to sow some crops in spring and retain over-winter stubbles, or advise those in wheat-dominated systems to grow some autumn-sown barley that would provide a source of grain food earlier in the summer.

Practitioners (and maybe the designers of agri-environment schemes) should also consider how management options could be linked together, allowing easier integration into farming systems. For example, from both a farmer's perspective and for Corn Bunting conservation, it might make sense to use a single field over two years for a weedy spring-sown cereal followed by overwinter stubble or left unharvested, and retained as fallow throughout the following spring and summer (Siriwardena 2010).

Finally, to maximise their effectiveness, management options designed to provide nesting habitat, such as delayed mowing of forage grasses and weedy spring-sown cereals, should target known nesting fields, or those in open areas away from woodlands. Their placement next to prominent song-posts such as wires will increase the likelihood of occupancy.

8.4.3. Scale of deployment required

As well as demonstrating that agri-environment scheme measures are capable of reversing Corn Bunting population declines in eastern Scotland, we also calculated the scale of deployment required to halt the national decline. Results of farm-scale population monitoring (Chapter 7) suggest that approximately 72% of the Corn Bunting population in mainland Scotland must receive targeted management through agri-environment schemes to halt the current decline, but in 2009, only 24% was targeted in this way. Targeted agri-environment scheme provision to the required level for Corn Buntings will cost approximately £120 000 per annum, with 500–600 ha under appropriate management. This is just 0.02% of annual subsidies paid to Scottish farmers, and 0.5% of land in the remaining mainland range of the Corn Bunting.

At the individual option level, for delayed mowing of forage grasses, at least 70% of females that nest in forage grasses would need to do so in meadows with delayed mowing (to 24 July) to achieve a 20% increase in the overall number of first broods fledged, to 0.45 per female (Chapter 5). Given that nests were found in one-fifth of meadows overall, this could be achieved by implementing delayed mowing across approximately 12–18% of the area under forage grasses on those farms with breeding Corn Buntings. For unharvested crops, the mean area per farm on farms deploying this option was approximately 2–3 ha, equivalent to 1.5–2.5 ha per km^2 (Chapter 7). Studies undertaken in eastern England suggest that the spatial distribution of such winter seed-providing habitats should be regular throughout the landscape, spaced 1–2 km from one another (Siriwardena *et al*. 2006, Siriwardena 2010). In regions such as eastern Scotland, where a large proportion of cereals are spring-sown and there are more overwinter stubbles, a lower rate of provision may be sufficient to meet the winter seed-food requirements of farmland birds. However, because Corn Buntings frequently nest in unharvested crops and use them as insect-rich foraging habitats during summer, their provision in 1–2 ha patches at 1–2 km intervals would be beneficial for this species even in landscapes with lots of overwinter stubbles.

8.5. Future prospects for Corn Buntings in Scotland

Based largely on the findings of the studies presented in this thesis, Scotland's main agri-environment scheme now includes a 'package' of options tailored to the habitat requirements of Corn Buntings (Table 8.2). With effective targeting, this scheme has the potential to halt and reverse the national Corn Bunting population decline. However, there is a risk that the full potential of the scheme will not be realised due to a number of constraints. These include insufficient uptake of the scheme, poor targeting and implementation of management options, and farmers' general reluctance to adopt within-field options (Butler *et al*. 2007). Overriding factors operating on a much larger scale, and affecting farmland bird conservation across Europe, include technological advances, rising commodity prices, climate change, and future changes to the CAP.

