

BTO Research Report No.191

**A Review of the Status and
Population Trends of Bird Species
on the Short and Middle
UK Biodiversity Steering Group Lists**

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EXECUTIVE SUMMARY

1. This report provides a review of the status and population trends of bird species on the short and middle UK Biodiversity Steering Group lists. Nine bird species are included on the short list: Aquatic Warbler, Skylark, Bittern, Stone Curlew, Corncrake, Scottish Crossbill, Grey Partridge, Capercaillie and Song Thrush. Action plans have already been produced for these species (Anon., 1995). Sixteen bird species are included on the middle list: Marsh Warbler, Nightjar, Linnet, Cirl Bunting, Reed Bunting, Wryneck, Red-backed Shrike, Woodlark, Common Scoter, Corn Bunting, Spotted Flycatcher, Tree Sparrow, Red-necked Phalarope, Bullfinch, Roseate Tern and Turtle Dove.
2. The first section of each account details the species' positions on the lists of the UK Biodiversity Steering Group, Species of Conservation Concern (Gibbons *et al.*, 1996) and Species of Conservation Importance (JNCC, 1996). Species of European Conservation Concern (Tucker & Heath, 1994) and those on the IUCN Red List are also indicated. Status within the EC Wild Birds Directive, Berne Convention, the Wildlife and Countryside Act 1981, the Wildlife Order (Northern Ireland) 1985 and other relevant acts is given. European population estimates are provided from the EBCC Atlas of European Breeding Birds (Hagemeijer & Blair, 1997).
3. Population trends are described using information from the BTO/JNCC Integrated Population Monitoring programme, from specific surveys and from the Rare Breeding Birds Panel. Latest population estimates are provided.
4. A second section summarizes existing knowledge of factors affecting populations. Although discussion centres on UK populations, the results of other European studies were also drawn upon.
5. In the final section, present research is summarized and the organisations undertaking it detailed. Topics requiring future research are also listed, together with the government bodies which the Biodiversity Steering Group action plans list as responsible for their undertaking.

1. INTRODUCTION

This report provides a literature review of the status and population trends of bird species on the short and middle UK Biodiversity Steering Group lists. Nine bird species are included amongst the 116 species on the short list and 16 among the nearly 300 species on the middle list. Action plans were produced for short list species in 1995 (Anon., 1995) and plans for those on the middle list are nearing completion.

Species accounts are divided into three sections: status and population trends, factors affecting populations and present and future research. Specific references are given at the end of each account and other general references later. In the first section of each account, the species' positions on the lists of the UK Biodiversity Steering Group, Species of Conservation Concern (Gibbons *et al.*, 1996) and Species of Conservation Importance (Joint Nature Conservation Committee (JNCC), 1996) are given. Species of European Conservation Concern (SPEC) (Tucker & Heath, 1994) and those on the International Union for the Conservation of Nature (IUCN) Red List are indicated and the reasons for these listings given. The status of each species within the EC Wild Birds Directive, Berne Convention, the Wildlife and Countryside Act (WCA) 1981, the Wildlife Order (Northern Ireland) (NI) 1985 and other relevant acts is also indicated. European population sizes are taken from the European Bird Census Council (EBCC) Atlas of European Breeding Birds (Hagemeijer & Blair, 1997).

Information on status and population trends is taken from a variety of sources. For more common species, data are drawn from long-running British Trust for Ornithology (BTO) schemes, which form part of the BTO/JNCC Integrated Population Monitoring (IPM) programme. The Common Birds Census (CBC) provides estimates of population trends from annual counts of birds during the breeding season on between 200 and 300 plots from around the UK. This survey has been running since 1962, but due to limitations it has recently been superseded by the BTO/JNCC/Royal Society for the Protection of Birds (RSPB) Breeding Bird Survey (BBS). This provides data on breeding bird populations from a stratified random sample of 1 km² survey squares within 83 sampling regions across the country. It has been running concurrently with the CBC since 1994. Results from the first three years of surveying are reported by Gregory *et al.* (1997). The Waterways Bird Survey (WBS) has monitored the breeding status of 19 riparian bird species on canals and rivers throughout the UK since 1974. There are currently 120 plots around the country. The Constant-Effort Sites (CES) Scheme uses catches of birds from 100 or more standardised mist-netting sites to monitor changes in the abundance of common passerines and their breeding success. Catch sites are in scrub or wetland habitats. The Nest Record Scheme (NRS) gathers data on the breeding performance of birds in the UK, through the recording of individual breeding attempts. The NRS provides information on laying dates, clutch sizes, brood sizes and daily nest failure rates during both egg and nestling stages. A summary of results from the IPM programme for the period 1971-1995 is provided by Crick *et al.* (1997), which also provides a more detailed account of each of the schemes. It is from this report that information on the population trends of commoner species has been taken. Graphs taken from the report show population trends from the CBC and WBS and trends in breeding performance from the NRS. Information on breeding range changes is taken from the *New Atlas of Breeding Birds in Britain and Ireland: 1988-1991* (BTO/Scottish Ornithologists Club (SOC)/Irish Wildbird Conservancy (IWC): Gibbons *et al.*, 1993). This publication provides a comparison with an earlier (first) atlas based on survey work carried between 1968 and 1972: *The Atlas of Breeding Birds in Britain and Ireland* (BTO/IWC: Sharrock, 1976). It also provides population estimates.

For rarer species, information is taken from the results of individual surveys and from the reports of the Rare Breeding Birds Panel (RBBP) (JNCC). Recent population estimates for all species are given by the Avian Population Estimates Panel (Stone *et al.*, (1997).

The second section summarizes existing knowledge of factors affecting populations. Although discussion centres on UK populations, the results of other European studies were also drawn upon. In the final section, present research is summarized and those organisations undertaking it indicated. These include the BTO, the Game Conservancy Trust (GCT), the RSPB and the Wildfowl and Wetlands Trust (WWT).

Topics requiring future research are also detailed, together with the government bodies which the Biodiversity Steering Group action plans list as responsible for their undertaking. These are:

The Countryside Council for Wales (CCW)
The Department of Agriculture, Northern Ireland (DANI)
The Department of the Environment, Transport and the Regions (DoETR)
The Environmental and Heritage Service (EHS) of the Department of the Environment for Northern Ireland
English Nature (EN)
The Forestry Authority (FA)
The Forestry Commission (FC)
The Joint Nature Conservation Committee (JNCC)
The Ministry of Agriculture, Fisheries and Food (MAFF)
Scottish Natural Heritage (SNH)
The Scottish Environmental Protection Agency (SEPA)
The Scottish Office, Agriculture, Environment and Fisheries Department (SOAEFD)
The Welsh Office Agriculture Department (WOAD)

2. SPECIES ACCOUNTS *Species on the Biodiversity Steering Group Short List*

2.1 Aquatic Warbler *Acrocephalus paludicola*

Status and Population Trends

IUCN Red List (Globally Threatened)

Current European Population 2,900-7,900 pairs

SPEC Category 1

SPEC Status Endangered (large decline; <10,000 pairs)

EC Birds Directive Annex I

Berne Convention Appendix II

Biodiversity Steering Group Short List

Species of Conservation Concern Red List

Species of Conservation Importance Table 1

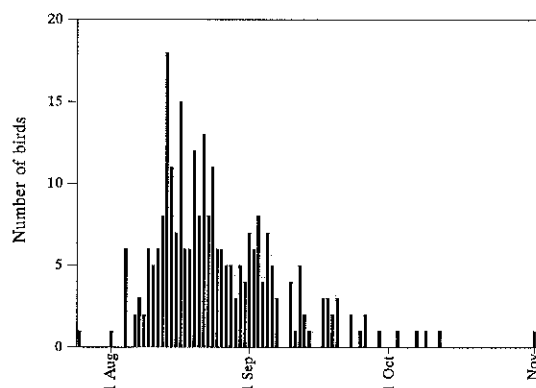


Figure 2.1.1 Dates of first occurrence of Aquatic Warblers in Dorset from 1971 to 1994 (from Lewis, 1996).

The breeding distribution of the Aquatic Warbler is restricted to eastern Europe, from Germany in the west to the River Ob in the east and south to Hungary (Cramp, 1992; Ravkin, 1993; Hagemejier & Blair, 1997). In winter it is found in wetlands in west Africa, from Senegal and Mali to Ghana (Hagemejier & Blair, 1997). The Aquatic Warbler is a rare but regular autumn migrant to wetland sites in the southern UK and also occurs at coastal sites in the Netherlands, Belgium, France and Portugal (de By, 1990). It is the rarest and most threatened passerine in the west Palearctic (Tucker & Heath, 1994).

The size of the Aquatic Warbler population that passes through the UK each year has yet to be estimated and there is no firm evidence as to whether numbers are increasing or decreasing. There has certainly been an increase in records and this led to the species being removed from the British Birds Rarities Committee list in 1982. There were, for example, 60 records in the UK in 1991 (Evans, 1991). The rise in records is probably a result of an increase in the number of observers and intensive ringing effort (Lewis, 1996). Of 395 records from 1969 to 1982, 251 (63.5%) were from ringing (de By, 1990). The majority of both ringing and sight records are from south coast counties; few are found in eastern counties (Sharrock & Sharrock, 1974; de By, 1990; Lewis, 1996). The passage of Aquatic Warblers through the UK is concentrated between mid-August and early September (Figure 2.1.1; Sharrock & Sharrock, 1974; Pattenden, 1988; de By, 1990; Lewis, 1996).

Trends in the sizes of autumn passage populations have been more closely monitored on the continent. Schuster *et al.* (1983), for example, found that records of Aquatic Warblers in the Bodensee area on the German/Swiss border fell from 89 in the 1950s to 49 in the 1970s, mainly due to a decrease in autumn records. Berthold *et al.* (1986) similarly found a fall in population level between 1974 and 1983 in their standardized mist-netting study in central Europe. De By (1990) also reports that Aquatic Warblers were more numerous than Sedge Warblers *Acrocephalus schoenobaenus* as victims of Dutch lighthouses in the first half of the century, thus indicating their former abundance.

The Aquatic Warbler's present disjunct breeding distribution is a relict of a once more continuous range. Since 1930 the species has become extinct in Austria, Bulgaria, France, Italy, the Netherlands, Romania, Slovakia and the former Yugoslavia (Tucker & Heath, 1994) and is now almost extinct in Germany (Glutz von Blotzheim & Bauer, 1991). Estimates of the Aquatic Warbler's present breeding populations are complicated by its polygynous and almost promiscuous mating system (Dyrce, 1989; Dyrce & Zdunek, 1993; Schulze-Hagen *et al.*, 1995), which makes it difficult to convert counts of singing males into numbers of breeding females (Dyrce & Czeraszkiwicz, 1993). Populations in Poland, its known stronghold, have declined to an estimated maximum of 7,640 males (Dyrce & Czeraszkiwicz, 1993). Up to 6,000 of these are in the Biebrza marshes, where numbers have fluctuated, but have shown no clear decline (Dyrce & Czeraszkiwicz, 1993; Dyrce & Zdunek, 1993). Preliminary surveys in Belarus found 5,000-20,000 singing males (Kozulin & Flade, 1997). There are also estimated average numbers of 4,500 singing males in both Russia and Ukraine (Hagemejier & Blair, 1997). There have been two ringing recoveries linking the UK with breeding grounds: two birds of the same brood ringed in Poland in June 1990 were recovered in Avon and Cornwall in August of the same year (Mead & Clark, 1991).

Factors affecting Populations

The Aquatic Warbler populations that pass through the UK in autumn are clearly dependent upon the number and quality of wetland habitats in southern counties. The loss of wetlands through this century may have led to a decline in passage numbers, but presently all known key sites are designated as nature reserves or Sites of Special Scientific Interest (SSSIs). Aquatic Warbler numbers in the UK and the timing of their occurrence may also vary annually according to weather patterns (Lewis, 1996).

The long-term future of the Aquatic Warbler in the UK is probably largely dependent upon the conservation of the species on its breeding grounds. The species' preferred breeding habitat is open, eutrophic marsh, dominated by sedge *Carex* and mosses, though it also occurs in partially drained hay meadows, calcareous marshes and in some coastal saltmarshes (Dyrce, 1993; Dyrce & Czeraszewicz, 1993; Dyrce & Zdunek, 1993; Hagemeyer & Blair, 1997). The loss of such habitats to land drainage associated with agricultural and industrial development, and the abandonment of traditional farming practices threatens many breeding populations in Poland and the former USSR (Dyrce & Czeraszewicz, 1993). On the Hortobágy in Hungary, however, active flooding of grasslands has helped to increase an Aquatic Warbler population from c. 20 singing males in 1977 to over 400 in 1994 (Kovács, 1994).

Present and Future Research

There is an obvious need to identify a method for monitoring Aquatic Warblers on passage through the UK and to implement a monitoring programme (CCW, EN, JNCC). At present, due to its secretive habits, the timing of the species' migration and its distribution in this country are best known from ringing activities (de By, 1990; Lewis, 1996). By tape-luring at night, for example, 300-400 birds were caught in Belgium and Holland between 1989 and 1991 (Hereida *et al.*, 1996). The use of tape-luring at night, however, may cause birds to stop at inappropriate sites that they would not have used otherwise (the species only migrates at night: de By, 1990) and may distort results. In order to identify passage sites, therefore, tape-luring should be restricted to daytime. Once key passage sites are identified (EN), the ecology and habitat use of the species also needs to be researched (EN).

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2.2 Skylark *Alauda arvensis*

Status and Population Trends

Current European Population 27,900,000-35,200,000 pairs

SPEC Category 3

SPEC Status Vulnerable (large decline)

EC Birds Directive

Berne Convention Appendix III

WCA 1981

Wildlife (NI) Order 1985

Biodiversity Steering Group Short List

Species of Conservation Concern Red List

Species of Conservation Importance Table 3

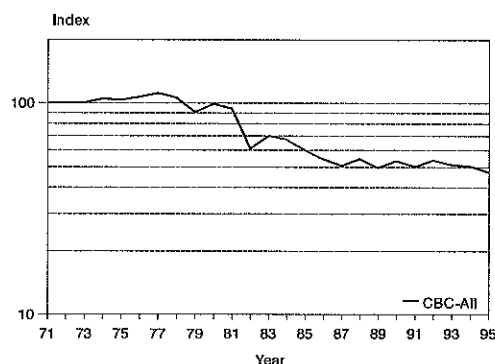


Figure 2.2.1 Population trend from the Common Birds Census

The range of the Skylark extends across central and northern Europe, through central Asia to the Pacific (Cramp, 1988; Hagemeyer & Blair, 1997). The nominate *Alauda arvensis arvensis* breeds in England, Wales and north Europe as far east as the Urals, whilst the subspecies *scotica* is endemic to Scotland, Ireland and the Faroes. In the UK, Skylarks leave high-altitude areas in winter, but elsewhere are largely sedentary (Hardman, 1974).

The Skylark occurs across a wide variety of open habitats and in the past benefited greatly from the clearance of forests and the increase in arable farmland (Tucker & Heath, 1994; Hagemeyer & Blair, 1997). Between the mid-1970s and mid-1980s, however, the species underwent a sharp decline. CBC data show that the population fell by 62% between 1971 and 1995 and by 61% on farmland alone (Crick *et al.*, 1997; Fig. 2.2.1). The population has been relatively stable since then and the BBS recorded only a 1% decline between 1994 and 1996 (Gregory *et al.*, 1997). Breeding atlas data show that the Skylark's range in Britain contracted by 2% between 1968-72 and 1988-91, with most losses in upland areas of Scotland, and by 6% in Ireland (Gibbons *et al.*, 1993). The British population was estimated at 2,000,000 territories in 1988-91 and that in Ireland at 570,000 (Gibbons *et al.*, 1993).

Population declines have also been recorded in many other European countries since the 1960s (Tucker & Heath, 1994; Hagemeyer & Blair, 1997). In Finland, for example, numbers of observed migrants fell by 10% a year between 1981 and 1988 (Norrdahl, 1990). In Germany, a decline of 93% was recorded between 1973 and 1986 on an area of grazed saltmarsh (Busche, 1989).

Factors affecting Populations

In the UK, declines in Skylark numbers have been greatest on farmland habitats and it is believed that this has been largely due to the intensification of agriculture. As with other seed-eating species, Skylarks show a particular preference for winter stubbles (Green, 1978; Wilson *et al.*, 1996) and their loss following the switch from spring- to autumn-sown cereals is likely to have increased winter mortality rates. The quality of remaining stubbles has also declined, as grain is now harvested more efficiently and as weeds are now more efficiently controlled with herbicides (Campbell *et al.*, 1997).

The population decline in the UK is not believed to have been a result of a reduced breeding performance (Chamberlain & Crick, in press). NRS data show that both clutch and brood sizes increased between 1971 and 1995 (Crick *et al.*, 1997; Fig. 2.2.2). It is likely, though, that the population decline is in part due to reductions in the proportion of the population that breeds each year and in the proportion that attempt to raise more than one brood (Chamberlain & Crick, in press). Breeding success and survival data (e.g. Delius, 1965) suggest that two to three nesting attempts are required for populations to be self-sustaining (Jenny, 1990a; Wilson *et al.*, in press). Reductions in crop diversity, in particular, have meant that pairs are often unable to nest in the same area more than once in a season (Schläpfer,

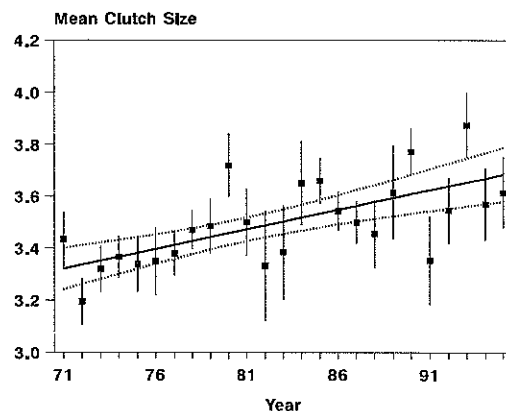


Figure 2.2.2 Long-term trend from the Nest Record Scheme

1988; Jenny, 1990b). Intensive pasture habitats, for example, become unsuitable very early in the year as the use of fertilizers leads to vegetation that is too tall and dense for nesting (Busche, 1989). Winter cereals are also unsuitable as breeding habitats, as they are too tall for nesting (Schläpfer, 1988; Wilson *et al.*, in press) and provide little food in comparison to other habitats, partly due to the widespread use of herbicides and insecticides (Jenny, 1990c; Campbell *et al.*, 1997). In broad-leaved crops such as oilseed rape and legumes, Skylarks form territories, but are unable to nest as crops grow too quickly (Wilson *et al.*, in press). In contrast, set-aside, organically farmed fields, upland and semi-natural grassland and non-agricultural habitats, such as saltmarshes and sand dunes, remain suitable for nesting throughout the breeding season and thus hold higher densities (Busche, 1989; Tucker & Heath, 1994; Wilson *et al.*, in press; Chamberlain & Gregory, in prep.).