8.5.1. Further agricultural intensification and technological advances

Whilst the CAP has helped to accelerate changes in agriculture by providing financial support, advances in technology are the major driving force behind intensification that enables farmers to maintain their livelihoods and increase profits (Shrubb 2003). Intensification is an ongoing process, despite increased awareness (at least amongst policy-makers) of the cost to biodiversity and recent CAP reform giving more financial support for wildlife-friendly farming through agri-environment schemes. One potential factor that may ultimately slow down the rate of intensification is increasing costs of inputs such as fuel and fertilisers (both of which trebled during 2006–08), associated with rises in global oil prices. Indeed, in relation to the reaction of livestock farmers to higher input costs, Cook (2008) states that in Aberdeenshire 'the main trend of the last five years has been extensification'. As well as reducing livestock densities (there were small declines in total numbers of poultry, pigs and cattle across Aberdeenshire during 2003–2007), some farmers are beginning to use alternative, low-input or fuel-saving methods such as minimum tillage methods of crop cultivation, or growing nitrogen-fixing crops such as clover to reduce dependence upon inorganic fertilisers. Agricultural technology will continue to improve, regardless of the CAP policy, but will perhaps bring opportunities as well as threats to farmland biodiversity. For example, further development of ‘selective’ herbicides could enable farmers to control ‘problem’ weeds but allow other plant species to persist, thus enriching cereal fields with a diversity of arable flora and associated insects (Smith *et al*. 2009). Such products already exist, but their high cost relative to broad-spectrum herbicides discourages their widespread use. One of the biggest threats, however, would be the eventual public acceptance and adoption by EU farmers of genetically modified herbicide-tolerant crops, which would see the efficacy of herbicides used on these crops increase further, resulting in yet more losses of seed and invertebrate food for birds (Watkinson *et al*. 2000).

8.5.2. Climate change

Another factor that will have a major influence on farming systems in future years is climate change. In eastern Scotland, climate change predictions are for wetter winters and drier summers over the next few decades (http://www.ukcip.org.uk/uk-impacts/scotland/key-findings/ Accessed 11 April 2012), but it is difficult to forecast how this will affect the types of crops grown, or the timing of agricultural operations such as sowing, harvesting and ploughing of stubbles. This is especially so given that climate change is likely to impact upon farming systems across most of Europe, probably leading to large-scale geographical shift in the growing area of some crop types. For example, southeast England is currently suffering its worse drought for 30 years, such that already in April water companies have imposed hosepipe bans. The drought has severe implications for farmers growing water-demanding vegetables and other crops, and some have even decided not to grow anything this year, leaving fields fallow. If summers in the south and east of England become too hot and dry to grow potatoes and other vegetables, their production may shift to cooler wetter regions such as northeast Scotland (Cook 2008). A warmer climate may also allow the northward spread of crops such as maize, whose area in Scotland increased fourfold from 564 ha in 2004 to 2235 ha in 2010 (Scottish Government 2010b), and which I saw in Aberdeenshire for first time in 2011. A longer growing season may also encourage more farmers in eastern Scotland to grow autumn-sown wheat at the expense of barley, although local demand for spring-sown barley from the distilling industry is likely to continue in the near future (Cook 2008).

8.5.3. Farmer attitudes to agri-environment schemes

In the previous section, I said that practitioners should select agri-environment scheme options that fill resource gaps within the conventional farming system on a particular farm in order to benefit Corn Buntings. Unfortunately, in most cases this is not how practitioners and farmers select which options to implement. The main priority is rarely wildlife conservation, but rather is to maximise the farmer's income from agri-environment schemes by choosing options that are easy to implement and have the least effect on existing farming operations and crop-yields. Across Scotland, land management options within the Rural Priorities scheme with the greatest level of uptake are those involving the creation and management of hedgerows (2211 management agreements between April 2008 and March 2012 worth £41.7 million over 5 years), and maintaining buffer strips along watercourses to reduce diffuse pollution ('Water Margins and Enhanced Riparian Buffer Areas' – 2202 agreements worth £10.5 million over 5 years) (Scottish Government 2012). Another option that has so far attracted 2011 agreements worth £30.5 million over 5 years is 'Open Grazed or Wet Grassland for Wildlife'. This option requires reduced grazing levels, but in practice, farmers can easily achieve this with little change to existing management. Furthermore, the Rural Priorities spend over the same period on options that involve no land management included £93.6 million on restructuring agricultural businesses and £30 million on manure and slurry storage and treatment. By contrast, there were just 126 agreements worth £110 000 over 5 years for 'Biodiversity cropping on in-bye', and 64 agreements worth £594 000 over 5 years for 'Mown Grassland for Corn Buntings' (Table 8.2).

For species such as the Corn Bunting that requires within-field options to reduce the intensity of cereal or forage grass management, the current lack of selection of these options could severely limit the effectiveness of schemes. The study presented in Chapter 7 perhaps demonstrates this point, given that Yellowhammer and Reed Bunting populations increased in response to Rural Stewardship Scheme management that included few within-field options, but those of Corn Bunting did not. Worryingly, even when farmers and practitioners do select in-field options, in some cases their use is inappropriate. Applying the 'Mown Grassland for Corn Buntings' option to rough grassland, rather than to improved grass cut for silage or hay, is a good example. Similarly, options such as unharvested crops are often located on poor ground to generate income from land that is otherwise unusable, with little consideration for maximising their effectiveness for the target species (Chapter 6). Poor location of agri-environment fallow plot options in English schemes (47% were in fields adjacent to woodland) limited their use by ground-nesting birds such as Lapwings, which were recorded in just 40% of plots overall, and nested in only a quarter of the plots (Chamberlain *et al*. 2009).

8.5.4. CAP reform and future agri-environment schemes

A new CAP (2014–2020) is currently under development, bringing both threats and opportunities for farmland bird conservation. Whilst agri-environment schemes are likely to remain the main tool, and the most targeted mechanism, for delivering bird conservation on farmland, funding will be under increasing budgetary pressure in future. Discussion is focussing on how direct farming subsidies can be better justified by linking them to environmental outcomes (Hart & Baldock 2011, Hart *et al*. 2011). This 'greening' of subsidies may include the introduction of Ecological Focus Areas (EFAs), whereby all farmers must manage 5–10% of their land for biodiversity (Hart & Baldock 2011), and greater support for traditional, small-scale, extensive farming systems that maintain biodiversity, otherwise known as High Nature Value (HNV) farming (Paracchini *et al*. 2008). Both of these approaches have the potential to benefit Corn Buntings. EFA-style management in the Swiss agri-environment scheme resulted in a doubling of Yellowhammer populations over 5 years (Birrer *et al*. 2007), whilst many of the regions identified as supporting HNV farming, such as parts of southeast Europe and the plains of central Iberia (Paracchini *et al*. 2008), currently hold large populations of Corn Buntings. In the UK, most HNV farming systems are typically livestock-based and found in the upland margins of the north and west. However, there may be pockets of farmland in the lowlands of northeast Scotland that could be considered HNV, and potentially qualify for any future financial support for HNV farming.

There is also increasing political support for 'ecosystem services' such as carbon sequestration, diffuse pollution and water quality, flood defence, soil structure and forestry (Whittingham 2011). This has the potential to divert funding away from biodiversity objectives, although the two approaches are not mutually exclusive. Several management options designed for farmland birds should also provide ecosystem services (Bradbury *et al*. 2010), and there is growing evidence that by enhancing biodiversity on farmland, agri-environment schemes do deliver agronomic benefits via services such as pollination, biological pest control and improved soil structure (Whittingham 2011 and references therein). Options designed for ecosystem services can also benefit farmland birds. One of the most popular options in Scotland ('Water Margins and Enhanced Riparian Buffer Areas' – Table 8.2) involves leaving unmanaged buffer strips alongside watercourses to reduce diffuse pollution, and this also provides insect-rich foraging habitat for Corn Buntings, and rank herbaceous vegetation for nesting Yellowhammers, Reed Buntings and other species (Chapter 7). However, some options may be detrimental, such as the landscape enclosure effect of planting trees for carbon sequestration in open areas favoured by breeding Corn Buntings (Chapter 1).

There is also a desire amongst administrators and practitioners to simplify agri-environment schemes by reducing the number and complexity of management options. Whilst some 'simple' options have widespread applicability and biodiversity benefits (e.g. overwinter stubbles, summer fallows) with the potential for effective delivery by all farmers through non-competitive 'broad and shallow' schemes (e.g. in Scotland, Land Manager's Options), other more complex and species-specific management options are necessary (e.g. Mown Grassland for Corn Buntings). This is because management requirements can differ markedly between species, and often for the same species across different parts of its range. The Corn Bunting is a good example, with the main conservation solutions to population declines varying between northeast Scotland (delayed mowing of forage grasses), the Western Isles and Fife/Angus (provision of winter grain), and southern England (provision of late-harvested cereals). Agri-environment schemes have been most effective at halting and reversing population declines when targeting a single species, with management tailored to provide food or nesting habitat not readily available in conventional farming. In the UK, the three best examples to date are Cirl Bunting, Stone Curlew and Corncrake (Wilson *et al*. 2009). We can now add Corn Buntings in eastern Scotland to this list, but in the next generation of Scottish agri-environment schemes, it is essential to retain a package of options tailored for this species. To ensure they are used to best effect, complex and highly tailored options should only be available through competitive 'narrow and deep' schemes such as Scotland's current Rural Priorities scheme, and their deployment backed by advice from experts with sound knowledge of the target species.