Present and Future Research

The Skylark's status is currently monitored by the BBS and breeding productivity assessed by the NRS (BTO, JNCC, RSPB). A BTO survey in 1997 aimed to estimate the size of the UK population, to compare breeding densities on different habitats (including different farmland types, heather moorland, lowland heath, young forestry, saltmarsh, dunes and coastal grazing marsh) and to examine the seasonal pattern of territory occupancy in different crops. A further survey in 1997/98 aims to assess the Skylark's winter distribution and population in Britain, to determine habitat preferences and to analyse movements both within and to and from Britain (BTO, JNCC). A recent study in Dorset and Hampshire showed that chicks were fed more frequently and with larger quantities of food in set-aside than in grass or spring barley fields (Poulsen, 1996). Current RSPB research aims to relate habitat and food supply to diet and breeding success.

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2.3 Bittern *Botaurus stellaris*

Status and Population Trends

Current European Population 10,000-11,700 pairs

SPEC Category 3

SPEC Status Vulnerable (large decline)

EC Birds Directive Annex I

Berne Convention Appendix II

WCA 1981 Schedule 1

Wildlife (NI) Order 1985 Schedule 1

Biodiversity Steering Group Short List

Species of Conservation Concern Red List

Species of Conservation Importance Table 2

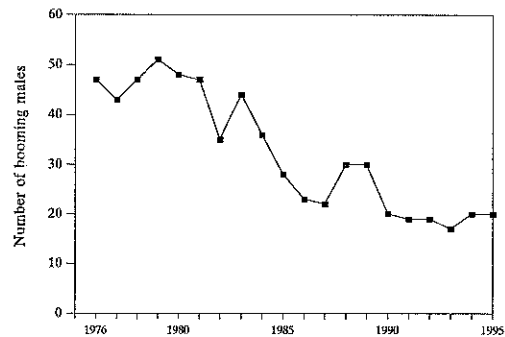


Figure 2.3.1 Maximum numbers of booming male Bitterns in the UK between 1976 and 1995 (from Day & Wilson, 1978; Spencer *et al.*, 1989; Ogilvie *et al.*, 1996b; Stone *et al.*, 1997).

The range of the Bittern extends across the Palearctic region from western Europe, through central Russia to the Pacific (Cramp & Simmons, 1977; Hagemeijer & Blair, 1997). There is also an isolated population in southern Africa (Hancock & Elliot, 1978). Its European distribution is fragmented, due its requirement for densely vegetated marshes, typically dominated by reeds *Phragmites*. The UK is at the north-western limit of its breeding range (Hagemeijer & Blair, 1997).

The Bittern was widespread throughout England, Wales and southern Scotland in the 17th century, but land drainage and hunting led to a decline and ultimately to extinction in the 1880s (Tyler, 1994; Holloway, 1996). Breeding was first confirmed again in 1911 and with protection, the population grew to around 70 pairs in the 1960s, distributed through eight counties (Smith & Tyler, 1993; Tyler, 1994). The Bittern's population and range in the UK have declined sharply since then, however, falling from 47 booming males in 1976 (Day & Wilson, 1978) to 20 in 1995 (Fig. 2.3.1; Stone *et al.*, 1997). Only 16 different sites were occupied in 1994, 12 of which were in East Anglia (Ogilvie *et al.*, 1996b). The largest declines have been in the Norfolk Broads, the Bittern's former breeding stronghold, where only three sites were occupied in 1990 (Smith & Tyler, 1993). Between 50 and 150 Bitterns are found in the UK in winter, some of continental origin (Bibby, 1981; Lack, 1986).

The population changes recorded in the UK reflect those seen across Europe. After declines in the 19th century, resulting from drainage of breeding areas and hunting, Bitterns colonized southern Sweden and Finland at the start of the century (Hagemeijer & Blair, 1997). Since 1970, however, declines have been recorded in half the countries in which the species breeds (Day, 1981; Hagemeijer & Blair, 1997). Eastern European populations have remained generally stable and increases have even been recorded in Estonia and Denmark (Hagemeijer & Blair, 1997).

Factors affecting Populations

The probable causes of the recent decline of the Bittern as a breeding bird in the UK include habitat deterioration and loss, disturbance, poisoning and hard weather (Day & Wilson, 1978; Smith & Tyler, 1993). The Bittern is particularly susceptible to severe winter weather and cold winters may cause influxes of continental birds into the UK (Bibby, 1981). Although in this country the severity of winter weather is correlated with winter mortality, it does not appear to affect the subsequent number of breeding birds (Bibby, 1981). In Finland, in contrast, following a period of mild winters the number of booming males increased from 70 in 1988 to 194 in 1990 (Koskimies, 1992).

Many large reedbeds have been lost over recent decades through drainage and only a few are now large enough to support Bitterns - research at Leighton Moss in Lancashire has shown that individuals may have home ranges of up to 70 ha (Tyler, 1992a, 1994). This research has also shown that Bitterns prefer to feed close to open water at the edges of reedbeds, where there is a greater variety of prey (Tyler 1992a; Smith & Tyler, 1994). However, much of this reedswamp edge has been lost, particularly in the Broads, as a result of water pollution (especially eutrophication caused by runoff from domestic and agricultural sources), increased boat traffic and, in some cases, grazing (George, 1992). Reduced reed harvesting has also led to scrub encroachment and the loss of waterways (Gibbons *et al.*, 1993). Those sites which have retained Bitterns have wetter reedbeds and more open water than those which have lost birds (Tyler, 1992b). Recent studies in the UK have also shown that Bitterns have been contaminated with the residues

of some pollutants including organochlorine pesticides, PCBs and mercury and that birds may accumulate these at fatal levels (Newton *et al.*, 1994). Other threats include increased disturbance from anglers, water-sports and other human activities (Hölzinger, 1987).

Present and Future Research

About 85% of UK Bitterns breed on nature reserves (65% at three SSSIs: Walberswick and Minsmere in Suffolk and Leighton Moss in Lancashire) and are thus currently protected (Batten *et al.*, 1990). However, they are extremely secretive and estimating their numbers is usually limited to counts of 'booming' males (Smith & Tyler, 1993). Techniques have recently been developed to identify individual Bitterns by their voices and this has allowed an accurate evaluation of the UK population (McGregor & Byle, 1992; Gilbert *et al.*, 1994; RSPB). Detailed RSPB research has led to a good understanding of the Bittern's habitat requirements (Tyler, 1992a, 1992b). This research should form the basis for the monitoring of reedbed habitats and food availability at key Bittern sites (CCW, EN). The results of this research are already being used by the RSPB and EN in management plans at present Bittern breeding sites and for reedbed creation. There should be continued analysis of Bittern corpses and addled eggs for heavy metals and pesticides (CCW, EN).

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2.4 Stone Curlew *Burhinus oedicnemus*

Status and Population Trends

Current European Population 32,700-45,700 pairs

SPEC Category 3

SPEC Status Vulnerable (large decline)

EC Birds Directive Annex I

Berne Convention Appendix II

WCA 1981 Schedule 1

Biodiversity Steering Group Short List

Species of Conservation Concern Red List

Species of Conservation Importance Table 2

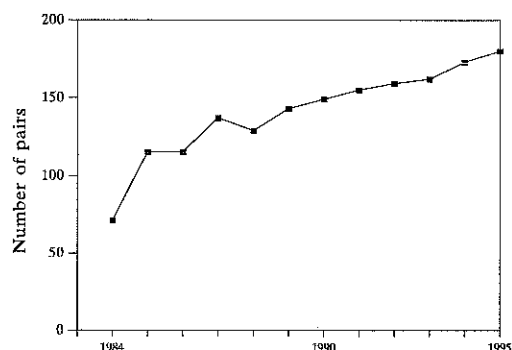


Figure 2.4.1 Maximum possible numbers of breeding Stone Curlews in the UK between 1984 and 1995 (from Ogilvie *et al.* 1996b; Stone *et al.*, 1997).

The Stone Curlew's breeding distribution extends from south-western Europe east into the Orient (Cramp & Simmons, 1983; Hagemeyer & Blair, 1997). The UK is at the extreme north-west of the range. The UK population winters in south-west France, Iberia and north-west Africa (Batten *et al.*, 1990; Green *et al.*, in press).

In the 19th century, the Stone Curlew was widespread in southern and eastern England, its range extending north to the Cotswolds, east Midlands and Yorkshire (Batten *et al.*, 1990; Holloway, 1996). By the 1930s, however, its range had contracted away from northern counties and numbers had declined to 1,000-2,000 pairs (Glue & Morgan, 1974; Green, 1995). The population was estimated to be 200-400 pairs in the 1960s (Parslow, 1967) and 300-500 pairs by the first breeding atlas (Sharrock, 1976). The population declined further in the late 1970s, but since then has slowly risen (Figure 2.4.1; Spencer *et al.*, 1986a; Ogilvie *et al.*, 1996b). Green (1988) gave an estimate of 160 pairs in the late 1980s and Green (1995) a figure of 167-169 pairs in 1991. RSPB data for 1995 give a population of 166-180 pairs (Stone *et al.*, 1997). The species is now concentrated in two areas: Breckland on the Norfolk/Suffolk border and the chalkland of Wiltshire, Dorset, Hampshire and Berkshire. Small numbers also occur in north Norfolk, Cambridgeshire and on the Suffolk coast (Green, 1995). The Stone Curlew formerly bred on the South Downs in Sussex, but became extinct in that county in 1995 (Prater, 1986). The Breckland population increased from 73 confirmed pairs in 1986 to 88 in 1991 (Green, 1995).

Stone Curlews have also declined across much of Europe (Hagemeyer & Blair, 1997). In Bulgaria, for example, the population fell from, perhaps, several thousand breeding pairs at the turn of the century to between 150 and 200 in the early 1990s (Uhlir & Baumgart, 1995). In Germany, breeding has not occurred since 1987 (Leipe, 1990). The majority of the European Stone Curlew population still breeds in Iberia (Hagemeyer & Blair, 1997).

Factors affecting Populations

The Stone Curlew breeds in warm, dry areas with short or cropped vegetation that allows good all-round visibility (Nipkow, 1990; Hagemeyer & Blair, 1997). In the UK it is associated with sparsely vegetated heathlands (e.g. Breckland) and chalk downlands (e.g. Salisbury Plain in Wiltshire) (Glue & Morgan, 1974; Green, 1988) but elsewhere in Europe it also nests among sand-dunes, on stony deposits in rivers or on dry river beds and stony plains (Gola, 1993; Uhlir & Baumgart, 1995; Hagemeyer & Blair, 1997).

The primary cause of the Stone Curlew's decline in the UK and the rest of Europe is believed to be the deterioration and loss of breeding habitat. Much downland and grass heath has been lost through conversion to arable farmland and forestry and also to vegetation succession following reductions in grazing (Glue & Morgan, 1974; Green, 1988; Hagemeyer & Blair, 1997). In both Breckland and Wiltshire, for example, numbers of nesting Stone Curlews have been affected by increases in vegetation height (Green & Griffiths, 1994; Green & Taylor, 1995). Changes in vegetation height were believed to be attributable to lower densities of rabbits *Oryctolagus cuniculus* and increased rainfall.

Stone Curlews now breed in many agricultural habitats. In the UK, the species regularly uses spring-sown tilled farmland for nesting (Green, 1988) and in Europe it also breeds in vineyards, orchards and in young forestry plantations (Nipkow, 1994; Malvaud, 1995). The switch from spring- to autumn-sown cereals, however, has probably resulted in the loss of many potential nesting areas.

There is no evidence that there has been a decline in the breeding success of the Stone Curlew in the UK. Indeed, in recent years in Breckland, breeding productivity has increased markedly (Green, 1995). Although many nests are nowadays at risk from farming operations, especially hoeing, the monitoring of breeding pairs by the RSPB and EN has helped to decrease nest losses (Glue & Morgan, 1974; Green, 1988). This protection work, which has also reduced the impact of illegal egg-collecting, is now included in a RSPB/EN Stone Curlew Recovery Project. Given current survival rates of adults and juveniles, it is probable that the improvement in breeding productivity is responsible for the recent increase in breeding numbers (Green *et al.*, in press).

Present and Future Research

The UK population of Stone Curlews is currently monitored by the RBBP (JNCC) and through the research of the RSPB and EN. This research has indicated that over half of the UK population now nests on arable land and that nest protection may increase breeding productivity (Green, 1988, 1995). There is an additional need to determine whether the nesting success and survival of birds on arable sites differs from that on semi-natural sites (DoETR, EN). The species' habitat requirements are already well understood (e.g. Green, 1988; Green & Griffiths, 1994; Green & Taylor, 1995) and should be taken into account in plans for the Breckland and South Wessex Downs Environmentally Sensitive Areas (ESAs) (EN).

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2.5 Corncrake *Crex crex*

Status and Population Trends

IUCN Red List (Globally Threatened)

Current European Population 87,000-97,000 pairs

SPEC Category 1

SPEC Status Vulnerable (large decline)

EC Birds Directive Annex I

Berne Convention Appendix II

WCA 1981 Schedule 1

Wildlife (NI) Order 1985 Schedule 1

Biodiversity Steering Group Short List

Species of Conservation Concern Red List

Species of Conservation Importance Table 1

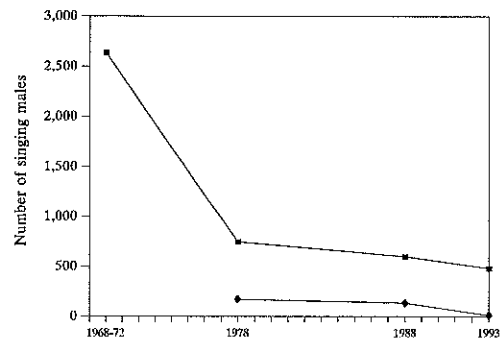


Figure 2.5.1 Maximum numbers of singing male Corncrakes in Britain (■) and Northern Ireland (●) from 1968-72 to 1993 (from Cadbury, 1980; Green, 1995; Sheppard & Green, 1995)

The Corncrake's breeding distribution extends from France and Ireland in the west, through central Europe to Siberia in the east (Cramp & Simmons, 1980; Hagemeyer & Blair, 1997). The small UK population occurs in Scotland and Northern Ireland (Gibbons *et al.*, 1993). The Corncrakes main wintering grounds are in south and central Africa (Stowe & Hudson, 1991a; Stowe & Becker, 1992).

In the 18th and early 19th centuries, the Corncrake was a widespread and, in places, abundant species in Britain and Ireland (Norris, 1945; Holloway, 1996). Over the last 150 years, however, the species has undergone a massive decline (Norris, 1945; Hudson *et al.*, 1990; Gibbons *et al.*, 1993; Stowe *et al.*, 1993). Cadbury (1980) estimated that there were 2,640 singing males in Britain at the time of the first breeding atlas in 1968-1972, but found that the population had fallen to 700-746 during a national BTO/SOC survey in 1978-79. The latter figure, however, is believed to be an underestimate (Hudson *et al.*, 1990; Green, 1995). RSPB surveys recorded a population of 551-596 singing males in 1988 and 480 in 1993 (Figure 2.5.1; Hudson *et al.*, 1990; Green, 1995). Corncrakes are now largely restricted to the Inner and Outer Hebrides in Scotland. Only 12-21 singing males were recorded outside of Scotland in 1978/79, five in 1988 and 10 in 1993 (Cadbury, 1980; Hudson *et al.*, 1990; Green, 1993). Of the 480 singing Corncrakes recorded in 1993, 36% were in the Inner Hebrides and 56% in the Outer Hebrides (Green, 1995).

There have been similar declines in Ireland. IWC/RSPB surveys recorded 1062 singing males in 1978, 903-930 in 1988 and just 174 in 1993 (O'Meara, 1979; Mayes & Stowe, 1989; Sheppard & Green, 1995). The population in Northern Ireland alone was 169 in 1978, 122-133 in 1988 and just nine in 1993 (Fig. 2.5.1).

The declines recorded in Britain and Ireland are matched by those seen in much of the rest of Europe (Hashmi, 1991; Stiefel, 1991; Szép, 1991). In France, for example, the population fell by 40% between 1983/84 and 1991/92 (Broyer, 1994). The population in the Netherlands has also declined greatly during this century, but may recently have begun to stabilize (van den Bergh, 1991). Large populations still remain in the Baltic states, Belarus and Russia and these probably represent 75-89% of the world population (Hagemeyer & Blair, 1997).

Factors affecting Populations

The main causes of the Corncrake's decline in Europe are believed to be habitat loss and the modernisation and mechanisation of farming practices. Corncrakes breed in damp areas of tall grass and herbs and are particularly associated with hay meadows (Flade, 1991). In North and South Uist, for example, Stowe and Hudson (1991b) found that Corncrakes used *Iris pseudocorus* beds in May and moved to hay meadows in June and July. The conversion of traditional hay meadows, partly through drainage, into short pasture dominated by *Juncus* or *Carex* has led to the Corncrake's demise in much of its former range (Green & Stowe, 1993; Stowe *et al.*, 1993). The mechanisation of hay cutting and the shift to earlier and more rapid harvesting has meant also that many more nests, young and adults are now destroyed (Norris, 1947; Cadbury, 1980; Stowe & Hudson, 1991b; Green, 1995). Much grass is now grown for silage, which is harvested earlier than hay and this too increases the impact of mowing on Corncrakes (Cadbury, 1980; Stowe & Hudson, 1991b; Green, 1995). The losses to mowing can be extreme - in a study in Poland of a population containing 700-900 singing males, it was believed that most, if not all broods were destroyed when hay was harvested in June (Schäffer, 1995).

On migration, the Corncrake is under threat from the incidental catching of birds in nets set for Quail *Coturnix coturnix*, notably in Egypt (Stowe & Hudson, 1991; Stowe & Becker, 1992). Corncrakes are not known to be seriously threatened on their southern African wintering grounds, however.

Present and Future Research

The UK Corncrake population should preferably be surveyed every three years, to continue the monitoring of the species' status (EHS, SNH). Key sites should be monitored annually. There is a need to study the economic, technical and agronomic aspects of modifying grassland management in key areas for the benefit of the species (DANI, EHS, SNH). Such measures may include leaving early-growing rough cover around old buildings and along field margins, leaving strips of unmown hay at field edges and maintaining agricultural diversity (Williams *et al.*, 1991). RSPB research has also highlighted the need to change mowing techniques. The traditional pattern of mowing traps chicks and adults in the centres of fields from where they cannot escape. Mowing from the inside of fields outwards may reduce casualties by pushing birds to the field margins (Stowe & Hudson, 1991b). Schäffer & Weisser (1996) alternatively suggest that areas cut prior to the main mowing date may provide refuges for nestlings when the rest of the hay is harvested. The effects of different mowing techniques and dates on breeding success and adult mortality, however, do need further investigation (EHS, SNH). Mortality caused by cats, minks *Mustela vison* and ferrets *M. furo* and possible control measures should also be assessed. A review of factors affecting corncrakes on migration and on their wintering grounds is also needed (EHS, JNCC, SNH).

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2.6 Scottish Crossbill *Loxia scotica*

Status and Population Trends

IUCN Red List (Data Deficient)

Current European Population 300-1,250 pairs

SPEC Category 1

SPEC Status Insufficiently known

EC Birds Directive Annex I

Berne Convention Appendix II

WCA 1981 Schedule 1

Biodiversity Steering Group Short List

Species of Conservation Concern Red List

Species of Conservation Importance Table 1

The Scottish Crossbill is the UK's only endemic bird species. Its specific status remains unclear, however, due to taxonomic confusion and the difficulty in distinguishing between this species and Common and Parrot Crossbills *Loxia curvirostra* and *L. pytyopsittacus* (Knox, 1976, 1990a; Proctor & Fairhurst, 1993; Summers *et al.*, 1996). Recent studies in north-east Scotland, though, have shown that Scottish and Common Crossbills can breed sympatrically without interbreeding (Knox, 1990b, 1990c).