8.6. Some general lessons for farmland bird conservation

Whilst the focus of this thesis was Corn Buntings in Scotland, several aspects have wider relevance to farmland bird conservation. Some of the most important ones are as follows.

8.6.1. Understanding fully the focal species' requirements

In Chapters 3 and 4, we showed that habitat associations of breeding Corn Buntings varied seasonally, and across years as the population size changed. The former demonstrates the dangers of drawing conclusions from studies that do not cover the entire breeding season. Mid-season shifts in breeding habitat association or territory location are likely to be frequent among multiple-brooded species within rapidly changing habitats such as farmland, but are poorly studied (Brambilla & Rubolini 2009, Gilroy *et al*. 2010). For effective conservation, it is critical to have a full understanding of a species' ecological requirements throughout the year.

Whilst constraints on and competition for funding often restrict the length of studies to two or three years, the between-year changes in habitat associations demonstrated in Chapter 3 highlights potential problems of short studies not detecting associations that may be important over the longer term. When studies are too brief, insights into possible causes of population declines (e.g. trends in land-use or populations; true baselines; temporal changes in habitat associations) are lost. Wherever possible, long-term data should be utilised to place the findings of short studies into historical context.

8.6.2. Adaptive management of agri-environment schemes

Monitoring is essential to assess the effectiveness of agri-environment schemes, but is also valuable because it can detect problems and allow adaptive improvement of management options. Agri-environment schemes targeted at Corn Buntings in Scotland were more effective after they had been adapted to include delayed mowing to allow sufficient time for nesting attempts in fields of forage grasses to successfully complete, and cereal-rich one-year unharvested crops to provide an annual source of overwinter cereal grain food (Chapters 5, 6 and 7). Other studies have also demonstrated how schemes can be improved by adapting existing options to improve their effectiveness (e.g. Douglas *et al*. 2009), or with the inclusion of entirely new management options (e.g. Morris *et al*. 2004, Buckingham *et al*. 2011).

In the previous section, I said that the unpopularity and lack of uptake of some management options could severely limit the effectiveness of Scottish agri-environment schemes for Corn Buntings, and this is a much wider problem (Butler *et al*. 2007, Davey *et al*. 2010a). In these current times of financial pressure on public funds, the possibility of raising the payment rates of options to make them more attractive to farmers seems unlikely. Adaptive management may therefore be necessary to ensure greater uptake of management options that are currently unpopular. One solution could be to adapt options to make them less onerous or costly for farmers to implement, as has been done with the 'Mown Grassland for Corn Buntings' option. The permissible mowing date for this option was brought forward by one week after our studies demonstrated this would be possible without significantly increasing the rate of nest loss to mowing (Chapter 5). Similarly, unharvested crop options could be made easier for farmers to implement by allowing part of a conventionally managed crop (e.g. cereal or oilseed rape) to be left unharvested, rather than requiring them to purchase and sow a special (and often expensive) seed mixture as the current schemes dictate (Chapter 6). However, another solution may be to adapt the way in which schemes are implemented. This could involve imposing greater restrictions on option choice, such that all management plans must include at least one within-field option, deployed on a scale appropriate for the target species.

8.6.3. Relating results from local studies to national biodiversity targets

Whilst there has been a large improvement in understanding of the effectiveness of agri-environment schemes in recent years, few studies have related local conservation initiatives to national biodiversity targets and species trends (Kleijn *et al*. 2011). Consequently, it is unknown to what extent agri-environment schemes have moderated biodiversity decline across Europe, despite the large financial investment in these schemes. Given that the EU aims to halt the loss of biodiversity by 2020, Kleijn *et al*. (2011) recommend that future studies assessing the effectiveness of agri-environment schemes should include scaling up the effects of local population responses to place them into the context of national species trends and objectives. The study presented in Chapter 7 does this, by using local population responses of Corn Buntings to agri-environment schemes to calculate the level of deployment needed at a national scale to halt the species' overall decline in Scotland. Whittingham (2011) describes this as a 'step forward' in agri-environment scheme monitoring studies, and it serves as a useful template for future studies to follow.

8.6.4. Tailoring agri-environment schemes to farming systems and local priorities

To maximise their effectiveness and value for money, agri-environment schemes should provide resources that are otherwise scarce or unavailable within the farming landscape (Davey *et al*. 2010b and see section 8.4.2). Therefore, the management options deployed should vary according to the farming system to which the scheme is applied. For example, in eastern Scotland, conservation solutions for Corn Buntings in arable-dominated areas mainly involve the provision of food resources, whereas in mixed farming landscapes an additional measure to protect nests in forage grasses is necessary (Chapter 7). Another example is the provision of undrilled patches in cereal fields to improve the reproductive success of Skylarks (Morris *et al*. 2004, Smith *et al*. 2009). This is a key option in England where most cereals are autumn-sown, but is not available in Scottish schemes because it is not necessary in spring-cropping and mixed farming systems.

Farmland bird conservation should also target species that are local as well as national conservation priorities. In northeast Scotland, because some aspects of agricultural intensification have been less apparent than in other parts of the UK (Table 8.1), farmland bird populations have remained relatively robust. Of 10 FBI species with significant long-term population declines across the UK (see Chapter 1, Table 1.1), only three (Corn Bunting, Yellowhammer and Grey Partridge) have shown a range contraction in northeast Scotland since the early 1980s (Table 8.3). Another three (Common Whitethroat, Tree Sparrow and Linnet) have shown a large expansion in their breeding range within the region, and population trends of farmland species have generally been more positive in Scotland than in England (see Table 1.1). It is right, therefore, that agri-environment schemes in eastern Scotland should focus on the declining population of Corn Buntings (Chapter 2), rather than the wider suite of seed-eating species whose populations within this region appear to be stable or increasing.

The intensity of the farming system upon which an agri-environment scheme is superimposed can also affect scheme performance and the scale of deployment necessary to halt species declines. Kleijn *et al*. (2011) hypothesise that agri-environment schemes will have a larger potential effect in low-input extensive farming systems than in high-input intensive systems, and in landscapes with low levels of semi-natural habitats than in those with either no or lots of semi-natural habitat. This is because, theoretically, farmland biodiversity declines exponentially with increasing land use intensity, and in landscapes with low levels (2–20%) of semi-natural habitat, species sources are still present but not in such numbers that they continually spill over onto intensively managed farmland from the surrounding landscape. Therefore, regional variation in population responses of species such as Corn Bunting, Yellowhammer and Reed Bunting to agri-environment schemes (Stevens & Bradbury 2006, Davey *et al*. 2010b, and see Chapter 7) could be attributable to regional variation in the intensity of farming systems. In regions with lots of spring-sown cereals and overwinter stubbles such as eastern Scotland, for example, unharvested crop patches will retain seed and attract birds later into the winter and spring than in landscapes dominated by autumn-sown crops such as southern and eastern England (Siriwardena *et al*. 2008 and see Chapter 6). Mechanisms for this include less depletion by target and non-target species (e.g. pigeons and corvids) of unharvested crop patches during the early-winter period in areas with lots of overwinter stubble. Therefore, to halt and reverse species declines, the required scale of deployment of winter seed options such as unharvested crops may be greater in intensively farmed areas such as southern and eastern England than in the less intensive systems found in eastern Scotland.

Finally, at a continental scale, Tryjanowski *et al*. (2011) suggest that more studies are needed from the low-input extensive farming systems of central and eastern Europe to better understand the ecology and habitat associations of farmland birds, and to develop regionally-adapted conservation solutions. The same argument could apply to the UK. More research on farmland birds in Scottish arable and mixed farming systems may reveal ecological mechanisms and habitat associations no longer detectable in the more intensive and specialised farming systems typical of many English regions. For example, our studies on Corn Buntings revealed a strong association with spring-sown cereals that are now scarce across much of England (Chapters 3 and 4), and that dense grass swards are a favoured nesting habitat in the mixed farming systems of northeast Scotland (Chapters 4 and 5). Small-scale plots of grass and spring cereals are now being trialled to provide safe nesting habitat for Corn Buntings in the autumn-cereal dominated arable systems of eastern England, demonstrating how the transfer of knowledge from one farming system or region to another can inform the design of agri-environment schemes.