Nethersole-Thompson (1975) estimated that there were 1,500 adult Scottish Crossbills in the early 1970s. Thom (1986) gave an estimate of between 300 and 400 pairs. At present, the population is estimated at between 300 and 1,250 pairs (Stone *et al.*, 1997). Accurate censusing is difficult, however, due to the problems of separating Scottish and Common Crossbills in the field (Gibbons *et al.*, 1993). Scottish Crossbills are confined to forests in the highlands of north Scotland, from Argyll, Perth and probably Kincardine north to Sutherland (Thom, 1986; Gibbons *et al.*, 1993).

Factors affecting Populations

Crossbills feed almost entirely on the seeds of conifers and their distributions are strongly correlated with those of the main tree species on which they feed. The Scottish Crossbill has a bill adapted to feed on the seeds of Scots pine *Pinus sylvestris* (the only conifer native to Britain apart from juniper *Juniperus communis*) and is thus largely restricted to forests of this species (Buckland *et al.*, 1990; Knox, 1990b; Gibbons *et al.*, 1993). Common Crossbills more commonly feed on spruce *Picea* and larch *Larix* seeds and in the UK are thus associated with plantations of these species (Buckland *et al.*, 1990; Knox, 1990b; Gibbons *et al.*, 1993). Both crossbills, however, will eat the seeds of other tree species when supplies of their preferred foods are low (Cramp & Perrins, 1994a).

The Caledonian pinewoods once covered 1.5 million hectares of the Scottish highlands (McVean & Ratcliffe, 1962), but over the last 4,000 years their extent has been much reduced by man's activities (Knox, 1990d). This century continued clear-felling and under-planting with exotic conifers reduced the size of the woods to 12,000 ha by 1987 (Bain, 1987). In some areas, grazing from the increasing numbers of Red Deer *Cervus elaphus* has prevented the natural regeneration of pinewoods since about 1820 (Watson, 1983). It is probable that the loss of these native woods has resulted in a contraction in the range and population of not just the Scottish Crossbill, but also the Capercaillie *Tetrao urogallus* and Crested Tit *Parus cristatus*. Although Scottish Crossbills do breed in mature plantations of Scots pine (Buckland *et al.*, 1990), densities may be lower than in native pinewoods (Newton & Moss, 1977). The Scottish Crossbill's long-term survival is thus largely dependent on the conservation and regeneration of these woods.

In the short-term, crossbill numbers and their distributions are largely determined by conifer seed crops. If crops fail in one area, crossbills may move long distances in search of alternative food (Newton, 1972). The failure of spruce crops in northern Europe, for example, has led to invasions of Common Crossbills into the UK (Marquiss & Rae, 1994; Summers *et al.*, 1996). Some of these birds are believed to have settled and bred (Nethersole-Thompson, 1975). Scottish (and Parrot) Crossbills move less in comparison, as a result of the more reliable seeding of pines (Catley & Hursthouse, 1985; Gibbons *et al.*, 1993) and their populations are thus more stable. Although Scottish Crossbills may move between years, they are reliant on old trees, which are becoming scarcer with the loss of native woods (Knox, 1990b, 1990d).

Present and Future Research

There is a clear need to clarify the taxonomic status of Scottish Crossbills (SNH) and the RSPB is currently funding DNA analysis to this end. In addition, there is a need for improved methods of distinguishing Scottish and Common Crossbills in the field (SNH) in order that the precise population and distribution of Scottish Crossbills can be determined and that a regular monitoring programme can be set up (SNH). Research on habitat and food requirements, breeding performance and the effects of predation also needs to be undertaken (FA, SNH). The FA completed an inventory of Caledonian pinewoods in 1994, registering the location of native woods, their extent and possible regeneration zones. FA initiatives promote the management and regeneration of native pinewoods and the FA and SNH are also due to produce a handbook on pinewood management. EC LIFE (Nature) programme funding has been received to assess the native pinewood resource in Scotland and to evaluate the impact of deer grazing on regeneration.

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2.7 Grey Partridge *Perdix perdix*

Status and Population Trends

Current European Population 1,700,000-3,000,000 pairs

SPEC Category 3

SPEC Status Vulnerable (large decline)

EC Birds Directive Annexes II/I & III/I

Berne Convention Appendix III

Game Acts

Game Preservation (Partridge and Hen Pheasant) Order (NI) 1967

Biodiversity Steering Group Short List

Species of Conservation Concern Red List

Species of Conservation Importance Table 3

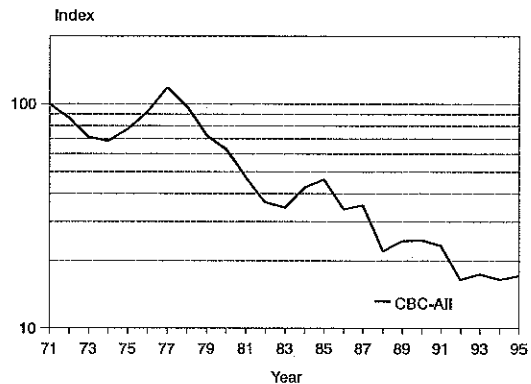


Figure 2.7.1 Population trend from the Common Birds Census

The Grey Partridge's distribution extends across central Europe and Asia, from Ireland in the west, to eastern Mongolia (Cramp & Simmons, 1980; Hagemeyer & Blair, 1997). It has also been successfully introduced to the northern USA and Canada. Although the UK population is sedentary, the species is a partial migrant in eastern Europe and Asia due to severe winters (Potts, 1986; Nikiforov, 1992).

In the UK, the Grey Partridge is a popular gamebird and was formerly one of the most common farmland birds (Gibbons *et al.*, 1993). Since the early 1950s, however, the UK population has been in severe decline (Potts, 1970; Potts & Aebischer, 1995). CBC data show that the population fell by 86% between 1971 and 1995 (Crick *et al.*, 1997; Fig. 2.7.1). The rate of decline may now be slowing, as BBS data suggest a slight increase between 1994 and 1996 (Gregory *et al.*, 1997). Breeding atlas data show that the Grey Partridge's range in Britain contracted by 19% between 1968-72 and 1988-91, with gaps appearing in south-west Scotland, Wales, south-west England and parts of the Fens and the Weald (Gibbons *et al.*, 1993). At the time of the second atlas, there were an estimated 140,000 territories in Britain.

The Grey Partridge is in decline throughout Europe (Tucker & Heath, 1994; Hagemeyer & Blair, 1997). In Ireland, its range contracted by 86% between 1968-72 and 1988-91 (Gibbons *et al.*, 1993) and a 1991 survey revealed that the species was concentrated in just two small areas in Co. Offaly (Kavanagh, 1992). The Grey Partridge is now extinct in Norway and is on the verge of extinction in Switzerland (Potts, 1986; Schifferli, 1993). Declines of 80% or more have been recorded since 1940 in many other European countries, including Germany, Italy, Austria and Hungary (Potts, 1986; Birkan & Jacob, 1988; Dwenger, 1991).

Factors affecting Populations

In western Europe, the Grey Partridge is associated with open, low-intensity mixed farmland, where hedges and wide field margins provide nesting cover (Potts, 1986). Adults feed primarily upon seeds, particularly on those of the hemp nettle *Galeopsis tetrahit*, black bindweed *Polygonum convolvulus* and chickweed *Stellaria media*, whilst chicks are largely insectivorous (Green, 1984; Rands, 1985, 1986; Potts, 1986). Research by the GCT has revealed that 40% of the Grey Partridge's decline in the UK has been associated with a decrease in chick survival rates (Potts, 1980). Between 1952 and 1962 chick survival fell from 45% to 30% and although the rate has not declined further since, this level has been insufficient to maintain stable populations (Potts, 1986). The primary environmental cause of the decline in chick survival is believed to be the intensification of agriculture and in particular, the increased use of herbicides which, through their control of weeds, have reduced the amount of insect food available to chicks (Potts, 1970, 1980, 1986, 1997; Rands, 1985, 1986; Campbell *et al.*, 1997). Potts and Aebischer (1995) reported that before the introduction of herbicides, chick survival rates averaged 49% and that after their use became commonplace, they averaged 32% (see also Campbell *et al.*, 1997). Breeding success has also been reduced through the increased use of insecticides, the loss of nesting cover through the removal of hedges and grass field margins, and increasing predation rates following the fall in keeping (Potts, 1980, 1986, 1997; Eislöffel, 1996; Tapper *et al.*, 1996).

In common with other lowland farmland species, Grey Partridges have probably also been hard hit by the loss of winter stubbles and their associated food supplies, following the switch from spring- to autumn-

sown cereals (Wilson *et al.*, 1996). In their Sussex study area, however, Potts and Aebischer (1995) found that annual over-winter survival rates rose between 1968 and 1993 (although there was evidence that these figures were distorted by immigration).

Present and Future Research

The Grey Partridge's status is currently monitored by the BBS (BTO, JNCC, RSPB). The GCT's National Game Census (which has run from 1933 to the present day) provides information on chick survival rates and shooting bag returns (CCW, EHS, EN, JNCC, SNH). GCT research has shown that chick survival rates may be restored to levels at which populations may increase again by the use of unsprayed 'conservation headlands' or by the traditional farming system of undersowing cereal crops with ley pasture (Gibbons *et al.*, 1993). The latter system particularly benefits sawflies, a favourite food of partridge chicks (Aebischer, 1990; Rands, 1985, 1986; Potts, 1986). The continuance of such work will help to provide further information on the ecological requirements of the Grey Partridge and help in the development of management advice (CCW, DANI, EHS, EN, JNCC, MAFF, SNH, SOAEFD, WOAD). Further research should focus on the effects of agrochemicals on food supplies and chick survival (CCW, EHS, EN, JNCC, SNH).

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2.8 Capercaillie *Tetrao urogallus*

Status and Population Trends

Current European Population 210,000-300,000 pairs

SPEC None

EC Birds Directive Annexes I, II/II & III/I

Berne Convention Appendix III

WCA 1981 Schedules 2, 3 & 9

Biodiversity Steering Group Short List

Species of Conservation Concern Red List

Species of Conservation Importance Table 2

The Capercaillie's distribution extends from Scotland, across Fennoscandia east to mid-Siberia (Cramp & Simmons, 1980; Hagemeyer & Blair, 1997). The species also occurs in forests in the Pyrenees, the Cantabrian mountains of Spain, the Alps and south-east Europe. The Scottish population forms only a small fraction of the world population.

The Capercaillie was formerly widespread across northern Britain and Ireland, but became extinct in the 1780s (Pennie 1950-51; Holloway 1996). It was initially reintroduced in the 1830s using birds trapped in Sweden (Pennie, 1950-51; Hagemeyer & Blair, 1997). Capercaillies were most widely distributed in the early 20th century, when they were found from Stirling in the south to Sutherland in the north and from Argyllshire in the west to Aberdeenshire in the east (Pennie, 1950-51; Buckland *et al.*, 1990; Gibbons *et al.*, 1993). More recently the species has shown a marked contraction in its range. The first breeding atlas (1968-72) found that Capercaillies were present on 182 10-km squares (Sharrock, 1976), but this had fallen to 66 at the time of the second atlas (Gibbons *et al.*, 1993). Shooting bag indices have also fallen dramatically since 1970 (Moss & Weir, 1987; Gibbons *et al.*, 1993). Surveys between 1992 and 1994, however, have shown that the species is more widespread than the second atlas suggests and gave a winter population estimate of 2,200 adults (Catt *et al.*, 1994a).

The Capercaillie is decreasing throughout its range (Klaus & Bergmann, 1994; Hagemeyer & Blair, 1997). There have been large declines in Norway (Wegge, 1980) where the annual shooting bag fell from 40,000 birds in 1960 to 13,000 in 1981 (Batten *et al.*, 1990). In Finland 47,000-104,000 birds were shot annually between 1959 and 1967, but only 14,500 between 1969 and 1976. A study in Switzerland recorded that densities of adults had fallen from 10.3/km² in 1980 to 7.7.3 between 1990 and 1992 (Gilliéron, 1993). The hunting of Capercaillies is now banned in both France and Germany.

Factors Affecting Populations

The probable causes of the Capercaillie's decline include habitat loss and degradation, increased predation, over-hunting and adverse climatic conditions. In Scotland, the Capercaillie is associated with Scots Pine forests, though in Europe it also occurs in forests of white fir *Abies alba*, Norway spruce *Picea abies*, oak *Quercus* and other hardwoods (Hagemeyer & Blair, 1997). The needles and shoots of Scots Pine are an important food source, particularly in winter, but in spring both adults and chicks feed on bilberry *Vaccinium myrtillus* and the invertebrates it supports (Storch, 1993).

The huge reduction in the size of the native Caledonian pinewoods (McVean & Ratcliffe, 1962) has undoubtedly resulted in a sharp contraction in the range and population of the Capercaillie in Scotland. The species notably declined following the large-scale felling of timber during the first world war (Pennie, 1950-51; Buckland *et al.*, 1990). A quarter of the remaining forests were lost between 1957 and 1987 (Bain, 1987). However, the Capercaillie has successfully colonized many 20th century plantations of Scots pine and recently also those of Sitka spruce *P. sitchensis* (Picozzi *et al.*, 1992).

Modern forestry practices affect both breeding success and survival. Capercaillies prefer mature, open forests which encourage the growth of bilberry and other field-layer plants and provide open areas for displaying (leks) (Rolstad & Wegge, 1987a, 1987b; Picozzi *et al.*, 1992). Bilberry provides not only food, but also cover from predators for chicks (Moss & Weir, 1987). High densities are thus often found in natural and semi-natural pinewoods (Picozzi *et al.*, 1992; Gibbons *et al.*, 1993). Modern plantations, in contrast, are often enclosed and provide little space for leks and little ground vegetation. Breeding success in plantations is also reduced by increased numbers of foxes *Vulpes vulpes* and Carrion Crows *Corvus corone*. Capercaillie (and other grouse) also suffer high mortality from collisions with overhead power

lines (Bevanger, 1995) and fences (Catt *et al.*, 1994b; Baines & Summers, 1997). The latter, perversely, are often put up to exclude Red Deer and encourage the natural regeneration of the pinewoods.

The breeding success of Capercaillie is reduced by wet June weather and as a consequence their distribution is limited to areas of low rainfall (Moss & Oswald, 1985; Moss, 1986). It is probable that climatic change could be leading to declines not just in the UK, but across Europe (Hagemeijer & Blair, 1997).

Present and Future Research

The surveys between 1992 and 1994 (SNH) were the first to accurately assess the size and distribution of the Capercaillie population in Scotland. It is important to continue this monitoring and to assess the numbers and breeding success of Capercaillie in relation to methods of habitat management and predator control (SNH). Research into habitat preferences is ongoing, under the inter-agency Capercaillie Working Group. Recent RSPB studies have investigated the impact of collisions with deer fences on overall mortality (Catt *et al.*, 1994b; Baines & Summers, 1997), but further research is required into the effectiveness of marking fences in reducing collisions (FC, SNH).

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2.9 Song Thrush *Turdus philomelos*

Status and Population Trends

Current European Population 14,100,000-18,500,000 pairs

SPEC Category 4

SPEC Status Secure

EC Birds Directive

Berne Convention Appendix III

WCA 1981

Wildlife (NI) Order 1985

Biodiversity Steering Group Short List

Species of Conservation Concern Red List

Species of Conservation Importance Table 3

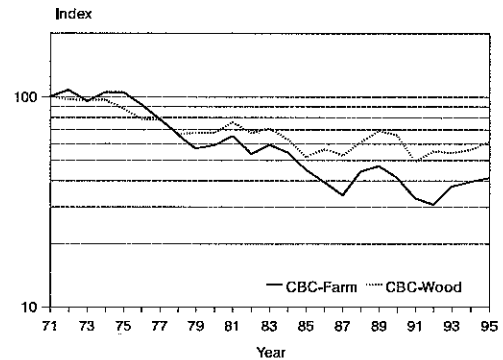


Figure 2.9.1 Population trends from the Common Birds Census

The Song Thrush's breeding distribution extends across central and northern Europe eastwards to Lake Baikal in Siberia (Cramp, 1988; Hagemeijer & Blair, 1997). As with other northern European populations, that in the UK is partially migratory, some birds move to France and Spain in winter (Gibbons *et al.*, 1993). The subspecies *clarkei* breeds in Britain, Ireland, north-west France and the Low Countries, whilst the subspecies *hebridensis* occurs only on Skye and the Hebrides (Hagemeijer & Blair, 1997).

The Song Thrush is a widespread and still common species in the UK and is only absent from Shetland and a few areas of northern Scotland (Gibbons *et al.*, 1993). In the first decades of this century it was more abundant than the Blackbird *Turdus merula*, despite a number of short-term decreases following severe winters (Ginn, 1969). No clear trend in population size was apparent, however, until after the 1960s when the species began a sharp decline. CBC data show that the population fell by 56% between 1971 and 1995, by 70% on farmland alone and by 45% in woodland (Crick *et al.*, 1997; Fig. 2.9.1). Likewise, CES data indicate a 24% decline in catches of adults between 1983 and 1995 and a 46% decline in catches of juveniles. BBS data suggest that this decline may now be slowing (Gregory *et al.*, 1997). Atlas data show that the Song Thrush's range in Britain had contracted by 2% between 1968-72 and 1988-91, with gaps appearing in northern Scotland (Gibbons *et al.*, 1993). At the time of the second atlas, there were an estimated 990,000 territories in Britain and 390,000 in Ireland.

Smaller declines have also been recorded recently in Ireland, the Netherlands and Andorra, whilst the population in France has been fluctuating (Tucker & Heath, 1994). Declines in central Europe have been associated with a withdrawal from suburban habitats, first colonised in the first half of the century (Tomiałojć, 1992). Elsewhere populations have been mostly stable, although the species has expanded its range in northern Scandinavia (Melde & Melde, 1991). The UK population represents between 5% and 10% of the European population (Tucker & Heath, 1994; Hagemeijer & Blair, 1997).

Factors affecting Populations

Thomson *et al.* (1997) used national ringing recoveries to investigate the annual survival rates of adult and first-year Song Thrushes between 1962 and 1993. Adult survival rates initially increased during this period and then fell. First-year survival rates, in contrast, showed a long-term decrease and alone were able to account for the population decrease. Both first-year and adult survival rates were lower in years with cold winters and the former in years with dry winters (see also Baillie, 1990 and Greenwood & Baillie, 1991). However, the effects of winter weather were not sufficient to explain the decline by themselves.

The decline in first-year survival may have been caused by a number of other factors. The increase in winter cereals and the loss of spring tillage may have reduced the available farmland feeding habitat for Song Thrushes during the early part of the breeding season (O'Connor & Shrubbs, 1986). The increased use of molluscicides to control slugs on farmland is likely to have reduced the availability of earthworms and snails, an important food source for Song Thrushes (Davies & Snow, 1965). These factors may help to explain why the decline seen on farmland is greater than that in woodland. It is also possible that the recent increase in numbers of Sparrowhawks *Accipiter nisus* (Newton & Haas, 1984) may have depressed the annual survival rates of Song Thrushes (Thomson *et al.*, 1997). Hunting may affect the proportion of the UK population which migrates to southern Europe. In the UK, however, an index of hunting pressure for the species has decreased since 1980 (McCulloch *et al.*, 1992).