8.7. Overall conclusions

Various aspects of their life-history and ecological traits (e.g. crop-nesting, late breeding season overlapping with crop harvest, dependence on cereal grain year-round and an abundance of chick-food invertebrates during summer, preference for open landscapes) make Corn Buntings particularly sensitive to agricultural intensification. However, within the UK at least, we now have a good understanding of the types of habitats and resources they require. Some management solutions have been tried and tested and do work, whilst others continue to be developed. Broadly speaking, we know what Corn Buntings need and how to manage farmland for them. The major hurdle that we have not yet crossed, though, is how to get these measures deployed on a sufficient scale throughout our farmed landscapes to halt and reverse population declines at a national scale. This is also true for most farmland species, and is a major challenge for policy makers at the national and European level. If we are to meet the EU's objective of halting biodiversity loss by 2020, the forthcoming reform of the CAP must include policy mechanisms to ensure greater support for HNV farming systems, and wider uptake of those agri-environment measures that deliver the greatest biodiversity benefits, supported by expert advice where necessary for effective targeting. Conservation scientists also have an important role to play, by continuing to develop and adapt management solutions for particular farming systems, and to make them more acceptable to farmers whilst still delivering substantial benefits for the target species. Our knowledge base of the ecology of European farmland birds is huge and continues to grow, but the focus of future research must involve the monitoring and adaptive management of conservation solutions, primarily those within agri-environment schemes.

Table 8.1. *Area covered by each major crop type and livestock numbers in Scotland and two English regions in 1985 and 2010. Data for northeast Scotland in 2000 and 2010 are also shown (1985 data were unavailable for this region), as are data on herbicide use in cereals for 1990 and 2010. Sources: Scottish Executive (2001), Scottish Government (2010b, 2010c), DEFRA (2012).*

Area (x 1000 ha)	**Southeast England** [a]		**East England** [b]		**Scotland**		**Northeast Scotland**	
	1985	**2010**	**1985**	**2010**	**1985**	**2010**	**2000**	**2010**
Autumn-cereals [c]	na	272	na	577	182	167	36	36
Spring-cereals [c]	na	53	na	60	346	257	106	103
Wheat	296	241	536	502	82	111	17	18
Oats	25	19	14	9	30	23	6	5
Barley	196	65	318	126	416	290	120	116
Oilseed rape	35	85	70	141	23	36	15	13
Crop for animfeed	31	2	49	3	52	23	7	5
Veg excl potatoes	15	8	53	30	8	16	2	2
Potatoes	11	4	35	32	34	31	6	6
Sugarbeet	<1	<1	115	81	<1	<1	0	0
Legumes	na	38	na	64	19	15	<1	<1
Maize	na	22	na	8	na	2	na	na
Dairy cattle	231	129	81	38	432	269	17	13
Beef cows [d]	48	76	35	43	435	457	99	91
Sheep	1761	1177	372	310	8578	6753	704	608
Pigs	795	205	1608	1032	420	409	353	278
Poultry	14524	9340	17649	28496	13223	14593	3035	2773
Grass <5years	145	72	40	34	470	423	91	125
Grass ≥ 5years	352	395	137	180	614	955	100	85
Rough grazing	na	16	na	16	4432	3775	206	231
Arable fallow	na	27	na	37	9	22	34	6
Herbicide use in cereals [e]	14.0	24.3	14.9	27.2	9.9	17.8	na	na
Total arable	635	526	1203	1024	676	572	203	174
Total grass	497	489	177	230	5517	5155	397	441
Total farmed area	1273	1141	1495	1381	6318	6227	645	702

[a] *Berkshire, Buckinghamshire, Hampshire, Isle of Wight, Kent, Oxfordshire, Surrey, Sussex.*
[b] *Bedfordshire, Cambridgeshire, Essex, Hertfordshire, Norfolk, Suffolk.*
[c] *assuming all wheat and 30% of oats are autumn-sown.*
[d] *refers to size of the beef breeding herd.*
[e] *data are from 1990 and 2010, and show the active substance treated area divided by the area of cereals.*

Table 8.2. *Agri-environment management options targeted at Corn Buntings in mainland Scotland through the Rural Priorities and Land Managers' Options schemes, and national uptake and expenditure up until the end of March 2012. Source: Scottish Government (2010a, 2012).*