The population decline is not associated with reduced breeding success. NRS data show that brood sizes increased between 1971 and 1995 and that daily nest failure rates decreased (Crick *et al.*, 1997; Fig. 2.9.2).

Present and Future Research

The Song Thrush's status is currently monitored by the BBS and CES scheme and breeding productivity assessed by the CES scheme and the NRS (BTO, JNCC, RSPB). There is a need to investigate the effects of agricultural changes and the use of molluscicides on population changes (CCW, EHS, EN, JNCC, SNH). Similarly, research is required on the effect of woodland and hedgerow structure on breeding success (FA). Ongoing analyses of national ringing recovery data (Thomson *et al.*, in press) aim to identify the role of long-term changes in post-fledging mortality rates on population changes.

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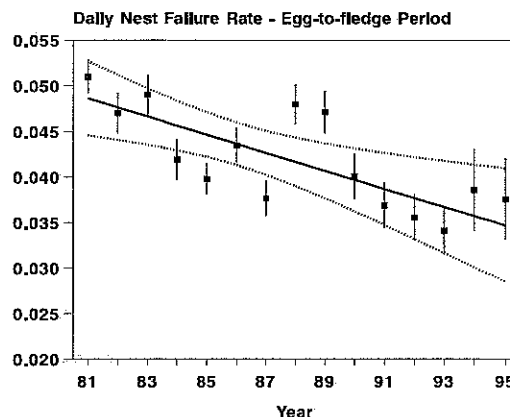


Figure 2.9.2 Long-term trend from the Nest Record Scheme

Species on the Biodiversity Steering Group Middle List

2.10 Marsh Warbler *Acrocephalus palustris*

Status and Population Trends

Current European Population 1,500,000-1,900,000 pairs

SPEC Category 4

SPEC Status Secure

EC Birds Directive

Berne Convention Appendix II

WCA 1981 Schedule I

Biodiversity Steering Group Middle List

Species of Conservation Concern Red List

Species of Conservation Importance Table 2

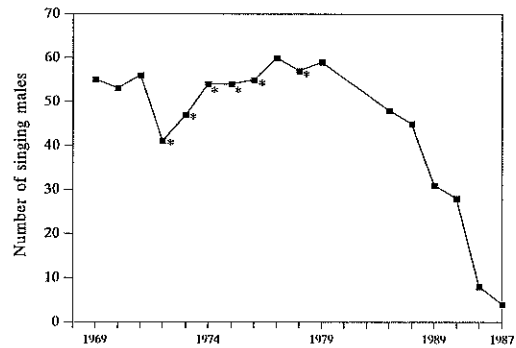


Figure 2.10.1 Number of territory-holding singing male Marsh Warblers in Worcestershire between 1969 and 1987 (from Kelsey *et al.*, 1989). * - less than 50 % of potential sites surveyed.

The Marsh Warbler's breeding range covers much of central Europe, reaching the Urals in the east and Finland in the north (Cramp, 1992; Hagemeijer & Blair, 1997). The UK is at the north-western limit of its distribution (Cramp, 1992; Kelsey *et al.* 1989; Hagemeijer & Blair, 1997).

The Marsh Warbler has never been common as a breeding bird in the UK, but has been proven to breed in 20 southern counties (Kelsey *et al.*, 1989). The population was formerly concentrated in three counties in the West Midlands (Worcestershire, Somerset and Gloucestershire) where there were at least 95 pairs in the 1960s (Gibbons *et al.* 1993). The population in this area has declined sharply since then, however, and is extremely vulnerable to extinction (Fig. 2.10.1; Kelsey *et al.* 1989). Breeding has not taken place in Somerset since 1961 and in Gloucestershire since 1984 (Kelsey *et al.*, 1989). The population in the UK at the time of the second breeding atlas (1988-1991) was no more than 12 pairs, the majority of which were in Kent (Gibbons *et al.*, 1993). This population has only recently become established, the first confirmed breeding record in the county coming in 1980 (Taylor, 1982). Breeding took place in Scotland for the first time in 1993 (Meek & Adam, 1997).

Marsh Warblers are, in contrast, common in mainland Europe. The densest populations, in the Low Countries, central and eastern Europe, are reported to have been stable between 1970 and 1990 (Hagemeijer & Blair, 1997). The species has also markedly extended its range since 1900, particularly into southern Scandinavia. Numbers in Sweden, for example, have increased from c.300 breeding pairs in the 1940s (Holmbrink, 1982) to 15,000 pairs in the late 1970s (Risberg, 1990; Hagemeijer & Blair, 1997). Only the Czech Republic and Moldova have reported slight declines (Hagemeijer & Blair, 1997). The Marsh Warbler winters in south-east Africa, from Zambia to South Africa (Kelsey, 1989a; Hagemeijer & Blair, 1997).

Factors affecting Populations

Marsh Warblers breed in damp areas with luxuriant, dense vegetation, often dominated by willowherb *Epilobium*, meadowsweet *Filipendula ulmaria*, umbellifers or nettles *Urtica dioica* (van der Hut, 1986; Kelsey *et al.*, 1989). It is believed that the loss or deterioration of such habitat may have led to a loss of about 30 pairs in the West Midlands between the 1960s and 1970s (Kelsey *et al.*, 1989). However, other sites, which have not changed, have also been abandoned.

In Germany, local population declines have been observed in areas of agricultural intensification (Kelsey *et al.*, 1989) and also in areas without apparent habitat change (Mildenberger, 1984).

The decline in the UK population is thought to be largely due to its isolation and uncompensated emigration losses (Kelsey *et al.*, 1989). Earlier losses associated with habitat or even climatic change would have made the population more vulnerable. Although breeding productivity in UK Marsh Warblers is lower than that seen on the continent (Kelsey, 1989b; Schulze-Hagen *et al.*, 1996), there has been no decline in breeding performance in recent years (Kelsey, 1989b).

Present and Future Research

The UK Marsh Warbler population is currently monitored by the RBBP (JNCC). Closer monitoring is needed, however, to determine breeding success and the causes of nest losses. Further research is also

needed on the habitat requirements of the species in the UK (EN), which are currently best known from its former stronghold in Worcestershire (Kelsey 1989b; Kelsey *et al.*, 1989). Research on the species' ecology should be supplemented by information from other areas in Europe where the species is more common.

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2.11 Nightjar *Caprimulgus europaeus*

Status and Population Trends

Current European Population 220,000-260,000 pairs

SPEC Category 2

SPEC Status Declining (moderate decline)

EC Birds Directive Annex I

Berne Convention Appendix II

WCA 1981

Wildlife (NI) Order 1985 Schedule 1

Biodiversity Steering Group Middle List

Species of Conservation Concern Red List

Species of Conservation Importance Table 2

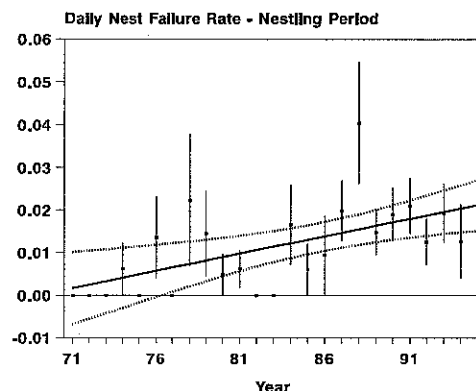


Figure 2.11.1 Long-term trend from the Nest Record Scheme

The Nightjar's breeding distribution extends from Ireland and Iberia, through central Europe and Asia, as far east as China (Cramp, 1985; Hagemeyer & Blair, 1997). The nominate *europaeus* breeds as far north as Scotland and southern Scandinavia, but is replaced in southern Europe by the subspecies *meridionalis*. The Nightjar winters in sub-Saharan Africa.

The Nightjar was formerly widespread across England, Wales and southern Scotland, though has always been more locally distributed in Ireland (Holloway, 1996). Since the turn of the century, however, the species has declined dramatically. The Nightjar's distribution was first recorded by a survey in 1952 (Norris, 1960) and a further BTO survey in 1957/58 found that the Nightjar had been in decline for at least 10 years, although that it was still common in southern England (Stafford, 1962). A BTO census in 1981 estimated that Britain held 2,100 churring males, the majority in southern England and East Anglia (Gribble, 1983). There was only one sighting in Ireland. A repeat BTO/RSPB survey in 1992, revealed that the British population had partially recovered to an estimated 3,400 males (Morris *et al.*, 1994). The greatest increases were recorded in Wales and south-west England. Breeding atlas data show that the Nightjar occupied 51% fewer 10-km squares in 1988-91 than in 1968-72, with gaps appearing throughout its distribution (Gibbons *et al.*, 1993). In Ireland, the species was only recorded in 11 10-km squares, the majority in Eire.

Declines have also been recorded through much of the rest of northern and western Europe. In The Netherlands, for example, the Nightjar's population has fallen by 80-95% since the 1950s and only 450-650 breeding pairs remained in 1992 (Maréchal & Taapken, 1989; van Dijk *et al.*, 1994). Spain holds the largest remaining population in Europe (c. 95,000 pairs: Hagemeyer & Blair, 1997), although here too the species is in decline (Tucker & Heath, 1994).

Factors affecting Populations

The primary cause of the Nightjar's decline in the UK and the rest of Europe is believed to be the deterioration and loss of habitat (Gribble, 1983; Morris *et al.*, 1994; Hagemeyer & Blair, 1997). The species nests on sparsely vegetated or bare ground, and is found in lowland heathlands, woodland edges and clearings (Cramp, 1985; Sierro, 1991; Morris *et al.*, 1994; Hagemeyer & Blair, 1997). It may also occur at higher altitudes, for example on the North York Moors (Leslie, 1985), though usually only if soils are well-drained and there is adequate cover. Its decline in the UK has been particularly associated with the loss of heathlands to housing, road development, forestry, agriculture and mineral extraction (Cadbury, 1989; Webb, 1990). Heathlands have also been lost to vegetational succession, following reductions in livestock grazing and the decline of the rabbit population in the 1950s, following a myxomatosis outbreak (Marrs *et al.*, 1986). In the 1992 survey, scrub invasion was recorded on 48% of 522 heathland sites occupied by Nightjars (Morris *et al.*, 1994). The reduction in coppicing has also led to a lack of open clearings in broad-leaved woodlands (Maréchal & Taapken, 1989; Morris *et al.*, 1994). Although these habitat changes have resulted in a loss of nesting sites, the foraging behaviour of Nightjars may have been less affected as birds may feed over a wide range of habitats (Alexander & Cresswell, 1990). Food supplies, however, may have been reduced by the increased use of insecticides (Tucker & Heath, 1994; Hagemeyer & Blair, 1997). The Nightjar's recent partial recovery in the UK has been largely due to the colonization of recently clearfelled and restocked areas of forestry (Ravenscroft, 1989; Bowden & Green, 1991; Morris *et al.*, 1994).

It is possible that climatic deterioration may also be a factor in the species' decline. Nightjars arrive later on their breeding grounds in years with cold springs (Kemp, 1983) and this may result in fewer second broods being laid. Berry and Bibby (1981), for example, found that second clutches were rare if first clutches were laid later than 4 June. This study found that only 20% of pairs laid second clutches in the late 1970s at their study site in Suffolk, in comparison to 50% in the 1920s (Lack, 1930). Late springs may also cause food shortages, which prevent adults from breeding or lead to the starvation of chicks. Although the NRS has found that there has been a recent increase in nest-failure rates during the nestling stage, this has continued during the species' recovery and has not been associated with a deterioration in climate (Crick *et al.*, 1997; Fig. 2.11.1). As Morris *et al.* (1994) also point out, the Nightjar's decline began prior to a period of climatic amelioration and if climate were a factor, the Nightjar should have declined most from the northern fringes of its range.

Present and Future Research

The next national survey is due to be carried out in 2002 (JNCC). Breeding productivity is assessed by the NRS (BTO, JNCC). The condition of heathland in the Nightjar's former range needs to be evaluated and possible remedial management identified (EN). RSPB research has shown that Nightjar numbers can be increased by creating glades in dense woodland, resculpting woodland margins to increase the length and width of woodland edge habitat, leaving woodland shelterbelts and providing bare ground for nesting (Burgess *et al.*, 1990). Further research is required to relate food availability to breeding success and productivity (EN).

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2.12 Linnet *Carduelis cannabina*

Status and Population Trends

Current European Population 7,100,000-9,100,000 pairs

SPEC Category 4

SPEC Status Secure

EC Birds Directive

Berne Convention Appendix II

WCA 1981

Wildlife (NI) Order 1985

Biodiversity Steering Group Middle List

Species of Conservation Concern Red List

Species of Conservation Importance Table 3

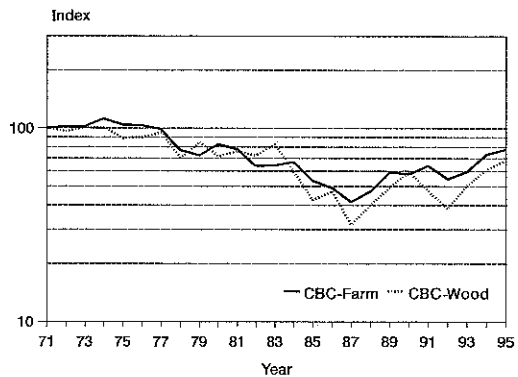


Figure 2.12.1 Population trends from the Common Birds Census

The Linnet's breeding distribution extends across the Palearctic from western Europe to central Siberia, as far north as the southern Scandinavia and south to north-west Africa and the Near East (Cramp & Perrins, 1994a; Hagemeijer & Blair, 1997). UK and northern European populations are partially migratory, some birds moving to France and Iberia in winter. The nominate subspecies *cannabina* breeds in Britain, Ireland and central and northern Europe, whilst the subspecies *autochithona* is endemic to Scotland (Hagemeijer & Blair, 1997).

The Linnet is a widespread and common species in the UK, occurring across a variety of scrub and farmland habitats. At the end of the 19th century, it was reportedly in decline, due to the loss of arable land and widespread trapping for the cagebird industry (Marchant *et al.*, 1990; Holloway, 1996). Populations stabilised after the turn of the century following the decline of trapping and an agricultural recession, when arable weeds (an important food source) increased through the widespread neglect of farmland. Between the mid-1970s and the mid-1980s, the species again declined sharply. CBC data show that the population fell by 46% between 1971 and 1995, by 49% on farmland alone and by 58% in woodland (Crick *et al.*, 1997; Fig. 2.12.1). Although these data indicate a recent population increase, CES data suggest that the decline is continuing, at least in scrubland habitats. There was a 85% decline in the numbers of adults caught on CES sites between 1983 and 1995 and a 91% decline in the numbers of juveniles (Crick *et al.*, 1997). BBS data, however, suggest little recent decline (Gregory *et al.*, 1997). Breeding atlas data show that the Linnet's range in Britain contracted by 5% between 1968-72 and 1988-91 and that in Ireland by 18% (Gibbons *et al.*, 1993). The species is now absent from large areas of north-west Scotland and parts of the Southern Uplands. At the time of the second atlas, there were an estimated 520,000 territories in Britain and 130,000 in Ireland.

Declines have also been recently recorded across much of the rest of central and northern Europe. Populations in the Netherlands and Finland, for example, have declined by over 50% (Hagemeijer & Blair, 1997). Mediterranean populations, in contrast, are stable or even increasing.

Factors affecting Populations

Although decreases in the Linnet's northern European populations may be associated with the effects of severe winters (e.g. Hildén, 1989), the declines elsewhere have been primarily attributed to agricultural changes. The Linnet is highly dependent upon weed seeds, both in winter and summer and as a food source for nestlings (Newton, 1967, 1972; Cramp & Perrins, 1994a). Consequently, the species has been affected by the increased use of herbicides and fertilisers and more intensive grassland management (O'Connor & Shrubbs, 1986; Gibbons *et al.*, 1993; Campbell *et al.*, 1997). In winter, Linnets are particularly reliant upon stubbles (Wilson *et al.*, 1996) and their loss following the switch from spring- to autumn-sown cereals is also likely to have increased mortality rates. The species' recent fortunes, however, are reflected by trends in daily nest failure rates, thus suggesting a link with breeding success (Crick *et al.*, 1997; Fig. 2.12.2).

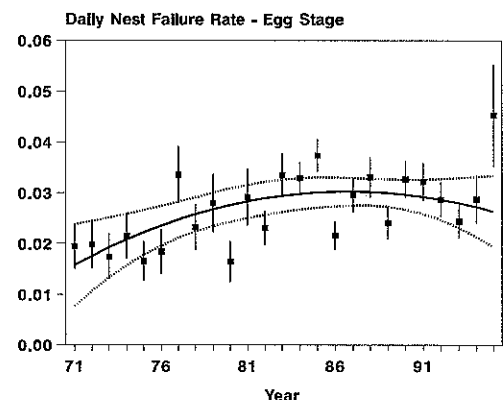


Figure 2.12.2 Long-term trend from the Nest Record Scheme

Another factor in the decline of the Linnet has probably been the loss and deterioration of habitat. Nest-sites have been lost through the removal of hedgerows, scrub and farmland woodland, and weeds depleted by increased grazing, for example, on heathland (Kurlavičius, 1987; Büsche, 1991). Linnets, however, have expanded into other new habitats, including suburban areas (Tast, 1968), and have benefited from the increase in oilseed rape, which constitutes both a food source and a potential nesting habitat (O'Connor & Shrubbs, 1986; Marchant *et al.*, 1990; Burton *et al.*, 1996). The species' recent partial recovery may be linked to the use of set-aside (Crick *et al.* 1997).

Present and Future Research

The Linnet's status is currently monitored by the BBS and CES scheme and breeding productivity assessed by the CES scheme and the NRS (BTO, JNCC, RSPB). A survey for monitoring winter populations should also be considered (CCW, EHS, EN, JNCC, SNH). An ongoing study in Oxfordshire has indicated that nestling diet has changed since the 1960s, following the decline of charlock *Sinapis arvensis* through herbicide control (see Campbell *et al.*, 1997). Further research is required to determine how diet varies between habitats and how food availability affects breeding success. There is also a need to improve understanding of the Linnet's winter habitat requirements (CCW, EHS, EN, JNCC, SNH) and to improve survival monitoring through increased ringing (BTO, JNCC).

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2.13 Cirl Bunting *Emberiza cirlus*

Status and Population Trends

Current European Population 1,300,000-2,300,000 pairs

SPEC Category 4

SPEC Status Secure

EC Birds Directive

Berne Convention Appendix II

WCA 1981 Schedule 1

Biodiversity Steering Group Middle List

Species of Conservation Concern Red List

Species of Conservation Importance Table 2

The Cirl Bunting is primarily a Mediterranean species, its distribution extending from Iberia and north Africa in the west to Turkey in the east (Cramp & Perrins, 1994b; Hagemeijer & Blair, 1997). The UK is at the northern edge of its range. As in most other parts of its range, Cirl Buntings in the UK are resident throughout the year.

The Cirl Bunting was first reported breeding in the UK in 1800 and by the late 19th and early 20th centuries it had become a common species in southern Britain, its range extending to north Wales and Yorkshire (Holloway, 1996; Evans, 1997a). A decline in numbers was evident in the 1940s, however, and in the following decades the species became increasingly confined to southern English counties. Numbers crashed in the 1970s (Sitters, 1982) and a BTO survey in 1982 revealed that the population had fallen to a maximum of 167 pairs (Sitters, 1985). In 1989, a joint RSPB and Devon Bird Watching and Preservation Society survey found just 118-132 pairs, 97% of which were in Devon (Evans, 1992). Recently, the population has partially recovered and in 1995, there were 373 known pairs (Evans, 1997a).

The decline in the UK has been consistent with a long-term range contraction away from north-west Europe. The species has become extinct in Belgium and Luxembourg and declines have also been recorded in France, Switzerland and Germany (Robins, 1986; Evans, 1992). In contrast, in Spain and eastern Europe, the species' range has recently expanded and breeding has spread to Austria and Hungary (Dvorak *et al.*, 1993; Ponz *et al.*, 1996; Hagemeijer & Blair, 1997).

Factors affecting Populations

Possible causes of the Cirl Bunting's decline in the UK include climate change, interspecific competition and habitat loss and deterioration. Little evidence exists, however, to support the former two hypotheses. Although rainfall may lead to a switch in nestling diet and reduce growth rates, for example, there has been no trend towards increased rainfall during the species' breeding season (Evans *et al.*, 1997). Whilst severe winter weather may lead to increased mortality in Cirl Buntings (Wilson, 1974), there has been no recent trend towards colder winters (Sitters, 1985). Cirl Buntings certainly do occupy a similar niche to that of the Yellowhammer *Emberiza citrinella*, but few studies have reported competition between the two species (Sitters, 1985; Evans, 1997a).

A number of habitat changes may have affected this species. In the UK, the Cirl Bunting is primarily associated with farmland and in particular with hedgerows. Sitters (1985), for example, found that of 123 territories, 117 contained hedgerows and the remaining six contained scrub. Most nests are in hedgerows, particularly in brambles *Rubus fruticosus*, dog-rose *Rosa*, hawthorns *Crataegus* or blackthorns *Prunus spinosa* (Cramp & Perrins, 1994b) and the trees within them are used as songposts. The removal of hedgerows certainly has resulted in the loss of many territories, but it is unlikely that this has limited numbers as there are still many unoccupied potential sites with apparently suitable hedges (Evans, 1997a). Similarly, the loss of hedgerow elms *Ulmus*, following the outbreak of Dutch elm disease, is also unlikely to have affected numbers. The species' decline began before the disease's spread in the 1970s, males will sing from any other available raised perch and on the continent, Cirl Buntings occur in many areas without elms (Sitter, 1985; Evans, 1997a).

It is more probable that the population decline has been a result of changes in farming practices and in particular, the loss of extensive mixed-farms (Fuller *et al.*, 1995; Campbell *et al.*, 1997; Evans 1997a). As with other seed-eating species, the Cirl Bunting is particularly dependent upon winter stubbles and their loss following the switch from spring- to autumn-sown cereals is likely to have increased mortality rates

(Evans & Smith, 1994). The quality of remaining stubbles has also declined, as grain is now harvested more efficiently and as weeds are now more efficiently controlled with herbicides (Evans & Smith, 1994, Campbell *et al.*, 1997; Evans, 1997a). During the breeding season, Cirl Buntings are particularly dependent upon grasshoppers (Evans *et al.*, 1997). Late season nestlings have a higher proportion of grasshoppers in their diet and have higher growth and survival rates. However, many traditional pastures, which supported high densities of grasshoppers due to their mix of long and short grass and bare ground, have either been lost to arable farming or treated with inorganic fertilizers to give a rich uniform sward of grass (Evans, 1997a; Evans *et al.*, 1997). Lack of food may also lead to increased nest predation rates, due to the noisy begging activity of starving nestlings (Evans *et al.*, 1997).

Present and Future Research

The Cirl Bunting's breeding population is currently monitored by the RBBP (JNCC). Research into habitat use and breeding performance has been carried out by the RSPB (e.g. Evans & Smith, 1994; Evans *et al.*, 1997) and has given a good understanding of the species' requirements. This has led to direct conservation action, mainly through an RSPB Species Action Plan and an EN Species Recovery Plan, which, for example, have encouraged farmers to retain stubble fields through the winter as set-aside (Smith *et al.*, 1992). It is probable that the recent increase in the Cirl Bunting population has in part been a result of this provision of winter habitat (Ogilvie *et al.*, 1996; Evans, 1997a, 1997b). It is important that the monitoring of breeding productivity is continued and that repeat surveys of the breeding population take place every ten years (EN).

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2.14 Reed Bunting *Emberiza schoeniclus*

Status and Population Trends

Current European Population 3,200,000-4,400,000 pairs

SPEC None

EC Birds Directive

Berne Convention Appendix II

WCA 1981

Wildlife (NI) Order 1985

Biodiversity Steering Group Middle List

Species of Conservation Concern Red List

Species of Conservation Importance Table 3

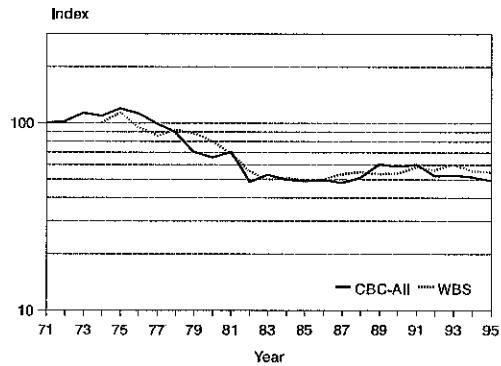


Figure 2.14.1 Population trends from the Common Birds Census and the Waterways Bird Survey

The Reed Bunting's breeding distribution extends across the Palearctic from western Europe to Japan and as far north as the subarctic (Cramp & Perrins, 1994b; Hagemeyer & Blair, 1997). Although north-eastern European populations are migratory, that in the UK is largely sedentary (Prŷs-Jones, 1984).

The Reed Bunting is a widespread and common species in the UK. Although typical of marshlands and riversides, the species expanded into drier habitats after the 1930s, probably as a result of a population increase (Kent, 1964; Summers-Smith, 1968; Bell, 1969; Gordon, 1972; Sharrock, 1976; Marchant *et al.* 1990). Reed Buntings now occur along hedgerows, on heaths, in young plantations and even in arable crops, such as barley and oilseed rape (Williamson, 1968; Burton *et al.*, 1996). During the 1970s and early 1980s, however, the species showed a sharp population decline. CBC data show that the population fell by 60% between 1971 and 1995, and by 61% on farmland alone (Crick *et al.*, 1997; Fig. 2.14.1). Similarly, the WBS indicates a 49% decline between 1974 and 1995. CES data show a 44% decline in catches of adults between 1983 and 1995 and a 61% decline in catches of juveniles. BBS counts declined by 11% between 1994 and 1996 (Gregory *et al.*, 1997). Breeding atlas data show that the Reed Bunting's ranges in both Britain and Ireland contracted by 12% between 1968-72 and 1988-91. Absences are now apparent in northern Scotland, the Pennines, Wales and the south-west of Britain (Gibbons *et al.*, 1993). At the time of the second atlas, there were an estimated 220,000 territories in Britain and 130,000 in Ireland.

In contrast to the UK, the population in Ireland is thought to be increasing, despite the apparent range contraction (Hutchinson, 1989; Gibbons *et al.*, 1993). However, declines have been recorded in Belgium, Finland, Germany, Greece, Italy, Lithuania and Moldova (Hagemeyer & Blair, 1997). The UK population represents between 5% and 10% of that in Europe as a whole.

Factors affecting Populations

In many parts of its range, the Reed Bunting is susceptible to severe winter weather, particularly when snow covers its food supplies (Prŷs-Jones, 1984). Marchant *et al.* (1990) reported that the UK population fell markedly after the cold winters of 1961/62 and 1962/63. The more recent decline, however, is thought to be related to changing farming practices. As with other buntings, the Reed Bunting has probably been affected by the loss of winter stubbles and their associated food supplies, following the switch from spring- to autumn-sown cereals (Wilson *et al.*, 1996). The increased use of herbicides has reduced the availability of weed seeds and more efficient harvesting and storage has also meant that there is less spilt grain for winter feeding (O'Connor & Shrubbs, 1986; Gibbons *et al.*, 1993; Campbell *et al.*, 1997). The reduction in winter farmland food supplies may be the cause of the increased use of suburban garden feeding stations recorded since the late 1970s (Thompson, 1988; BTO Garden BirdWatch survey).

Other habitat changes have also been linked to the decline, notably increased drainage, the loss of wetlands, removal of cover at field margins and the disappearance of traditional pastures (Møller, 1980; Hagemeyer & Blair, 1997). Such changes may have contributed to decreased productivity: daily nest failure rates have increased significantly since 1971 (Crick *et al.*, 1997; Fig. 2.14.2).

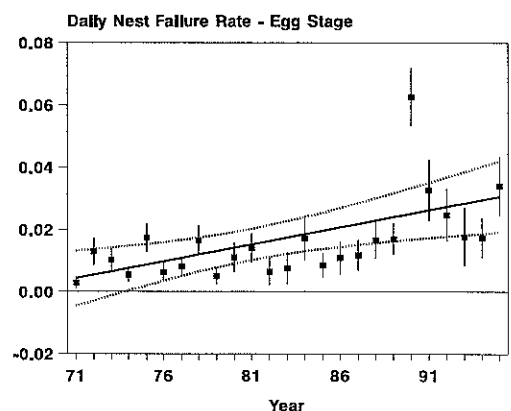


Figure 2.14.2 Long-term trend from the Nest Record Scheme

Present and Future Research

The Reed Bunting's status is currently monitored by the BBS and CES scheme and breeding productivity assessed by the CES scheme and the NRS (BTO, JNCC, RSPB). A survey for monitoring winter populations should also be considered (CCW, EHS, EN, JNCC, SNH). Data collected during the BTO Corn Bunting survey in 1993 should provide information on the Reed Bunting's habitat preferences (CCW, EHS, EN, JNCC, SNH). Further research should investigate the effect of habitat and food supply on breeding success (CCW, EHS, EN, JNCC, SNH).

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2.15 Wryneck *Jynx torquilla*

Status and Population Trends

Current European Population 350,000-420,000 pairs

SPEC Category 3

SPEC Status Declining (moderate decline)

EC Birds Directive

Berne Convention Appendix II

WCA 1981 Schedule 1

Biodiversity Steering Group Middle List

Species of Conservation Concern Red List

Species of Conservation Importance Table 2

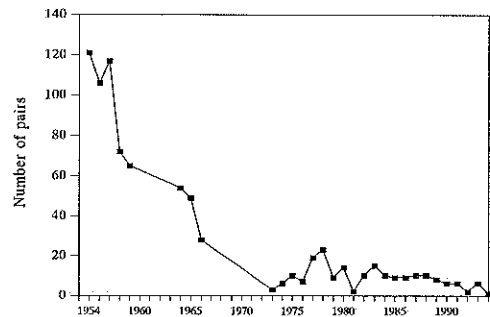


Figure 2.15.1 Maximum possible numbers of breeding Wrynecks in the UK between 1954 and 1994 (from Monk, 1963; Peal, 1968; Spencer *et al.*, 1986a; Ogilvie *et al.*, 1996b).

The Wryneck's breeding range extends across the boreal, temperate and subtropical zones from western Europe to Japan (Hagemeijer & Blair, 1997). The UK is at the extreme north-west of its range. The European Wryneck population winters predominantly in sub-Saharan Africa (Cramp, 1985; Hagemeijer & Blair, 1997).

The Wryneck was formerly widespread in the UK, breeding, at least occasionally, north to Durham and Cumbria and in all counties of Wales (Holloway, 1996). Over this century, however, the species has undergone a massive decline. The maximum possible number of pairs fell from 121 to 65 on surveys of England and Wales between 1954 and 1958 (Monk, 1963) and from 54 to 28 between 1964 and 1966 (Peal, 1968). The majority of records during both survey periods came from Kent (64% between 1954 and 1958 and 51% between 1964 and 1966). The decline has continued since then and breeding is now very rare in southern England (Fig. 2.15.1; Ogilvie *et al.*, 1996b). Breeding was confirmed for the first time in the Scottish Highlands in 1969 at three different sites (Burton *et al.*, 1970), but no more than four pairs have been proven to breed since then (Gibbons *et al.*, 1993).

The Wryneck is now more familiar in the UK as a migrant on passage to and from northern breeding grounds. Large and stable populations still occur in eastern Europe - around the Baltic, in Russia and Belarus (Hagemeijer & Blair, 1997). The species' decline in the UK reflects that seen over much of the rest of western Europe. In Germany the species has declined considerably over this century, despite increases in the 1940s and 1950s (Epple, 1992; Winkel, 1992; Bammerlin, 1993). Wrynecks have become extinct in northern France and have also declined in the Netherlands, Belgium, Luxembourg and Denmark (Cramp, 1985; Hagemeijer & Blair, 1997). The Finnish population declined by over 50% between 1970 and 1990, although the species' range there has not contracted (Koskimies, 1989).

Factors affecting Populations

The Wryneck's dramatic decline in north-west Europe has been associated with changes in temperature and rainfall (Hagemeijer & Blair, 1997). The species prefers dry, sunny climates and in Sweden, in the north of its range, it has been noted to be more common on south or south-east facing slopes (Peal, 1968). Wet, cool summers, which do not favour ants, the Wryneck's main food source (Peal, 1968; Cramp, 1985), are thought to have accelerated the species' decline in the UK between 1890 and 1920 (Burton, 1995). In contrast, warm summers in the mid-1970s led to population increases in Holland (van Dijk *et al.*, 1994).

In recent decades, loss and deterioration of habitat have been important factors in the decline. Wrynecks breed in open woodlands, both deciduous and coniferous, and rely on tree cavities for nesting, although they can use nestboxes (Linkola, 1978; Cramp, 1985; Murai & Higuchi, 1991; Jacobsen, 1993). They are often found in parks, gardens and orchards (Walz & Walz, 1993), as they formerly were in southern England (Monk, 1963; Peal, 1968; Gibbons *et al.*, 1993). The loss of natural and grazed open woods in many parts of Europe and their replacement with dense managed forests, devoid of dead and dying trees, has reduced the availability of both ants and nesting cavities (Linkola, 1978). Many suitable forests, parklands and orchards still remain, however. Intensive fruit and cereal production, involving the large-scale use of pesticides and herbicides may have more directly affected food supplies (Hagemeijer & Blair, 1997). In the UK, reductions in grazing following the rabbit myxomatosis epidemic in the 1950s probably led to a decline in the abundance of the ant *Lasius flavus* (which requires short vegetation to found its colonies) and this too may have contributed to the Wryneck's decline (Potts, 1970).

The Wryneck is clearly on the verge of extinction as a UK breeding bird and the future of the species in this country is dependent upon immigration from other breeding areas. Gibbons *et al.*, (1993) noted that the Scottish population of Wrynecks was highest when large numbers of birds were observed on passage through the country, some perhaps deflected west on their migration to Scandinavia by high pressure weather systems. Both adult and first-year Wrynecks are strongly philopatric, however (Peal, 1972) and the population in Scotland is thus unlikely to expand quickly even with high rates of breeding success and survival.

Present and Future Research

There is a clear need to more accurately assess and monitor the UK's population of Wrynecks (EN, SNH). Whilst only single breeding attempts have been recorded in the UK in most recent years, other birds have been seen in suitable habitat (Gibbons *et al.*, 1993; Ogilvie *et al.*, 1996b). It is possible that some breeding pairs may have been overlooked as birds become almost completely silent once egg-laying is complete (Cramp, 1985).

The habitat requirements of these pairs should also be investigated (EN, SNH). This work may be supplemented by the results from a recent symposium on Wrynecks in Germany (Schmid, 1992) and other studies in Scandinavia (e.g. Jacobsen, 1993).

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2.16 Red-backed Shrike *Lanius collurio*

Status and Population Trends

Current European Population 2,600,000-3,700,000 pairs

SPEC Category 3

SPEC Status Declining (moderate decline)

EC Birds Directive Annex I

Berne Convention Appendix II

WCA 1981 Schedule 1

Biodiversity Steering Group Middle List

Species of Conservation Concern Red List

Species of Conservation Importance Table 2

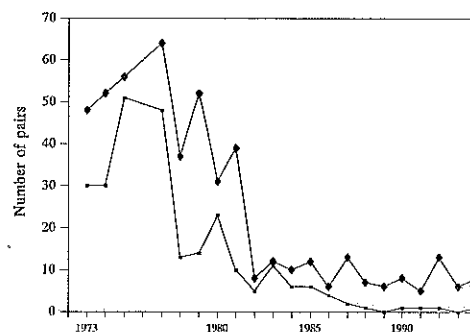


Figure 2.16.1 Confirmed (■) and total possible (●) numbers of breeding Red-backed Shrikes in the UK from 1973 to 1994 (from Spencer *et al.*, 1986a; Ogilvie *et al.*, 1996b).

The Red-backed Shrike's breeding distribution extends from northern Iberia in the west across most of Europe to mid-Siberia in the east (Cramp & Perrins, 1993; Hagemeijer & Blair, 1997). The UK is at the far north-west of its range. The species winters in sub-Saharan Africa and is concentrated in the Kalahari (Bruderer & Bruderer, 1993; Hagemeijer & Blair, 1997).

In the latter 19th century, the Red-backed Shrike bred throughout England and Wales, with the exception of the most northerly counties (Peakall, 1962, 1995; Holloway, 1996). Between 1850 and 1950, however, its range began to contract away from north-western England and Cornwall and since then it has declined sharply. A BTO survey in 1960 estimated the population at 172 pairs (re-estimated as 253 pairs by Bibby, 1973) the majority of which were on the heaths of the New Forest, Surrey and East Anglia (Peakall, 1962). A second BTO survey in 1971 revealed that the population had fallen to just 81 pairs, 59 (73%) of which were in Norfolk and Suffolk (see also Sharrock, 1976). Although Red-backed Shrikes bred for the first recorded time in Scotland in 1977, the population continued to decline and in 1989 there were no breeding records at all (Spencer *et al.*, 1986a; Peakall, 1995; Ogilvie *et al.*, 1996b). Breeding has been proven only occasionally since then (Fig. 2.16.1) and only in East Anglia (e.g. Garrod, 1994). Ash (1970) reported a detailed study of the Red-backed Shrike's decline in Hampshire. Within his study areas, the population fell from 68 pairs in 1957 to 31 in 1966. Breeding was last proven in Hampshire in 1978 (Peakall, 1995).

Red-backed Shrikes have also declined across much of Europe, particularly in the north and west (Tucker & Heath, 1994; Hagemeijer & Blair, 1997). In the Netherlands, for example, the population fell from 5,000-15,000 pairs in 1900 to 150-220 in 1989-1990 (Hustings & Bekhuis, 1993). In Sweden, where the population was estimated at 80,000 pairs in the early 1970s, numbers fell by 40% between 1975 and 1989 (Olsson, 1995). Breeding populations in the former Czechoslovakia are estimated to have declined by 90% in the last 30 years (Holan, 1995).