Management option	Payment rate £ per ha per year	Description	Main resource for Corn Buntings	Uptake
Rural Priorities				
Wild bird seed mix/unharvested crop [L]	391.26	Spring-sown crop mix that must include a cereal, left unharvested and unploughed until 15 March the following year. Max. plot size 2 ha. Pesticides only allowed to aid crop establishment. 2-year crops are permitted in Corn Bunting areas, but must involve at least two plots sown in alternate years. Must be located next to a hedge, isolated bush/tree, or wires, but not dense woodland.	Winter seeds; summer insects; nesting habitat	1106 / £4.1m
Biodiversity cropping on in-bye [L]	70.94 / 400	Spring cereal, fodder roots or fodder rape in 2 ha plots, max. 4 ha per farm. Cultivations/fertiliser applications only allowed between 1 March and 15 May (with exceptions for rape/roots, but any nests must be marked and avoided). No pesticides allowed, No ploughing before 1 March the next year. Arable silage not permitted. Premium payment where a cereal is harvested by binder and stooks gathered into stacks. Overwinter grazing permitted.	Summer insects; nesting habitat; winter seeds	126 / £0.1m
Management of species rich grassland	111.00	In Corn Bunting areas, no mowing or grazing permitted from 16 April to 15 August, but must manage grazing levels to create sward 5–15 cm tall in September–March in neutral and acid grasslands, or 2–10 cm tall in calcareous grasslands. Where no grazing, must cut once in autumn or next spring. Cuttings must be turned in field then removed. No fertilisers or pesticides allowed.	Summer insects; nesting habitat	1334 / £8.0m
Creation and management of species rich grassland	223.57	As above, but involves establishment of a new species-rich sward by sowing low productivity grasses and herbs.	Summer insects; nesting habitat	335 / £2.2m
Water margins and enhanced riparian buffer areas	286.63	Maintain 12–24 m wide margin alongside still water, or 3–12 m wide margins either side of a watercourse (minimum width dependent on the bed width of the watercourse). Light grazing/cutting permitted to control rank vegetation growth. No pesticides, fertilisers or cultivations allowed.	Summer insects	2202 / £10.5m

Table 8.2 cont.

Management option	Payment rate £ per ha per year	Description	Main resource for Corn Buntings	Uptake
Rural Priorities				
Grass margins and beetlebanks [L]	473.76 in year 1, then 407.92	3–6 m wide grass strip around arable fields. Establish by sowing a grass mix including at least one nectar source (e.g. clover). No pesticides, fertilisers or scrub control allowed. Grazing allowed down to 10 cm.	Summer insects	544 / £2.8m
Mown grassland for Corn Buntings	216.00 / 224.48	Exclude livestock from hay or silage fields from 1 May and delay mowing until after 24 July or 1 August (premium payment). Fields must be cut, but in a wildlife-friendly manner (from the centre outwards). No rolling, harrowing or grazing from 1 May until after the field has been cut. 2 m boundary strip must be left uncut.	Nesting habitat	64 / £0.6m
Total including forestry, business and rural development				£501m
Total agri-environment schemes				£190m
Land Managers' Options (additional to [L] above)				
Retention of winter stubbles	96.00	Applies to cereals, oilseeds and protein crops. Must not be ploughed or cultivated before 1 March. Not applicable to under-sown crops or arable silage. No pre-harvest desiccants or post-harvest pesticides allowed. Grazing permitted.	Winter seed	Data unavail able
Management of conservation headlands	70.00 / 135.14	Applies to cereals, oilseeds and protein crops. Not applicable to arable silage. No pesticides allowed. Minimum 6 m wide, and headland must go all the way round a field. Premium payment for no use of nitrogenous fertilisers.	Summer insects	Data unavail able

Table 8.3. *Breeding season distribution (2-km squares), estimated population size and proportion of the UK population found in northeast Scotland of eight farmland species with long-term UK population declines. Data relate to Aberdeenshire and Moray in 2002–2006. Also shown are the percentage changes in distribution since 1968–72 (occupied 10-km squares) and since 1981–84 (occupied recording units). The two other farmland species with long-term UK declines (European Turtle Dove and Yellow Wagtail) do not regularly occur during the breeding season in northeast Scotland as they have a more southerly breeding distribution. Source: Francis & Cook (2011).*

Species	Occupied 2-km squares	% change since 1968–72	% change since 1981–84	Breeding pairs	% UK popn
Grey Partridge	539	-16	-9	3400–3650	5
Skylark	1648	-4	+2	50000	3
Common Whitethroat	943	+5	+30	7500	<1
Common Starling	1451	-8	+1	40000–50000	6
Eurasian Tree Sparrow	520	+49	+174	2500–3500	4
Eurasian Linnet	1264	-1	+40	18000–22000	4
Yellowhammer	1347	-11	-8	40000–45000	5
Corn Bunting	280	-34	-26	550–600	6

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