Factors affecting Populations

The primary cause of the Red-backed Shrike's decline in the UK and the rest of Europe is believed to be the deterioration and loss of habitat (Peakall, 1962, 1995; Bibby, 1973; Jakober & Stauber, 1987; Kowalski, 1992; Olsson, 1995). The species is found in a variety of semi-open habitats with scrub for nesting and perching and open ground for hunting (Carlson, 1985; Guerrieri, 1995; Olsson, 1995). It primarily preys on insects, but may also take small mammals and birds (Cramp & Perrins, 1993; Wagner, 1993; Esselink *et al.*, 1995). In the UK, it has recently been restricted to heathlands, but elsewhere occurs in low-intensity farmland interspersed with hedgerows, orchards, young forestry plantations and overgrown clearings (Cramp & Perrins, 1993; Tucker & Heath, 1994; Hagemeijer & Blair, 1997). The intensification of farming, increases in the area of cultivated land and afforestation have all resulted in the reduction of suitable habitat. In Sweden and Poland, for example, extensive cattle pastures, previously frequented by shrikes, have been lost to vegetational succession and planting with conifers (Diehl, 1995; Olsson, 1995). In intensively farmed areas, adults are unable to feed their young adequately as they expend too much energy flying long distances to sparse feeding areas (Diehl & Myrcha, 1973; Lügger, 1992). The fragmentation of habitat may also lead to an increase in predation rates of nests (Diehl, 1995; Matyjasik, 1995). The availability of hunting habitat has been reduced further by the use of nitrogen fertilizers, which cause vegetation to grow early, dense and high (Ellenberg, 1986; Marechal, 1993). Pesticides are also believed to have affected breeding success (Jakober & Stauber, 1987). In areas where habitat is maintained, populations may remain stable or, with good management, even increase. The

improvement of foraging conditions on a peat-moor reserve in the Netherlands, for example, has resulted in improved breeding success and survival of young and in a population increase (Esselink *et al.*, 1995). In the UK, however, much suitable habitat still remains, particularly in the Brecklands of East Anglia and in the New Forest. The loss of the species here, and from much of north-west Europe, may be partly due to climate change, as cooler, wetter summers may have reduced the availability of insect food (Bibby, 1973). Droughts in the Kalahari, the species' main wintering quarters, over the last 20 years may have also adversely affected numbers (Bruderer & Bruderer, 1993).

Present and Future Research

Breeding attempts are monitored by the RBBP (JNCC). To improve the chances of re-colonisation and the success of any breeding attempts, habitat management plans should be encouraged, based upon research such as that carried out in the Netherlands by Esselink *et al.*, (1995) (JNCC).

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2.17 Woodlark *Lullula arborea*

Status and Population Trends

Current European Population 1,100,000-2,200,000 pairs

SPEC Category 2

SPEC Status Vulnerable (large decline)

EC Birds Directive Annex I

Berne Convention Appendix III

WCA 1981 Schedule 1

Biodiversity Steering Group Middle List

Species of Conservation Concern Red List

Species of Conservation Importance Table 2

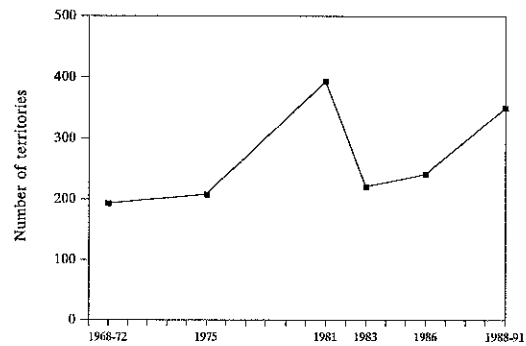


Figure 2.17.1 Maximum estimates of the United Kingdom Woodlark population between 1968 and 1991 reported by Sitters *et al.*, (1996).

The Woodlark is a predominately European species and outside of the continent only breeds in north-west Africa, the Levant, Iran and south Turkmenistan (Cramp, 1988; Hagemeijer & Blair, 1997). The UK population is at the extreme north-west of its range.

The species formerly bred in many English and Welsh counties south of the Pennines (Holloway, 1996), but from the 1950s had disappeared from many areas following habitat loss and the effects of hard winters (Sitters *et al.*, 1996). Populations in the New Forest and on the Hampshire/Surrey border, for example, were greatly reduced following the 1962/63 winter. The first breeding atlas (1968-72) estimated a population of 200-450 breeding pairs (Sharrock, 1976). Numbers dropped to 170-208 occupied territories in 1975, but rose to 369-393 territories in 1981 (Sitters *et al.*, 1996). The hard winter of 1981/82, however, reduced the number of occupied territories to between 208 and 221. The first national survey (organised by the BTO) in 1986 estimated the number of breeding territories at 241 and found that the species was confined to five core areas: Devon, Dorset and south Hampshire (including the New Forest), the Hampshire/Surrey border, Breckland and the Suffolk coast (Sitters *et al.*, 1996). The population was estimated at 350 pairs by the second breeding atlas (Gibbons *et al.*, 1993). Since then the population has continued to rise and there were an estimated 400 territories in 1991 (Burgess, 1991) and a possible 624 breeding pairs reported by the RBBP in 1994 (Ogilvie *et al.*, 1996b). Moderate increases have been noted in the New Forest and Hampshire/Surrey populations (Burgess, 1991; Sitters *et al.*, 1996) and marked increases in the Breckland and Suffolk coast populations (Sitters *et al.*, 1996). Woodlarks have also recently increased in Sussex, Lincolnshire and Nottinghamshire, and have recolonised Buckinghamshire (Sitters *et al.*, 1996). Many British Woodlarks leave their breeding sites in autumn and in winter most birds are found in Hampshire, Surrey and Devon. Lack (1986) estimated the winter population at 150-200 birds, a figure probably affected by the hard winter of 1981/82.

The majority of the Woodlark population in Europe breeds in Iberia (Hagemeijer & Blair, 1997). Continental populations have also fluctuated markedly in recent years, as a result of changes in habitat availability and the severity of weather in the species' main wintering area of south-west France and Iberia (Hagemeijer & Blair, 1997).

Factors affecting Populations

Sitters (1986), Sitters *et al.*, (1996) and Hagemeijer and Blair (1997) all concur that the fluctuations seen in Woodlark populations since the 1950s are primarily the result of changes in the availability of habitat and fluctuations in the severity of winter weather. Climatic change has formerly been proposed as a factor in the species' decline in the UK (Harrison, 1961; Parslow, 1971) but presently is believed to be of only minor importance (Sitters, 1986). As has been noted above, numbers in parts of the UK were markedly reduced following the hard winters of 1962/63 and 1981/82 (Sitters *et al.*, 1996). In continental Europe, three severe winters from 1985 also depleted many populations (Hagemeijer & Blair, 1997). The Woodlark, however, is a prolific breeder, raising two or sometimes three broods annually (Cramp, 1988) and recent mild winters have allowed populations in northern Europe to expand (Bijlsma *et al.*, 1985, 1988; Sitters *et al.*, 1996). The population increase in the UK, though, has not been associated with improved breeding performance (Crick *et al.*, 1997).

Changes in habitat have been responsible for both decreases and increases in the Woodlark's range and numbers. The species forages in areas of bare ground and low vegetation, but requires scattered trees for

songposts and some taller cover for nesting (Harrison & Forster, 1959; Sitters, 1986). It is found on lowland heaths, unimproved pasture and in open oak *Quercus* and pine woods and forestry plantations (Hagemeijer & Blair, 1997). Vegetation succession, as a result of reductions in grazing, has reduced the quality and availability of both heathland and downland habitats (Marrs *et al.*, 1986). In the UK, many downland populations were lost in the 1950s and 1960s following a reduction in rabbit numbers and the cessation of traditional downland farming (Sitters *et al.*, 1996). Heathland has also been lost to agriculture, housing and forestry (Cadbury, 1989; Webb, 1990). Woodlarks, however, are quick to respond to the availability of new habitat. In Breckland, for example, the species has colonized recently cleared and restocked conifer plantations (Bowden & Hoblyn, 1990; Bowden & Green, 1992). Wind damage to plantations and subsequent clearance has also created new open habitat (Wright, 1990) whilst heathland fires are thought to have aided the population increases on southern heaths in the early 1980s (Sitters *et al.*, 1996). Recent conservation effort has been aimed at restoring degraded heathland and traditional grazing and ploughing practices. As a result of such management, the Woodlark recolonized the Minsmere RSPB reserve in Suffolk in 1992 and there were 19 territories in 1994 (Sitters *et al.*, 1996).

Present and Future Research

A second national survey of breeding Woodlarks (organised by the BTO) took place in 1997. The survey will report on population and distribution changes and on habitat preferences. The next survey is planned for 2007 (JNCC). Breeding productivity is currently monitored by the NRS (BTO, JNCC). Research on the habitat use of Woodlarks in restocked conifer plantations in Breckland has been carried out by the RSPB and FC. The results of this have been used in subsequent Forest Design Plans to help maintain good numbers of Woodlarks.

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2.18 Common Scoter *Melanitta nigra*

Status and Population Trends

Current European Population 4,900-14,100 pairs

SPEC None

EC Birds Directive Annexes II/II & III/I

Berne Convention Appendix III

WCA 1981 Schedule 1

Wildlife (NI) Order 1985

Biodiversity Steering Group Middle List

Species of Conservation Concern Red List

Species of Conservation Importance Table 4

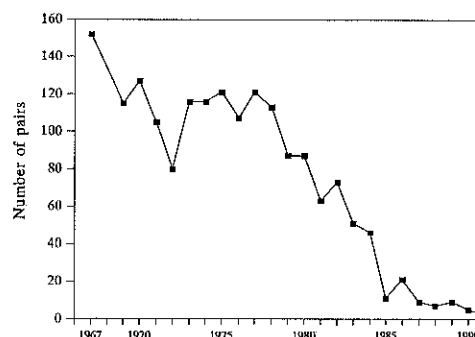


Figure 2.18.1 Maximum possible numbers of breeding Common Scoters at Lower Lough Erne, between 1967 and 1991 (from Hutchinson, 1989; Gibbons *et al.*, 1993).

The Common Scoter has a circumpolar breeding distribution encompassing tundra and shrubby wetlands in the low-arctic and boreal zones (Cramp & Simmons, 1977; Hagemeijer & Blair, 1997). The European subspecies *Melanitta nigra nigra* breeds from Iceland, north-west Ireland and Scotland through Fennoscandia to the River Lena in central Siberia.

The Common Scoter expanded its breeding range between 1850 and 1950 during a period of climatic amelioration (Burton, 1995). Breeding was first reported in Scotland in 1855 and in Ireland in 1905, and during the first decades of this century numbers increased steadily. At Lower Lough Erne in Ireland, seven pairs were recorded in 1917, a minimum of 50 in 1950 and 152 in 1967 (Hutchinson, 1989). The species spread to Co. Mayo in the Irish Republic in 1948. From 1967, however, the Northern Irish (Lower Lough Erne) population underwent a severe decline and the species became extinct there in the early 1990s (Fig. 2.18.1; Ruttledge, 1987; Partridge, 1989; Hutchinson, 1989; Gibbons *et al.*, 1993; Gittings & Delaney, 1996). The Irish breeding population is now confined to four sites: Loughs Conn and Cullin (Co. Mayo), Lough Ree on the Shannon and Lough Corrib (Co. Galway). The largest populations are found at the former two sites and these are believed to be stable, whilst that at Lough Corrib may be increasing (Ruttledge, 1987; Partridge, 1989; Gibbons *et al.*, 1993). The Scottish population has been less well monitored, due to the remoteness of many of the breeding sites. In the flow country, numbers declined from 55 pairs in 1988 to only 28 in 1996 (Underhill & Hughes, 1996). At the most well documented site, Loch Lomond, the Common Scoter was first recorded breeding in 1971 and although numbers increased to an estimated seven pairs in 1976 (Mitchell, 1977), the species was not recorded there at the time of the second breeding atlas (Gibbons *et al.*, 1993). Breeding atlas data, however, suggest that the Common Scoter's range expanded by 21% in Britain and 7% in Ireland between 1968-72 and 1988-91.

The first detailed survey of breeding Common Scoter in Britain and Ireland was undertaken in May-June 1995 and covered five different areas: Shetland, the flow country, west and south Scotland, Islay and Ireland. Eighty-nine females were found in Scotland and 111 in Eire (Gittings & Delaney, 1996), and in total 455 adult birds were recorded (Underhill *et al.*, in press). This survey was repeated in 1996, due to concern for the effects on the British and Irish breeding populations of the *Sea Empress* oil spill which killed a minimum of 1,700 Common Scoters in February 1996, the majority in Carmarthen Bay (Parr *et al.*, 1997). A total of 376 adults was recorded, suggesting a 17% decline in numbers, and there was a 31% decrease in the number of occupied sites (Underhill & Hughes, 1996; for Ireland see, Delaney & Gittings, 1996).

The British and Irish populations of Common Scoter are negligible in comparison to those in Scandinavia and Russia, where an estimated 105,000-134,000 pairs breed (Hagemeijer & Blair, 1997). Estimates of winter numbers (up to 1.3 million Common Scoter winter between Fennoscandia and north-west Africa: Durinck *et al.*, 1994) suggest that the western Palearctic population as a whole is stable (Hagemeijer & Blair, 1997).

Factors affecting Populations

The decline at Lower Lough Erne is believed to be due to a reduction in invertebrate food supply. This has primarily been a consequence of eutrophication, which has been caused by increases in the fertilizer, sewage and industrial effluent inputs into the Erne's catchment (Partridge, 1989). It is also believed that invertebrates have been depleted by introduced roach *Rutilus rutilus*. The reduced food supply was probably a major factor in the poor survival of ducklings at Lower Lough Erne (Partridge, 1989). A study

in Iceland concluded that population size was determined by the food supply available to young ducklings (Gardarsson, 1979). Water quality has not deteriorated at other Irish sites and here populations remain stable.

Breeding birds are also threatened by predation, for example from Hooded Crows *Corvus corone cornix*, Magpies *Pica pica*, foxes and mink (Cramp & Simmons, 1977; Partridge, 1989). At Lower Lough Erne, the latter are believed to have caused nest desertions and to have killed incubating females (Partridge, 1989). In the flow country of Scotland, numbers of crows and foxes have increased following afforestation, which also resulted in a direct loss of nesting habitat (Stroud *et al.*, 1987; Partridge, 1989).

As the *Sea Empress* oil spill highlighted, wintering concentrations of Common Scoter are particularly susceptible to pollution. The species is also threatened in winter by the over-fishing of its bivalve food stocks (Leopold, 1993).

Present and Future Research

Research following the *Sea Empress* oil spill has emphasized the need for more regular monitoring of both breeding and wintering populations (CCW, EHS, EN, JNCC, SNH). Although recorded breeding numbers declined between 1995 and 1996, it remains unclear as to whether this was a consequence of the spill, a result of a natural population fluctuation or due to the delayed arrival of birds after a late spring in 1996 (Underhill & Hughes, 1996). It is also unclear as to whether UK breeding birds were involved in the spill, as their wintering grounds have yet to be determined (CCW, EHS, EN, JNCC, SNH). Current monitoring in Carmarthen Bay and Cardigan Bay aims to record the distribution and numbers of scoters and make comparisons to information collected before the oil spill (Stewart, 1995; CCW, WWT).

There is also a need for further investigation into Common Scoter breeding ecology, which at present is best known from the former Lower Lough Erne stronghold. In particular, there is a need to understand the role of water quality on food availability and thus on breeding success, and to determine the impacts of predators (FC, SNH, SEPA).

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2.19 Corn Bunting *Miliaria calandra*

Status and Population Trends

Current European Population 3,500,000-6,800,000 pairs

SPEC Category 4

SPEC Status Secure

EC Birds Directive

Berne Convention Appendix III

WCA 1981

Wildlife (NI) Order 1985 Schedule 1.

Biodiversity Steering Group Middle List

Species of Conservation Concern Red List

Species of Conservation Importance Table 2.

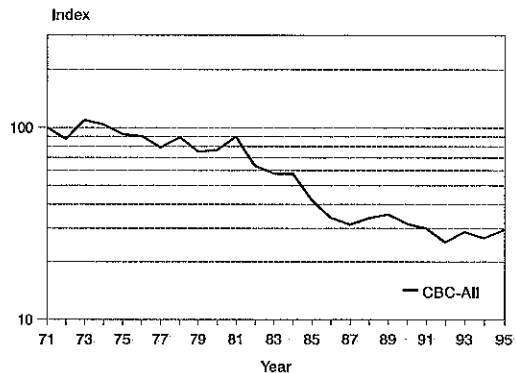


Figure 2.19.1 Population trend from the Common Birds Census

The Corn Bunting's breeding distribution extends from the Canary Islands in the west, to central Asia in the east and from the UK and Denmark in the north to Iran, Iraq and north Africa in the south (Cramp & Perrins, 1994b; Harper, 1995; Hagemeijer & Blair, 1997). Although the UK population is largely sedentary, partially or totally migratory populations do occur in central and eastern Europe.

At the end the 19th century, the Corn Bunting was a widespread and often common species in lowland Britain and Ireland (Donald *et al.*, 1994; Holloway, 1996; Donald, 1997; Shrubbs, 1997). This century, however, there have been two massive population declines, the first in the 1920s and 1930s, which was followed by a period of slow recovery and a second since the early 1970s (Donald *et al.*, 1994; Donald, 1997; Shrubbs, 1997). During both declines, the Corn Bunting's range has contracted away from the north and west and at present the species is virtually extinct in Wales, the western Scottish mainland, south-west England and Ireland (Gibbons *et al.*, 1993; Donald, 1997). The CBC index for the species indicates a decline of 70% between 1971 and 1995 (Crick *et al.*, 1997; Fig. 2.19.1) and the BBS recorded a 21% decline between 1994 and 1996 (Gregory *et al.*, 1997). A BTO survey estimated the British population at 20,000 territories in 1993 (Donald & Evans, 1995).

Corn Buntings have also recently declined over much of north-west and central Europe (Donald *et al.*, 1994; Hagemeijer & Blair, 1997). In the Netherlands, for example, the population fell from 1,100-1,250 pairs in 1975 to 125-175 in 1989 (Hustings *et al.*, 1990). Similarly, the population in Schleswig-Holstein in Germany crashed from 3,000-4,000 pairs in 1955 to just 40 in 1987 (Busche, 1989). Much larger populations still exist in Spain, Portugal, France, Italy, Greece and Bulgaria (Hagemeijer & Blair, 1997). Estimates of the Corn Bunting's breeding populations are complicated by its mating system, which varies between co-operative polyandry and polygamy (Ryves & Ryves, 1934; Thompson & Gribbin, 1986; Hartley *et al.*, 1995; Hartley & Shepherd, 1997).

Factors affecting Populations

The Corn Bunting is a species of open country and in the UK is found almost exclusively on farmland. It occurs on downland in southern England and on the machair of the Outer Hebrides, but over much of its UK distribution is associated with cereal farming (Donald & Evans, 1995; Gilling & Watts, 1997). Its decline in the UK and north-west Europe is believed to be largely associated with changes in farming practices, although the exact causes are not well known. Breeding success may have been affected by a loss of invertebrate food resultant from an increased use of pesticides and the switch from spring- to autumn-sown cereals and by nest losses caused by the earlier harvesting of both cereals and grass (Donald *et al.*, 1994; Campbell *et al.*, 1997; Donald, 1997). Recent studies have shown, however, that Corn Bunting productivity has actually risen since the 1970s (NRS; Crick *et al.*, 1994; Crick, 1997). Ward and Aebischer (1994) have also found that the area of crop treated with pesticide does not affect densities of breeding Corn Buntings.

As breeding productivity has not fallen, it seems probable that the Corn Bunting's decline is largely a result of increased winter mortality. The change from spring- to autumn-sown cereals has reduced the area of winter stubbles, an important feeding habitat for Corn Buntings at this time (Donald & Evans, 1994; Brickle, 1996). More efficient harvesting and storage has also meant that there is less spilt grain for winter feeding (Donald *et al.*, 1994). The reduction in the area of barley, the most extensive spring-sown crop in the UK and one particularly associated with the Corn Bunting (Ward & Aebischer, 1994; Gillings & Watts, 1997), may have been particularly important (Donald, 1997). Although pesticides do not have an obvious effect on breeding success, it is probable that they have severely depleted winter food supplies (Donald, 1997).

It is also possible that the Corn Bunting's decline is related to non-agricultural factors - Donald and Forrest (1995), for example, found that population changes on individual farms were uncorrelated with any measures of agricultural change. Hustings *et al.*, (1990) suggested that climate change may have been a factor in the species' decline in the Netherlands, but there is little evidence that it has been important in the UK (Donald, 1997).

Present and Future Research

The monitoring of the Corn Bunting is continuing through the BBS (BTO, JNCC, RSPB). An equivalent survey for monitoring winter populations should also be considered (CCW, EHS, EN, JNCC, SNH). Breeding productivity is currently assessed by the NRS (BTO, JNCC). Further research on the ecological requirements of the Corn Bunting should consider the effects of ESAs, the pilot arable scheme and other initiatives that may encourage farmland birds (CCW, DANL, EHS, EN, JNCC, MAFF, SNH, SOAEFD, WOAD).

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2.20 Spotted Flycatcher *Muscicapa striata*

Status and Population Trends

Current European Population 7,100,000-8,800,000 pairs

SPEC Category 3

SPEC Status Declining (moderate decline)

EC Birds Directive

Berne Convention Appendix II

WCA 1981

Wildlife (NI) Order 1985

Biodiversity Steering Group Middle List

Species of Conservation Concern Red List

Species of Conservation Importance Table 3

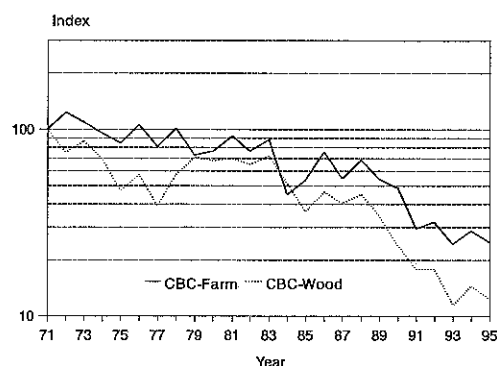


Figure 2.20.1 Population trends from the Common Birds Census

The Spotted Flycatcher's breeding distribution includes most of Europe and extends eastwards to Lake Baikal in Siberia (Cramp & Perrins, 1993; Hagemeyer & Blair, 1997). Its strongholds in Europe are in Finland, Sweden and Belarus and in comparison, the UK population is small. The majority of Spotted Flycatchers winter south of the equator in Africa (Cramp & Perrins, 1993).

The Spotted Flycatcher is a widespread species in the UK, although is largely absent from the Outer Hebrides, Orkneys and Shetlands (Gibbons *et al.*, 1993). It has been in decline at least since the early 1960s. CBC data show that the population fell by 79% between 1971 and 1995, by 76% on farmland alone and by 83% in woodland (Crick *et al.*, 1997; Fig. 2.20.1). Likewise, CES data indicate a 58% decline in catches of adults between 1983 and 1995 and a 46% decline in catches of juveniles. The BBS has recorded a 14% decline between 1994 and 1996 (Gregory *et al.*, 1997). Breeding atlas data show that the Spotted Flycatcher's range in Britain contracted by 2% between 1968-72 and 1988-91, and that in Ireland by 18% (Gibbons *et al.*, 1993). At the time of the second atlas, there were an estimated 120,000 territories in Britain and 35,000 in Ireland.

Breeding numbers of Spotted Flycatchers fluctuate markedly, so it is often difficult to judge population trends (Peklo, 1987). However, it is evident that there have also been declines in many other parts of Europe, including Finland, Germany and Spain (Koskimies, 1993; Tucker & Heath, 1994; Hagemeyer & Blair, 1997). In some areas, populations have remained stable, for example in the Białowieża forest in Poland (Tomiałojć & Wesolowski, 1994).

Factors affecting Populations

The Spotted Flycatcher is an insectivorous species which catches its prey in flight on sallies from exposed perches. It is characteristically found in open woodland, but is also associated with parkland, gardens and orchards. High densities are often found in parks and cemeteries (Flade, 1994; Leibak *et al.*, 1994).

The reasons for the Spotted Flycatcher's declines in the UK and elsewhere in Europe are not fully understood. It is possible that breeding success may have been affected by the cool, wet summers of recent decades - pairs breed earlier and clutch sizes are larger in warm, sunny springs (Marjakangas, 1982; O'Connor & Morgan, 1982; Epprecht, 1985). In poor weather, the availability of flying insects decreases and birds may be forced to search for prey among the foliage of trees (Davies, 1977). NRS data, however, show that nest losses decreased between 1971 and 1995 (Fig. 2.20.2) and that brood sizes increased (peaking in the 1980s) (Crick *et al.*, 1997).

Droughts in wintering areas may also have affected populations. There is some correlation, for example, between the decline in the UK and the failures of rains in the Sahel in 1983 and 1984 (Marchant *et al.*, 1990). However, the Spotted Flycatcher was certainly not as severely affected by the 1969 drought as other species such as the Whitethroat *Sylvia communis* and Sand Martin *Riparia riparia*. The lack of a uniform decline across Europe suggests that local factors may be important (Tucker & Heath, 1994; Hagemeyer & Blair, 1997). Habitat degradation, notably the loss of old trees on

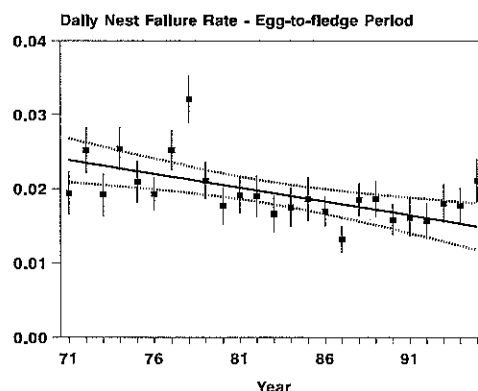


Figure 2.20.2 Long-term trend from the Nest Record Scheme

farmland and through modern forestry practices, and the widespread use of insecticides may, for example, have affected populations (Marchant *et al.*, 1990; Sachslehner, 1992).

Present and Future Research

The Spotted Flycatcher's status is currently monitored by the BBS and CES scheme and breeding productivity assessed by the CES scheme and the NRS (BTO, JNCC, RSPB). Further research is needed to investigate the species' habitat requirements and the relationship between food availability and the growth and survival of chicks and fledging success (CCW, EHS, EN, JNCC, SNH). Analysis of NRS data from the 1960s may help indicate whether changes in productivity could have led to the species' decline (CCW, EHS, EN, JNCC, SNH).

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2.21 Tree Sparrow *Passer montanus*

Status and Population Trends

Current European Population. 13,900,000-17,500,000 pairs

SPEC None

EC Birds Directive

Berne Convention Appendix III

WCA 1981

Wildlife (NI) Order 1985

Biodiversity Steering Group Middle List

Species of Conservation Concern Red List

Species of Conservation Importance Table 3

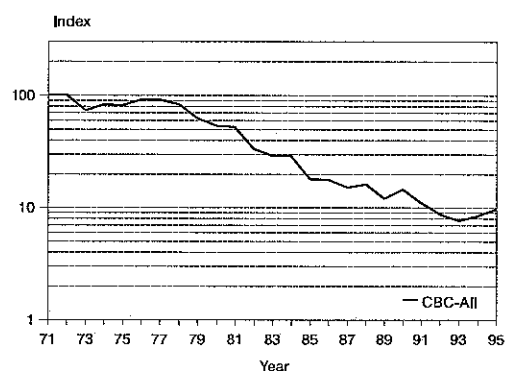


Figure 2.21.1 Population trend from the Common Birds Census

The Tree Sparrow's range extends across central Eurasia, from Portugal to south-east Asia (Cramp & Perrins, 1994a; Summers-Smith, 1988, 1995). It has also been introduced to the United States and Australia (Summers-Smith, 1995). The UK is at the western edge of the range and the species only recolonized Ireland in the late 1950s (Sharrock, 1976; Gibbons *et al.*, 1993; Summers-Smith, 1995).

The Tree Sparrow is widely distributed in the UK with strongholds in central and eastern England, adjoining counties of Wales and in eastern Scotland (Holloway, 1996). Over the last 150 years, however, both its numbers and range have shown marked fluctuations. It is believed that the population was at a maximum at the turn of the century, that a decline occurred between 1930 and 1955 and that a dramatic increase followed from 1960 to 1978 (Summers-Smith, 1989). Since then the population has crashed, the CBC recording a decline of 95% between 1971 and 1995 (and an 89% decline on farmland alone; Crick *et al.*, 1997; Figure 2.21.1). The BBS revealed a decline of 15% between 1994 and 1996 (Gregory *et al.*, 1997). Breeding atlas data show that the Tree Sparrow's range in Britain contracted by 20% between 1968-72 and 1988-91 (Gibbons *et al.*, 1993). At the time of the second atlas, there were an estimated 110,000 territories in Britain and 9,000 in Ireland.

Similar fluctuations have occurred elsewhere in north-west Europe, for example in Germany and Switzerland, where the species is also currently in decline (Moritz, 1981; Wesolowski, 1991; Winkel, 1994). In contrast, the species' numbers and range have recently increased in both Scandinavia and Iberia (Koskimies, 1989; Hagemeyer & Blair, 1997). The UK population is small in comparison to those in Spain, Germany and eastern Europe.

Factors affecting Populations

Summers-Smith (1989) in his review of the Tree Sparrow's population fluctuations over the last 150 years, suggested that upsurges in the UK population were the result of irruptions from the continent following high population levels there. It is unclear what has driven these European population increases, although that in the 1950s coincided with the decline of the Sparrowhawk following the introduction of organochlorine pesticides (Summers-Smith, 1995). The subsequent recovery of the Sparrowhawk (Newton & Haas, 1984) may be linked to the Tree Sparrow's current decline, but a more probable cause is the reduction in food availability associated with changes in agricultural practices (Fuller *et al.*, 1995). As with other seed-eating birds, the loss of winter stubbles following the switch from spring- to autumn-sown cereals is likely to have increased mortality rates. In the remaining stubbles, there is now less spilt grain as harvesting is more efficient and also fewer weeds due to the use of herbicides (Campbell *et al.*, 1997). Pinowski and Pinowski (1985) also propose that the survival of Tree Sparrows during winter is dependent upon the duration of snow cover, which prevents feeding.

Wesolowski (1991), in a study in Switzerland, suggested that the Tree Sparrow's decline there was associated with a fall in breeding success. The reproductive rate of the species was 21% lower between 1962 and 1979 than between 1940 and 1961, probably due to the large-scale use of organochlorines in farming. Summers-Smith (1995) likewise showed that annual productivity in the UK was highest when the population was at a maximum in 1965 and has since declined.

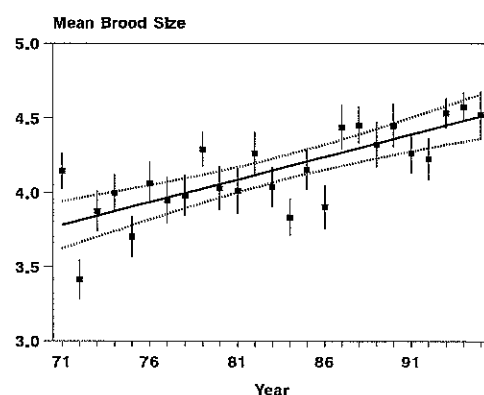


Figure 2.21.2 Long-term trend from the Nest Record Scheme

However, more recent analysis of NRS data has shown that both clutch sizes and brood sizes (Fig. 2.21.2) increased between 1971 and 1995 (Crick *et al.*, 1997).

It is also possible that Tree Sparrows may have been affected by a loss of nest-sites following the loss of elms through Dutch elm disease and the removal of other hedgerow trees. Tree Sparrows are not strongly dependent upon elms, however, and utilise a wider variety of tree species than many other hole-nesting birds (Osborne, 1982; Marchant *et al.*, 1990). Tree Sparrows also readily use nest-boxes (Summers-Smith, 1989, 1995).

Present and Future Research

The Tree Sparrow's status is currently monitored by the BBS and breeding productivity assessed by the NRS (BTO, JNCC, RSPB). A survey for monitoring winter populations should also be considered (CCW, EHS, EN, JNCC, SNH). There is a need to carry out a more thorough analysis of NRS scheme data, in order to investigate possible reasons for the decline (CCW, EHS, EN, JNCC, SNH). Whilst the Tree Sparrow's breeding biology is well known (see Summers-Smith, 1995) there is a need for more detailed investigation of the relationships between breeding performance (perhaps through nest-box schemes) and habitat use and food availability (CCW, EHS, EN, SNH).

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2.22 Red-necked Phalarope *Phalaropus lobatus*

Status and Population Trends

Current European Population 65,500-94,300 pairs

SPEC None

EC Birds Directive Annex I

Berne Convention Appendix II

WCA 1981 Schedule 1

Biodiversity Steering Group Middle List

Species of Conservation Concern Red List

Species of Conservation Importance Table 2

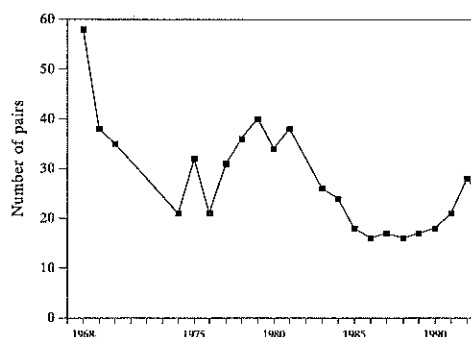


Figure 2.22.1 Maximum possible numbers of breeding Red-necked Phalaropes in the UK between 1968 and 1993 (from Everett, 1971; Sharrock *et al.*, 1982, 1983; Spencer *et al.*, 1986a, 1986b, 1988a, 1988b, 1989, 1990, 1991, 1993; Ogilvie *et al.*, 1994, 1995, 1996a).

The Red-necked Phalarope has a circumpolar breeding distribution encompassing tundra wetlands in the low-arctic and subarctic (Cramp & Simmons, 1983; Hagemeyer & Blair, 1997). The UK population is at the southern edge of its range and may be a relict from early post-glacial times (von Haartman *et al.*, 1963-1972). The majority of the European population winters pelagically in the Arabian Sea (Hagemeyer & Blair, 1997).

The Red-necked Phalarope has undergone a marked population decline in the UK since the 19th century, when it was believed to be widely distributed over the Scottish Islands and much of the mainland (Everett, 1971; Holloway, 1996). Over this century, however, it has become increasingly confined to strongholds in the Outer Hebrides and Shetlands. RSPB surveys revealed that the population had fallen to 54-65 pairs in 1968 and to c.45 in 1970, the majority in the Shetlands (Everett, 1971). The population has fallen further since then and is now largely confined to the island of Fetlar (Fig. 2.22.1; Batten *et al.*, 1990; Ogilvie *et al.*, 1996b). Thirty-eight breeding males were recorded in the UK in 1995 (Stone *et al.*, 1997). The Red-necked Phalarope has simultaneously declined in Ireland. Up to 50 pairs were present at one 'colony' in Co. Mayo in 1905, but numbers decreased in following decades. Seven to 10 pairs were present in Ireland between 1968 and 1970 (Everett, 1971), but only one pair bred in 1988 (Batten *et al.*, 1990; Gibbons *et al.*, 1993).

Much larger populations of Red-necked Phalaropes occur elsewhere in Europe. Hagemeyer and Blair (1997) estimated a total population of 65,500-94,300 pairs, although this is perhaps conservative – Piersma (1986) proposed that there were 50,000-100,000 pairs in Iceland alone and 50,000 pairs in Sweden. There is no clear information on population trends, however. In Finland, where 15,000 pairs may breed (Piersma, 1986), numbers are reported to fluctuate greatly (Grenquist, 1965). Only the small population in the Faroes is known to be in decline (Cramp & Simmons, 1983). Estimates of the Red-necked Phalarope's breeding populations are complicated by its variable mating system. Females are usually monogamous, but occasionally successively polyandrous (Hildén & Vuolanto, 1972; Reynolds, 1987; Whitfield, 1989).

Factors affecting Populations

The declines seen in the British and Irish populations of Red-necked Phalaropes at the turn of the century have been attributed largely to collecting (Everett, 1971). More recent declines, however, are probably a consequence of habitat change and the populations' isolation. In the UK, Red-necked Phalaropes breed in marshland sites with open water and emergent vegetation. In Shetland, they are associated with flooded peat cuttings, but in the Outer Hebrides are found near machair pools in hay fields and rough pasture (Everett, 1971; Gibbons *et al.*, 1993). Many breeding sites have been lost this century through drainage, notably in the Outer Hebrides (Everett, 1971). Other sites have been lost through vegetational succession as a result of the drying out of marshes and under-grazing (Batten *et al.*, 1990).

Recently the UK population has been restricted to between two and five localities (Ogilvie *et al.*, 1994, 1995, 1996a, 1996b) and is thus at great risk of extinction. The Red-necked Phalarope exhibits strong philopatry (Colwell *et al.*, 1986) and it is unlikely that there is much immigration into the UK from other populations. The future of the species in this country, therefore, is largely dependent upon the conservation of birds on these remaining sites.

Present and Future Research

Many traditional Red-necked Phalarope breeding sites are now protected as SSSIs or RSPB reserves and are thus accurately monitored. Fetlar has been an RSPB reserve since 1973. Other former or potential sites should also be monitored so that the UK population can be accurately assessed annually (SNH).

Research by the RSPB in the Shetlands and the Outer Hebrides has identified many of the habitat characteristics of phalarope breeding sites (Yates *et al.*, 1983). All sites had a mix of open pools, emergent vegetation (often bottle sedge *Carex rostrata*) and marsh and had neutral or base-rich water. Although areas of open water were not extensive, birds preferred larger pools for courtship and mating. Feeding was mostly restricted to vegetated areas. This and further RSPB research has also highlighted many of the threats to nesting birds. These include human disturbance, trampling from grazing and drinking cattle, flooding, cold and wet weather, and predation, particularly by Arctic Skuas *Stercorarius parasiticus*.

Management plans have been put into operation on RSPB reserves and at SSSIs using the results of the above research. Cattle are excluded from breeding sites during the nesting season, but grazing at other times of the year helps to maintain the vegetation structure. It is also vital to keep control over water levels and to maintain the mosaic of pools. By managing their Fetlar reserve in this way and by increasing the available habitat the RSPB has helped to increase the population of phalaropes at the site. Such management practices should be extended to other former sites (SNH).

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2.23 Bullfinch *Pyrrhula pyrrhula*

Status and Population Trends

Current European Population 2,800,000-3,900,000 pairs

SPEC None

EC Birds Directive

Berne Convention Appendix III

WCA 1981 Schedule 3, Part 1

Wildlife (NI) Order 1985

Biodiversity Steering Group Middle List

Species of Conservation Concern Red List

Species of Conservation Importance Table 3

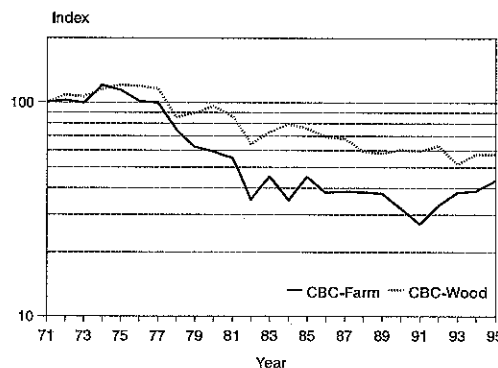


Figure 2.23.1 Population trends from the Common Birds Census

The Bullfinch's breeding distribution extends across the Palearctic region from western Europe to Japan and Kamchatka (Cramp & Perrins, 1994a; Hagemeyer & Blair, 1997). In Europe the species is predominantly found in the north and west, although populations also occur in forested parts of the Carpathian and Bulgarian/Greek mountains (Hagemeyer & Blair, 1997). Bullfinches in the UK may be resident or dispersive, their movements dependent upon food supplies (Summers, 1979). Populations in northern and eastern Europe are partially migratory (Savolainen, 1987). The subspecies *pileata* is endemic to Britain and Ireland.

The Bullfinch is a widespread and common species in the UK and is absent only from the Isle of Man, Outer Hebrides, Orkneys, Shetlands, Tiree and Coll (Gibbons *et al.*, 1993). Its numbers this century peaked in the 1950s (Newton, 1993), but over the last 20 years the species has shown a sharp population decline. CBC data show that the population fell by 57% between 1971 and 1995, by 75% on farmland alone and by 56% in woodland (Crick *et al.*, 1997; Fig. 2.23.1). Likewise, CES data indicate a 24% decline in catches of adults between 1983 and 1995. The BBS recorded a 17% decline between 1994 and 1996 (Gregory *et al.*, 1997). Breeding atlas data show that the Bullfinch's range in Britain contracted by 7% between 1968-72 and 1988-91, with absences now apparent in Scotland and the Fens (Gibbons *et al.*, 1993).

The British population was estimated at 190,000 territories in 1988-91 and that in Ireland at 100,000 (Gibbons *et al.*, 1993). Together they form almost 10% of the European population (Hagemeyer & Blair, 1997). Declines were also recorded in Denmark and Sweden in the early 1980s, but, in the Netherlands, there has recently been a small increase in numbers (Hustings, 1988).

Factors affecting Populations

The Bullfinch is a bird of woodland edge and scrubland and is often found along hedgerows and in gardens and churchyards. It predominantly feeds on the seeds and fruits of woody and herbaceous plants, notably those of ash *Fraxinus excelsior*, wych elm *Ulmus glabra*, birch *Betula*, common nettle, bramble and various docks *Rumex* (Newton, 1967, 1972; Gatter, 1989). In late winter and spring, however, it increasingly turns to tree buds and in fruit-growing regions can be a serious pest (Newton, 1964, 1993; Greig-Smith & Wilson, 1984). For this reason, thousands of Bullfinches used to be trapped and killed annually (Newton, 1972, 1993). An analysis of BTO ringing data revealed that mortality rates were greater for females and that this was perhaps because they were preferentially killed to minimize breeding rates (Dobson, 1987). In spite of the large-scale trapping of the species, however, there is little evidence to suggest that populations have been affected. In Herefordshire, for example, numbers in a 9 km² area remained stable over a five year period even though 200 birds were trapped annually to prevent damage to an apple orchard (Evans, 1984). Recent research suggests that less fruit would be lost if birds were encouraged to spread their feeding so that fewer individual trees would suffer concentrated attacks (Greig-Smith, 1987). Trapping under general licence has only been allowed in Kent since October 1996.

It is possible that the former abundance of the Bullfinch and its subsequent decline may be associated with the fortunes of one of its main predators, the Sparrowhawk (Newton, 1993). In the 1950s, Sparrowhawk numbers were low, following the introduction of organochlorine pesticides in agriculture, but with the reduction in their use, Sparrowhawk numbers have recovered (Newton & Haas, 1984). There is, however, no real evidence to support this theory.

The Bullfinch's decline is most likely to be related to loss of habitat and the intensification of agriculture (Marchant *et al.*, 1990). The removal of trees and hedgerows has resulted the loss of nesting habitat and food sources. Such changes may have affected breeding performance - although nest losses decreased significantly between 1971 and 1995, and brood sizes have decreased since the early 1980s (Crick *et al.*, 1997; Fig. 2.23.2). Populations may have also been affected by the loss of winter food sources, which have been reduced by herbicides and the loss of winter stubbles. Bullfinches do not forage far from woods or hedgerows, however, and are more confined to field margins than other farmland species (Newton, 1993).

Present and Future Research

The Bullfinch's status is currently monitored by the BBS and CES scheme and breeding productivity assessed by the CES scheme and the NRS (BTO, JNCC, RSPB). A survey for monitoring winter populations should also be considered (CCW, EHS, EN, JNCC, SNH). Further ecological research should identify appropriate habitat management for the species (CCW, EHS, EN, JNCC, SNH). Ongoing BTO/RSPB research aims to assess the role of Sparrowhawk predation in the decline of the species.

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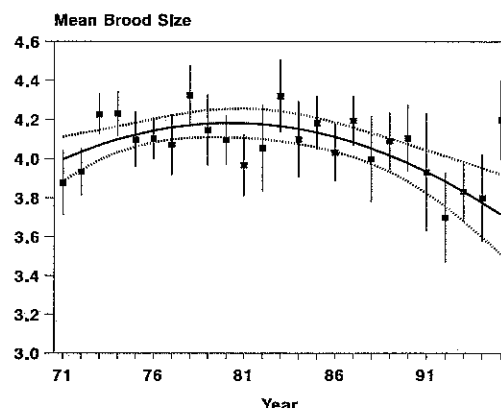


Figure 2.23.2 Long-term trend from the Nest Record Scheme

2.24 Roseate Tern *Sterna dougallii*

Status and Population Trends

Current European Population 1610 pairs

SPEC Category 3

SPEC Status Endangered (large decline; < 2,500 pairs)

EC Birds Directive Annex I

Berne Convention Appendix II

WCA 1981 Schedule 1

Wildlife (NI) Order Schedule 1

Biodiversity Steering Group Middle List

Species of Conservation Concern Red List

Species of Conservation Importance Table 2

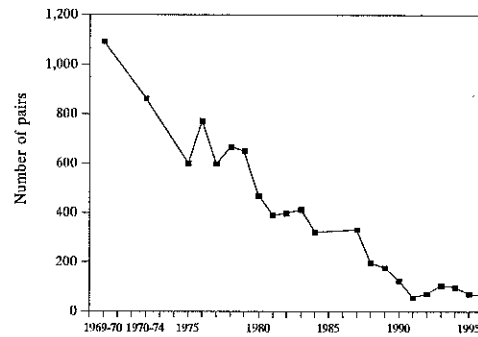


Figure 2.24.1 Maximum possible numbers of breeding Roseate Terns in the UK from 1969 to 1996 (from Thomas, 1982; Thomas *et al.*, 1989; Ogilvie *et al.*, 1996b; RSPB, unpublished data).

Outside of Europe, the Roseate Tern's breeding distribution is largely tropical. It occurs in the Caribbean and west Atlantic, the Indian Ocean, east Asia, Australia and Melanesia (Gochfield, 1983; Avery, 1991; Hagemeyer & Blair, 1997). Only a small proportion (c.3%) of its global population of 25,000 to 50,000 pairs breeds in Europe, the majority of which are found in the Azores (Gochfield, 1983; del Nevo *et al.*, 1993; Tucker & Heath, 1994; Hagemeyer & Blair, 1997). The British and Irish populations are at the northern edge of the species' range. British and Irish Roseate Terns winter from south-west Europe to west Africa, but are concentrated on the Ghanaian coast (Hepburn, 1986; Everett *et al.*, 1987; Avery, 1991).

The numbers of Roseate Terns in Britain and Ireland have fluctuated widely over the last 100 years. At the turn of the century, the species may have been near to extinction, probably due to egg-collection and hunting (Avery, 1991; Holloway, 1996). A rise in numbers in the first half of this century saw the population peak at around 3000 to 3500 pairs in the 1950s (Parslow, 1967). The population has been in decline since then, however, and since 1969 numbers have fallen by about 80% (Fig. 2.24.1; Avery, 1991). The species is still present at most of its English and Welsh colonies, albeit in low numbers, but has been lost from many sites in Scotland and Ireland (Avery, 1991; Gibbons *et al.*, 1993). It is now concentrated at one Irish colony at Rockabill, Co. Dublin.

The Roseate Tern is also in decline in France (Hagemeyer & Blair, 1997) and the Azores, where the population fell from 100 pairs in 1989 to 550 in 1994 (Ramos *et al.*, 1995).

Factors affecting Populations

The Roseate Tern is a colonial nester and often associates with other species, including the Common Tern *Sterna hirundo* and Sandwich Tern *Sterna sandvicensis*. Colonies are usually situated on coastal islets, but may also occur on islands in freshwater coastal lagoons (Avery, 1991; Tucker & Heath, 1994). Nest sites are concealed amongst rocks or vegetation and in some colonies, where such sites are limited, the species has been encouraged to use nest-boxes (Avery, 1991; Avery & del Nevo, 1991; Casey *et al.*, 1995). The size of Roseate Tern colonies may be limited by competition for nest sites with other species.

The declines seen at the end of the 19th century are believed to be associated with human persecution, notably egg-collection and hunting (Avery, 1991; Holloway, 1996). Nesting Roseate Terns are also sensitive to human disturbance, and avian and ground predators (e.g. Peregrines *Falco peregrinus*, gulls, rats, mustelids and foxes) and may move between colonies as a result (del Nevo *et al.*, 1990; Avery, 1991; Avery & del Nevo, 1991; Brookhouse, 1992; Moralee, 1992). In Ireland, for example, human disturbance and predation from rats, minks, foxes and gulls at the Lady's Island Lake colony in Co. Wexford caused the population to fall from 275 pairs in 1981 to 10 pairs in 1982. Many of these birds probably moved to Rockabill, where numbers increased substantially throughout the 1980s (Avery, 1991; Casey *et al.*, 1995). It is believed that Roseate Terns from declining colonies in Northern Ireland and perhaps from other UK sites may have also moved to Rockabill (Thomas *et al.*, 1989; Avery, 1991). This one colony presently holds 557 of the estimated 741 pairs of Roseate Terns in Britain and Ireland (RSPB unpublished data). It is improbable that emigration to more distant colonies has led to the species' decline in north-west Europe, however (Avery, 1991).

In its west African wintering quarters, the Roseate Tern is under threat from trapping for sport and food. Birds are snared with hooks or nooses or, in some cases, simply knocked from the air with stones (Dunn &

Mead, 1982; Anon., 1990; Ntiamoa-Baidu, 1991). It is unlikely that the species' decline in Europe is a consequence of this practice, however, as Roseate Terns form only a small proportion of the total tern population of west Africa and trapping is not directed just at them. There is no reason to believe, either, that Roseate Terns are more susceptible to trapping than any other species (Avery, 1991). The Roseate Tern's peak presence in west Africa in September and October coincides with the arrival in inshore waters of large stocks of small fish (e.g. *Sardinella*) (Dunn & Mead, 1982; Avery, 1991). It is possible that the decline of these fish stocks, which may be related to long-term changes in sea-surface temperatures, could be affecting populations.

Present and Future Research

Roseate Terns are currently monitored in the UK through the Seabird Monitoring Programme (JNCC). Although the censusing of tern colonies may be problematic, notably due to the confusion of species and the difficulty of avoiding disturbance (Avery, 1991), the status of the Roseate Tern in both the UK and the Republic of Ireland is currently well known. Many remaining colonies are now protected as reserves and research by the RSPB and other organisations has identified many of the potential causes of disturbance and nesting failure (Avery & del Nevo, 1991; Brookhouse, 1992; Moralee, 1992; Casey, 1995). There is a need for improved monitoring of breeding success (clutch sizes, hatching and fledging success) and survival rates (CCW, EHS, EN, SNH, JNCC), although there are no signs that breeding success in UK and Irish colonies, in the absence of predators, is low (Batten *et al.*, 1990; Casey *et al.*, 1995). An RSPB colour-ringing scheme aims to provide information on recruitment rates to breeding colonies and to establish whether birds regularly shift colonies. It is also hoped that this will provide better information on the wintering quarters of UK Roseate Terns. At present, little is known of their distribution after December (Avery, 1991).

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2.25 Turtle Dove *Streptopelia turtur*

Status and Population Trends

Current European Population 2,000,000-2,400,000 pairs

SPEC Category 3

SPEC Status Declining (moderate decline)

EC Birds Directive Annex II/2

Berne Convention Appendix III

WCA 1981

Wildlife (NI) Order Schedule 1

Biodiversity Steering Group Middle List

Species of Conservation Concern Red List

Species of Conservation Importance Table 2

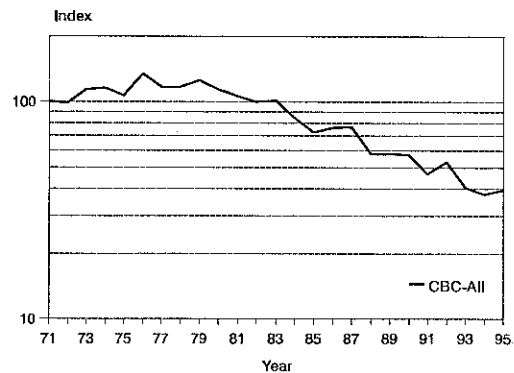


Figure 2.25.1 Population trend from the Common Birds Census

The Turtle Dove's breeding distribution extends across the Palearctic region from the Canary Islands through Europe and North Africa to central Asia. In Europe, the Turtle Dove is predominantly a southern species and is largely absent from Scandinavia (Cramp, 1985; Hagemeijer & Blair, 1997). The Turtle Dove is a trans-Saharan migrant, the western European population wintering in the savannas of Senegal and Mali (Cramp, 1985; Jarry, 1992).

The Turtle Dove was formerly widespread across southern and eastern England, its range extending into the Welsh and Scottish borders. The population was probably at a long-term high in the mid-1970s, but since then has undergone a sharp decline. CBC data indicate an overall decline of 69% between 1971 and 1995 and declines of 79% on both farmland and woodland (Crick *et al.*, 1997; Fig. 2.25.1). BBS data suggest that this decline may now be slowing (Gregory *et al.*, 1997). Breeding atlas data show that the Turtle Dove's range in Britain contracted by 25% between 1968-72 and 1988-91, with absences now apparent in the south-west, Wales and north-east of England (Gibbons *et al.*, 1993). At the time of the second atlas, there were an estimated 75,000 territories in Britain.

Turtle Doves have also declined in other parts of Europe, for example in France and Romania where populations have fallen by over 50% (Tucker & Heath, 1994; Yeatman-Berthelot & Jarry, 1995; Hagemeijer & Blair, 1997). Other populations have been more stable, however, including those in Austria, Belarus, Hungary, Italy, Poland, Russia, Slovakia and Slovenia. Large populations still occur in France and, in particular, Spain (Hagemeijer & Blair, 1997).

Factors affecting Populations

The Turtle Dove feeds on wild seeds and cultivated grain and in the UK is strongly associated with lowland agriculture (Cramp, 1985). Preferred habitat includes hedgerows, scrub, spinneys and the borders of larger woodlands (Tucker & Heath, 1994; Calladine *et al.*, 1997). The Turtle Dove's distribution has previously been linked with that of fumitory *Fumaria*, but whilst the seeds of this plant do form a large part of its diet (Murton *et al.*, 1964), the species is not dependent upon them (Sitters, 1988; Gibbons *et al.*, 1993).

The Turtle Dove's decline in Europe is in part probably related to the degradation of breeding habitat and the intensification of agriculture. The destruction of hedgerows has resulted in the loss of many nesting sites, whilst the widespread use of herbicides has substantially reduced its food supply (Marchant *et al.*, 1990). The change from hay to silage and associated earlier mowing, as well as the increased use of fertilizers on grassland has probably also reduced the availability of weeds (Gibbons *et al.*, 1993). Although breeding success is low relative to other species (Murton, 1968), NRS data provide no evidence to suggest that nesting success has decreased or that the predation of eggs, the main cause of breeding failures (Murton, 1968; Bijlsma, 1985; Pieró, 1990), has increased (Crick *et al.*, 1997). Overall productivity may have fallen, however. In a study in East Anglia in 1996, pairs were only occasionally found to lay second clutches (Calladine *et al.*, 1997) in contrast to previous studies (Murton, 1968; Pieró, 1990; Nankinov, 1994; Fontoura & Dias, 1996). It is probably unlikely that Turtle Doves have suffered from interspecific competition with the Collared Dove *Streptopelia decaocto* (Goodwin, 1987; Duckworth, 1992).

Another possible cause of the Turtle Dove decline is the severe drought which has affected its African wintering grounds (Jarry & Baillon, 1991; Jarry, 1992). This has not only reduced the Turtle Dove's food supply (Skinner, 1987) but also led to a scarcity of drinking water, an important daily requirement for the

species (Hagemeijer & Blair, 1997). The wintering habitat is additionally being degraded by the loss of the acacia forests, which are being cut for charcoal (Jarry & Baillon, 1991).

Tens of thousands of Turtle Doves are killed annually by hunters, both on the wintering grounds and on migration (El Mastour, 1988; Razin & Urcun, 1992; Urcun, 1993) and this may have accelerated the decline in Europe (Tucker & Heath, 1994).

Present and Future Research

The Turtle Dove's status is currently monitored by the BBS and breeding success by the NRS (BTO, JNCC, RSPB). Analysis of NRS data from the 1960s may help indicate whether long-term changes in productivity are associated with the population decline (CCW, EN, JNCC). A pilot EN study was carried out in East Anglia in 1996 in order to assess the feasibility of a future detailed study into the summer autoecology and conservation needs of the species (Calladine *et al.*, 1997). This stressed the importance of radio-tracking to assess habitat requirements and to locate nesting, roosting and feeding sites. It also highlighted the need for an assessment of dietary preferences. Current diet and habitat use should also be compared to studies carried out in the 1960s (e.g. Murton *et al.*, 1964; Murton, 1968) (CCW, EN).

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